R-08-35

Safety assessment of a KBS-3H spent nuclear fuel repository at Olkiluoto

Complementary evaluations of safety

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L Johnson, Nagra

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Svensk Kärnbränslehantering AB
Swedish Nuclear Fuel
and Waste Management Co
Box 250, SE-101 24 Stockholm
Tel +46 8 459 84 00

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Abstract

The KBS-3H design is a variant of the more general KBS-3 method for the geological disposal of spent nuclear fuel in Finland and Sweden. In the KBS-3H design, multiple assemblies containing spent fuel are emplaced horizontally in parallel, approximately 300 m long, slightly inclined deposition drifts. The copper canisters, each with a surrounding layer of bentonite clay, are placed in perforated steel shells prior to deposition in the drifts; the assembly is called the "supercontainer". The other KBS-3 variant is the KBS-3V design, in which the copper canisters are emplaced vertically in individual deposition holes surrounded by bentonite clay but without steel supercontainer shells.

SKB and Posiva have conducted a Research, Development and Demonstration (RD&D) programme over the period 2002–2007 with the overall aim of establishing whether KBS-3H represents a feasible alternative to KBS-3V. As part of this programme, the long-term safety of a KBS-3H repository has been assessed in the KBS-3H safety studies. In order to focus the safety studies, the Olkiluoto site in the municipality of Eurajoki, which is the proposed site for a spent fuel repository in Finland, was used as a hypothetical site for a KBS-3H repository.

The present report is part of a portfolio of reports discussing the long-term safety of the KBS-3H repository. The overall outcome of the KBS-3H safety studies is documented in the summary report, "Safety assessment for a KBS-3H repository for spent nuclear fuel at Olkiluoto".

The purpose and scope of the KBS-3H complementary evaluations of safety report is provided in Posiva's Safety Case Plan, which is based on Regulatory Guide YVL 8.4 and on international guidelines on complementary lines of argument to long-term safety that are considered an important element of a post-closure safety case for geological repositories.

Complementary evaluations of safety require the use of evaluations, evidence and qualitative supporting arguments that lie outside the scope of the quantitative safety assessment. These arguments include:

- Support from natural and anthropogenic analogues for both key process understanding and total system performance.
- Comparison of the methodology and results with the earlier TILA-99 and SR-Can safety assessments, as well as other international safety assessments, to ensure completeness, consistency and reasonableness of the present assessment.
- Use of safety indicators other than dose and activity to avoid uncertainties in future human lifestyles and also in geological processes on very long timescales.
- Consideration of the calculation results from a wider perspective to consider significance of their impact compared to other risks.

Sammanfattning

KBS-3H-utformningen är en variant av den mera generella KBS-3-metoden för geologisk deponering av använt kärnbränsle i Finland och i Sverige. I KBS-3H-utformningen deponeras flera kapslar med använt kärnbränsle horisontellt i ca 300 m långa deponeringshål. Varje kopparkapsel omges av bentonitlera och placeras i en perforerad stålcylinder, paketet kallas ”supercontainer”, innan de deponeras i deponeringshål. I referensutformningen KBS-3V deponeras kopparkapslarna i individuella deponeringshål omringade av bentonit men utan stålcylinder.

Posiva och SKB har utfört ett gemensamt forsknings-, utvecklings- och demonstrationsprogram (FUD) under 2002–2007 med det övergripande målet att utvärdera om KBS-3H kan utgöra ett alternativ till referensalternativet KBS-3V. Säkerhetsstudierna, som utförts som en del av detta program, omfattar en säkerhetsanalys av en preliminär utformning av KBS-3H för ett slutförvar för använt kärnbränsle i Olkiluoto, som är den föreslagna platsen för ett slutförvar för använt kärnbränsle i Finland.

Denna rapport är en av ett flertal rapporter som diskuterar långtidssäkerheten för ett slutförvar baserat på en KBS-3H-utformning. De övergripande resultaten av KBS-3H-säkerhetsstudierna har sammanfattats i rapporten ”Safety assessment for a KBS-3H spent nuclear fuel repository at Olkiluoto-Summary report”.

Avsikten och omfattningen för denna KBS-3H-rapport om kompletterande utvärdering av säkerheten har definierats i Posiva’s Safety Case-plan, som baserats på myndighetsföreskrifter YVL guide 8.4 och på internationella riktlinjer för kompletterande argument för långtidssäkerheten som anses vara viktiga element för upprättandet av en säkerhetsanalys avseende perioden efter förslutning av ett slutförvar.

Den kompletterande utvärderingen av säkerheten kräver användning av utvärderingar, bevis och kvalitativt stödjande argument som ligger utanför räckvidden för den kvantitativa säkerhetsanalysen. Dessa argument omfattar:

Stöd från naturliga och antropogena analoger för förståelse av huvudprocesser och hur hela systemet fungerar.

Jämförelse av metodik och resultat från de tidigare säkerhetsanalyserna TILA-99 och SR-Can, såväl som med andra internationella säkerhetsanalyser, för att försäkra fullständighet, följdriktighet och rimlighet av den utförda analysen.

Användandet av säkerhetsindikatorer andra än dos och aktivitet för att undvika osäkerheter i framtida mänskliga livsstilar samt även i geologiska processer i mycket långa tidsskalor.

Övervägande av resultaten från radionuklidtransport och dosanalys i ett vidare perspektiv för att bedöma betydelsen av deras inverkan jämfört med övriga risker.

Denna rapport finns även tryckt i Posiva rapportserie POSIVA 2007-10.

Nyckelord: KBS-3H, horisontell deponering, safety case, långtidssäkerhet, använt kärnbränsle, slutdeponering, kompletterande utvärdering av säkerheten.

Contents

Foreword

This study was coordinated by Margit Snellman (Saanio & Riekkola Oy) on behalf of Posiva Oy. The progress of the study was supervised by the KBS-3H Review Group consisting of Aimo Hautojärvi (Posiva), Jukka-Pekka Salo (Posiva), Marjut Vähänen (Posiva), Barbara Pastina (Saanio & Riekkola Oy), Margit Snellman (Saanio & Riekkola Oy), Jorma Autio (Saanio & Riekkola Oy), Stig Pettersson (SKB), Erik Thurner (SKB), Börje Torstenfelt (Swedpower), Lennart Börgesson (Clay Technology) and Lawrence Johnson (Nagra). This report was largely written by Fiona Neall (Neall Consulting Ltd), with contributions from Barbara Pastina (Saanio & Riekkola Oy), Paul Smith (SAM), Peter Gribi (S+R Consult), Margit Snellman (Saanio & Riekkola Oy) and Lawrence Johnson (Nagra). Thomas Hjerpe (Saanio & Riekkola Oy) provided input on text related to the biosphere and risks related to radiation exposure. Other members of the KBS-3H Review Group also contributed in cross-checking and completing the report. Christine Bircher (Nagra), Aline Playfair (Nagra), Marina Molin (Adlibrakonsult AB) and Heini Laine (Saanio & Riekkola Oy) provided editorial support.

The report was reviewed in draft form by the following individuals: Per-Eric Ahlström (SKB, Sweden), Johan Andersson (Streamflow AB, Sweden), Jordi Bruno (Enviros Spain LS, Spain), Allan Hedin (SKB, Sweden), Alan Hooper (formerly Radioactive Waste Management Directorate of the Nuclear Decommissioning Authority, UK; currently Alan Hooper Consulting Limited, UK), Nuria Marcos (Saanio & Riekkola Oy, Finland), Fredrik Vahlund (SKB, Sweden), Fred Karlsson (SKB, Sweden) and Stratis Vomvoris (Nagra).

1 Introduction

1.1 The KBS-3H design

The KBS-3H design is a variant of the more general KBS-3 method for the geological disposal of spent nuclear fuel in Finland and Sweden. In the KBS-3H design, multiple canisters containing spent fuel are emplaced horizontally in parallel, approximately 300 m long, slightly inclined deposition drifts. The copper canisters, each with a surrounding layer of bentonite clay, are placed in perforated steel shells prior to deposition in the drifts. The other KBS-3 variant is the KBS-3V design, in which the copper canisters are emplaced vertically in individual deposition holes surrounded by bentonite clay and without steel supercontainer shells. The KBS-3H and KBS-3V alternative designs are illustrated in Figure 1-1.

The copper canister and its contents are the same as in KBS-3V. The package comprising a copper canister with a surrounding layer of bentonite clay (the reference bentonite for this report is MX-80) in a steel shell is referred to as a “supercontainer” (Figure 1-2).

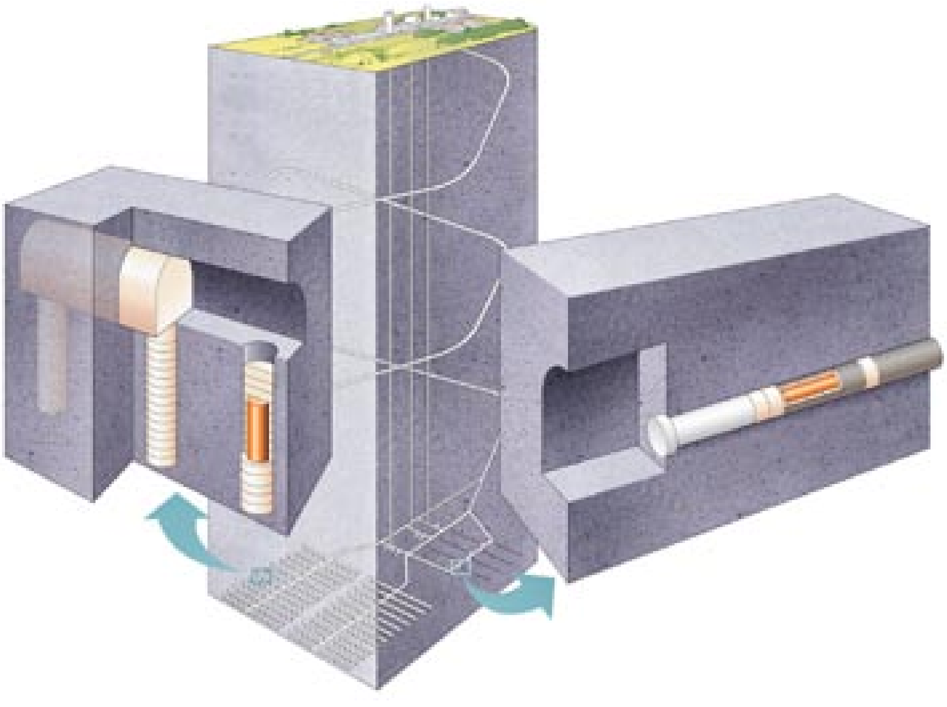

Figure 1-1. *The KBS-3V (left) and KBS-3H (right) alternatives of the KBS-3 spent fuel disposal method.*

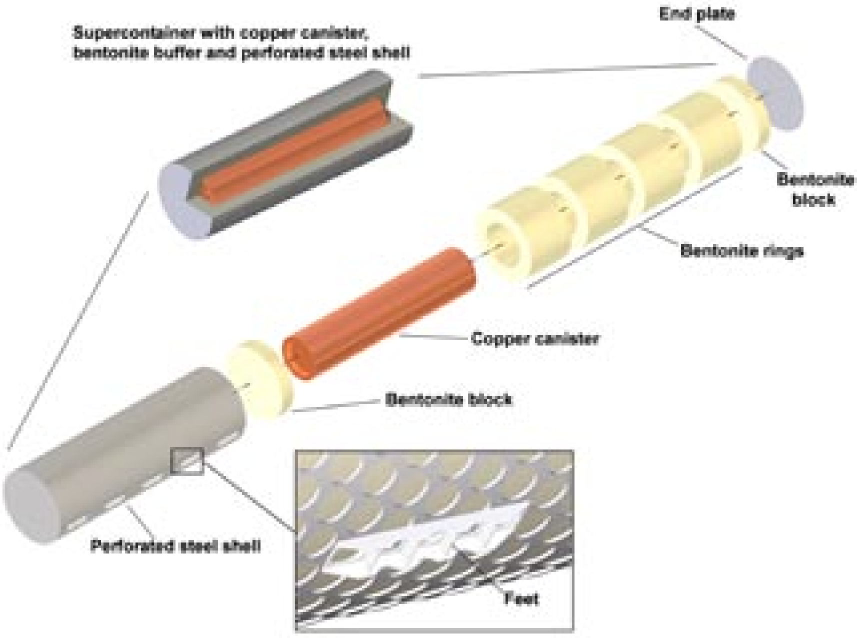

Figure 1-2. *The supercontainer with buffer and copper canister.*

The supercontainers are deposited coaxially in the drifts, supported on steel feet to leave an annular gap to the drift wall (about 4 cm). Bentonite distance blocks separate the supercontainers, one from another, along the drift. One supercontainer and one distance block are referred to as "supercontainer unit". The bentonite emplaced as part of the supercontainers and the bentonite distance blocks are jointly termed the "buffer", in contrast to the KBS-3V terminology in which the buffer refers to only the bentonite surrounding the canister. The KBS-3H drift and its components are shown in Figure 1-3. A section of drift with two supercontainers and one distance block is shown in Figure 1-4; the main dimensions are also indicated. Void spaces around the supercontainers and distance blocks will become filled with bentonite as the drift saturates and the bentonite swells, although the rate at which this occurs may vary considerably along the drift due the heterogeneity of the rock and the variability of water inflow, as discussed in the KBS-3H Evolution Report /Smith et al. 2007a/.

From the central tunnel, the initial section of each deposition drift (before the drift end plug) is a 15-metre long, wider section of the tunnel (with a 50 m^2 cross section) that hosts the deposition equipment for supercontainers and distance blocks. This section is called the "deposition niche" and is considered part of the drift for material inventory purposes. The maximum length of the drifts is 300 m, the estimated minimum length is 100 m and the average length is about 272 m, based on site-specific features /Autio et al. 2007/. In the current design, the drifts are dead-ended, i.e. there is no access tunnel on the other end.

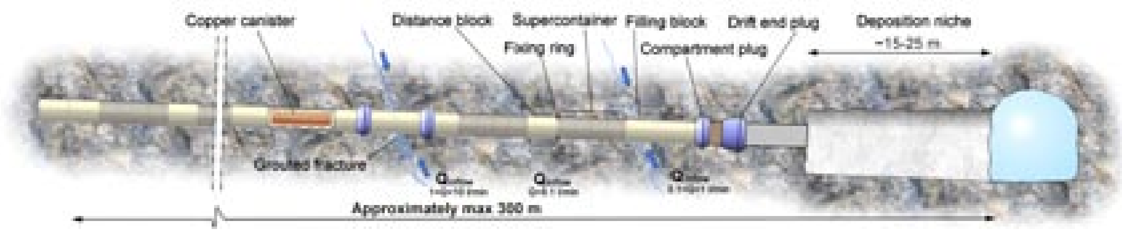

Figure 1-3. *Illustration of a generic KBS-3H drift showing one canister in copper colour for better visualisation. At one end of the drift, a wider area (deposition niche) hosts the deposition equipment while the other end of the drift is closed off. The components are described in Section 1.6.4 of the Process Report /Gribi et al. 2007/.*

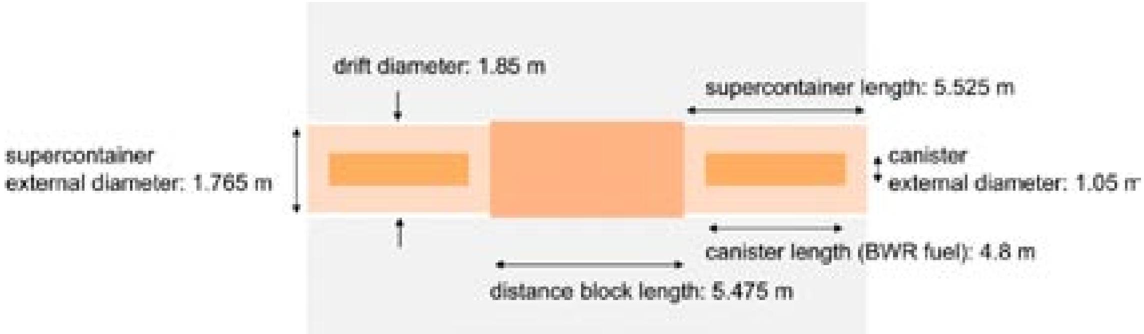

Figure 1-4. *Schematic illustration of the KBS-3H design, showing a section of a deposition drift with two supercontainers separated by a distance block (for additional details and references see Appendix A of the Process Report: /Gribi et al. 2007/). The 11 m canister spacing is for Finnish BWR spent fuel.*

1.2 KBS-3H long-term safety studies

SKB and Posiva have conducted a Research, Development and Demonstration (RD&D) programme over the period 2002–2007 with the overall aim of establishing whether KBS-3H represents a feasible alternative to KBS-3V, which is the reference alternative for both SKB and Posiva.

There are differences between the Olkiluoto site in the municipality of Eurajoki, which is the proposed site for a spent fuel repository in Finland, and the sites under consideration in Sweden. There are also different fuel types for disposal, leading to differences between the characteristics and inventories of canisters. There are also a number of design variants under consideration for implementing KBS-3H. Therefore, in order to focus the KBS-3H studies of long-term safety[1], the Olkiluoto site was used for the purpose of the studies. The reference fuel is the Finnish BWR fuel from Olkiluoto 1&2 reactors and the reference design for KBS-3H implementation is the Basic Design, as described in the Design Description 2006 /Autio et al. 2007/[2]. Releases of radionuclides from the repository are also compared to Finnish regulatory guidelines, with only brief comments about the Swedish regulatory context.

Specific high-level questions addressed by the KBS-3H safety studies are:

- Are there safety issues specific to KBS-3H with the potential to lead to unacceptable radiological consequences?
- Is KBS-3H promising at a site with the broad characteristics of Olkiluoto from the long-term safety point of view?

Due to the limitations in the scope of the KBS-3H safety studies, the following questions are not currently addressed:

- Is KBS-3H more or less favourable than KBS-3V from a long-term safety point of view?
- Does the specific realisation of the KBS-3H repository design considered in the safety studies satisfy all relevant regulatory guidelines?

Regarding the first question, a comparative study of KBS-3H vs. KBS-3V is beyond the scope of the safety studies carried out to date. Regarding the second question, although the performance of a KBS-3H repository has been analysed in a number of cases representing alternative evolutions of the repository system and the results compared to Finnish regulatory guidelines, the

[1] The KBS-3H studies of long-term safety are referred to as the "KBS-3H safety studies" throughout this report but only the long-term safety is implied by this term.

[2] The Design Description 2006 also presents a number of preliminary design variants under consideration for implementing KBS-3H.

analyses have a number of limitations, as described in the KBS-3H Safety Assessment Summary Report /Smith et al. 2007c/. These limitations would have to be addressed before it could be judged whether all relevant regulatory guidelines are satisfied. Differences between the fuel, canisters and repository sites under consideration in Sweden and Finland will have to be taken into account when transferring the detailed findings of the current safety studies to a Swedish context.

The safety studies refer to long-term or post-emplacement safety, i.e. safety from the time of emplacement of the first canisters in the repository. Construction and operation of the repository drifts will continue over several decades following emplacement of the first canisters. Safety studies consider the evolution and performance in this period, as well as in the period subsequent to repository closure. The safety of the workforce and the public during construction, operation and closure of the repository (operational safety) is not included in the present safety studies and will be addressed in thenext phase of the KBS-3H programme.

Safety studies specific to KBS-3H are complemented by detailed studies of:

- The function of the buffer bentonite.
- Repository design and layout adaptation to the Olkiluoto site in Finland.
- Deposition equipment.
- The retrievability of the canister in KBS-3H.
- The comparative costs of the KBS-3H and KBS-3V designs.

These are intended to be sufficiently comprehensive that they can be used, along with the technical demonstration, environmental and cost studies, as a technical basis for a decision at the beginning of 2008 on whether or not to continue the development of KBS-3H. The main conclusions from these KBS-3H studies and answers to the high-level questions above will be presented in the KBS-3H Study Report /SKB/Posiva 2008/.

1.3 Reporting of KBS-3H safety studies

The several reports that document and support the safety studies of a KBS-3H repository at Olkiluoto are shown in Figure 1-5 (although some are common to the KBS-3H and KBS-3V designs and will be developed in the context of Posiva's KBS-3V programme).

The overall outcome of the KBS-3H safety studies is documented in the summary report, "Safety assessment for a KBS-3H repository for spent fuel at Olkiluoto" /Smith et al. 2007c/. The summary report is supported by a number of further high-level reports (those shown in Figure 1-5), one of which is the present Complementary Evaluations of Safety Report.

The geoscientific basis of the safety studies is provided in site reports /Posiva 2003a, 2005, Andersson et al. 2007/, including the present situation at, and past evolution of, the Olkiluoto site as well as disturbances caused by Onkalo[3]. Data from the most recent Olkiluoto Site Description 2006 /Andersson et al. 2007/ are used whenever possible in this report, although further work is required to incorporate this data fully in future safety assessments.

The engineering basis is provided by the reports on the characteristics of spent fuel /Anttila 2005/, canister design /Raiko 2005/, and repository design /Autio 2007, Autio et al. 2007/. The repository design report, Design Description 2006 /Autio et al. 2007/, presents some preliminary

[3] Onkalo is the Olkiluoto Underground Rock Characterisation facility for site-specific underground investigations. Onkalo has been under construction since mid-2004 and will serve as an access route to the repository and the first disposal tunnels are planned to be adjacent to Onkalo`s main characterisation level.

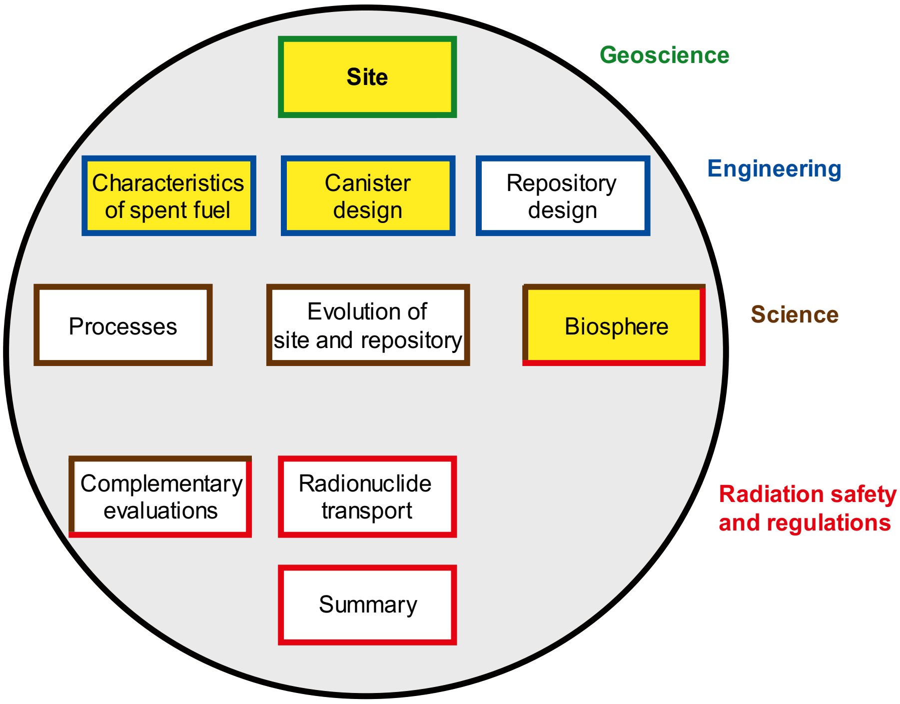

Figure 1-5. *The reporting structure for the KBS-3H safety studies 2007. The colours of the boxes indicate the areas covered by the reports (as listed on the right-hand side of the figure). Yellow filling indicates reports common to the KBS-3H and -3V safety studies.*

candidate designs based on KBS-3H. This report and all the other safety studies for KBS-3H are based on a preliminary design (called Basic Design) that was frozen at the beginning of 2007. Subsequent design studies are presented in the Design Description 2007 /Autio et al. 2008/. The repository design report also discusses long-term safety features, together with manufacturing and installation aspects of the buffer and backfill for KBS-3H.

The Process Report /Gribi et al. 2007/ provides a description of the main processes potentially affecting the long-term safety of the system. In contrast, the Evolution Report /Smith et al. 2007a/ provides a detailed description of the evolution of the repository in different time frames, based on the scientific information on the processes documented in the Process Report. From these reports, a set of calculation cases is defined. The calculations of radionuclide releases to the near field and the far field are documented in the Radionuclide Transport Report /Smith et al. 2007b/. Radiological safety and compliance with regulatory requirements are mainly dealt with in the Biosphere Analysis Report /Broed et al. 2007/, and in the Summary Report. Independent and less quantifiable lines of reasoning in support of long-term safety are presented in the present Complementary Evaluations of Safety Report.

These high-level reports are further supported by more detailed technical reports compiled for the KBS-3H safety studies, including thermal analyses /Ikonen 2003, 2005/, thermo-mechanical analyses /Lönnqvist and Hökmark 2007/, layout studies, based on analyses of data from the Olkiluoto site /Hellä et al. 2006/, discrete fracture network modelling of the site /Lanyon and Marschall 2006/, analyses of hydro-mechanical, chemical, gaseous and microbiological (HMCGB) processes related to the steel components /Johnson et al. 2005/, experimental and modelling studies on the interaction of iron and bentonite /Carlson et al. 2006, Wersin et al. 2007/, and solubility estimation in support of radionuclide release and transport calculations /Grivé et al. 2007/.

1.3.1 A difference analysis approach

In order to judge the feasibility of implementing the KBS-3H design from a long-term safety point of view, relevant safety issues must be understood as well for KBS-3H as they are for KBS-3V. There is, however, a broad scientific and technical foundation that is common to both designs and much of the work carried out by both Posiva and SKB in the context of KBS-3V is also directly applicable to KBS-3H. Thus, there is comparatively much more limited documentation that has been developed specifically relating to KBS-3H and this documentation focuses primarily on the differences identified between the KBS-3H and KBS-3V alternatives in a systematic "difference analysis" approach. The main differences between KBS-3H and KBS-3V are summarised in Appendix A.

1.3.2 Input data

A project decision was made not to prepare a separate data report (in contrast to the case of SR-Can, see /SKB 2006e/) and all the main data used in the reports of the KBS-3H safety studies are reported in Appendix A of the Process Report /Gribi et al. 2007/. Furthermore, in the Process /Gribi et al. 2007/, Evolution /Smith et al. 2007a/ and Radionuclide Transport /Smith et al. 2007b/ reports, additional data are derived as a result of modelling calculations for the cases where no other sources exist. Data used in this report are based on the information available at the time of report writing (2006–2007). Input data were selected on the basis of the preliminary design information presented in the KBS-3H Design Description 2006 /Autio et al. 2007/, laboratory data, field data, modelling, and calculations and, in some cases, expert judgment. The bases for data selection and assumptions used have been reported as far as possible in the Process Report. A more complete data report for both KBS-3H and -3V designs at Olkiluoto will be published at a later date.

1.4 Purpose and scope of this report

The purpose and scope of the KBS-3H complementary evaluations of safety report is outlined in Posiva's Safety Case[4] Plan 2005 /Vieno and Ikonen 2005/, which is based on Regulatory Guide YVL 8.4 (see Section 1.5 below) and on international guidelines on complementary lines of argument to long-term safety that are considered an important element of the safety case. An updated version of Posiva's safety case /Posiva 2008/ was published during the final stages of the present report's production.

Regulatory Guide YVL 8.4 /STUK 2001/ outlines the role and specific tasks of complementary considerations as follows: "*The importance to safety of such scenarios that cannot reasonably be assessed by means of quantitative analyses shall be examined by means of complementary considerations. They may include e.g. bounding analyses by simplified methods, comparisons with natural analogues or observations of the geological history of the disposal site. The significance of such considerations grows as the assessment period of interest increases, and the judgment of safety beyond one million years can mainly be based on the complementary considerations. Complementary considerations shall also be applied parallel to the actual safety analysis in order to enhance the confidence in results of the whole analysis or a part of it.*"

According to international guidelines on the safety case, "… *multiple lines of argument are useful for building a convincing safety case. Some lines of argument are more qualitative in nature than others, and there may be an emphasis on different types of argument and different indicators of performance and safety in different time frames*" /NEA 2002b/.

[4] According to the NEA and IAEA definition, a safety case is a synthesis of evidence, analyses and arguments that quantify and substantiate the safety, and the level of expert confidence in the safety, of a geological disposal facility for radioactive waste /NEA 2004a, IAEA/NEA 2006/.

Also, as has been more recently observed at the NEA Timescales Initiative /de Preter et al. 2006, NEA 2007a/: "... *complementary lines of argument are required, not only to compensate for increasing uncertainties affecting calculated releases at distant times but also to address other aspects of safety, especially continuing isolation, even at times beyond when quantitative safety assessments can be supported. Complementary arguments might be based, for example, on the absence of resources that could attract inadvertent human intrusion and on the geological stability of the site, with low rates of uplift and erosion*."

Concerning the timescales over which a safety case needs to be made, in /NEA 2002a/ it is stated that "… *the use of safety and performance indicators other than dose and risk can give indications of safety independent of both the limited predictability of the surface environment and, on a far longer timescale, the limited predictability of the geological environment*." /De Preter et al. 2006/ also note that: "*The argumentation for safety in the very long term is ... an issue of ongoing discussion that is likely to require a consideration of ethical principles, since it relates to our ability and responsibility to protect the environment in the very remote future*".

According to /NEA 2004a/, complementary evaluations of safety require the use of evaluations, evidence and qualitative supporting arguments that lie outside the scope of the other reports of the quantitative safety assessment. These arguments include, for example:

- Support from natural systems for both key process understanding and total system performance.
- Comparison of the methodology and results of safety cases made for other repository projects to ensure completeness, consistency and reasonableness of the present assessment.
- Simplified bounding analyses of extreme, unrealistic cases for scenarios not considered in the quantitative safety assessment.
- Use of safety indicators other than dose and activity to avoid uncertainties in future human lifestyles (e.g. food production and consumption) and also geological processes on very long timescales.
- Consideration of the calculation results from a wider perspective to consider significance of their impact compared to other risks.

These are entirely consistent with the arguments that are assembled in this report (see Section 1.6).

1.5 Finnish regulatory context

As stated above, the KBS-3H safety studies documents refer mainly to the Finnish regulatory context. The regulatory requirements for a spent fuel repository at Olkiluoto are set forth in the Government Decision on the safety of the disposal of spent nuclear fuel /STUK 1999/ and, in more detail, in Regulatory Guide YVL 8.4 issued by the Finnish regulator /STUK 2001/. These requirements are, however, currently under revision. A detailed discussion of regulatory requirements related to the safety case, including dose and radionuclide release constraints in different time frames, is given in Posiva's TKS-2006 report describing the programme for research, development and technical design /Posiva 2006/. Some key points relevant to the present report are summarised below.

Finnish regulations distinguish between the "environmentally predictable future" (lasting "several thousand years"), during which conservative estimates of dose must be made, and the "era of large-scale climate changes", when periods of permafrost and glaciations are expected, and radiation protection criteria are based on constraints on nuclide-specific activity fluxes from the geosphere ("geo-bio flux" constraints). Posiva's interpretation of the duration of the "environmentally predictable future" is typically 10,000 years, which is consistent also with the duration of the quantitative assessment period in SR-Can, although Swedish regulations also requires a more detailed assessment for the first 1,000 years following repository closure /SSI 2005/. In the very long term, after at least several hundred thousand years, no rigorous quantitative safety assessment is required and the judgement of safety can be based on more qualitative considerations.

Regarding the characteristic and performance of the engineered barrier system, YVL 8.4 requires that: "*The barriers shall effectively hinder the release of disposed radioactive substances into the host rock for several thousands of years*."

The importance to long-term safety of unlikely disruptive events shall, according to regulations, be assessed. According to STUK, these events are to include at least:

- Boring a deep water well at the disposal site.
- Core drilling intersecting a spent fuel canister.
- A substantial rock movement occurring in the environs of the repository.

The likelihood and consequence of the first two events is not considered to differ significantly between KBS-3V and KBS-3H repositories (although there will be some easily evaluated difference in the probability of a vertical borehole intersecting vertically, as compared to horizontally, emplaced canisters) and these are not discussed in the present report. The impact of substantial rock movement occurring in the environs of the repository is, however, discussed in the context of post-glacial earthquakes.

In addition to the criteria discussed above, there are also regulatory requirements on the protection of plants and animals. According to the YVL 8.4 guideline /STUK 2001/, "*exposures shall remain clearly below the levels which, on the basis of the best available scientific knowledge, would cause decline in biodiversity or other significant detriment to any living population*" and "*moreover, rare animals and plants as well as domestic animals shall not be exposed detrimentally as individuals*." The compliance with these criteria is not discussed in the present report but in the Biosphere Analysis Report /Broed et al. 2007/. According to an earlier study covering rather pessimistic cases /Smith and Robinson 2006/, the expected dose rates to any relevant biota are far below the levels of any reported effects.

Section 2.4 of Guide YVL 8.4 states that, whenever practicable, estimates of the probabilities of activity releases and radiation doses arising from unlikely disruptive events impairing long-term safety should be made. These probabilities should be multiplied by the calculated annual radiation dose or activity in order to evaluate the importance to safety of an event. The main differences between the Finnish and the Swedish regulatory systems are described in Appendix C.

1.6 Structure of this report

In accordance with Regulatory Guide YVL 8.4 /STUK 2001/, Posiva's Safety Case Plan 2005 /Vieno and Ikonen 2005/ and international guidelines on safety case contents (see Section 1.4), the elements discussed in this report include:

- General discussion of the hazards involved in spent fuel disposal (Chapter 2).
- The strength of the geological disposal concept and the support for the concept from natural analogues and from other international safety cases (Chapter 3).
- Understanding of the geological environment of the Olkiluoto site, based on investigations from the surface and on observations from the Onkalo monitoring programme and their implications, especially the changes in the salinity distribution of groundwater, and the long-term effects of stray materials introduced into the Onkalo and the repository (Chapter 4).
- Support for the robustness of approach and key assumptions of the KBS-3H safety assessment, based on a detailed comparison of assessment cases, models and databases with those used in other safety assessments (Chapter 5).
- Support for the robustness of KBS-3H safety assessment results by discussion of complementary safety indicators (Chapter 6).

To complete the report, a summary and conclusions are given in Chapter 7.

2 Hazard presented by spent fuel

2.1 Types of hazard

The types of hazards presented by spent fuel have been discussed in numerous documents published by the radioactive waste management community. A particularly useful discussion has been provided by /Hedin 1997/. Following Hedin's terminology, the risk associated with spent fuel is expressed as the product of probability of exposure and hazard. The probability of exposure is a measure of the degree to which a person is exposed to radiation in different situations, i.e. during transport, interim storage or deep disposal of spent fuel. The hazard of spent fuel is mainly related to its radiotoxicity, which is a function of the time-dependent activity of the spent fuel and the type of radioactive disintegrations involved. Spent fuel also presents a hazard due to its chemotoxicity, i.e. when exposed to the human body, spent fuel causes harmful effects by way of chemical reactions. The hazard related to the chemotoxicity of spent fuel is discussed briefly in Section 2.5, below.

The hazard (or radiotoxicity) presented by spent fuel is caused by external and internal radiation. When the source of radiation is outside the human body, the main hazard of spent fuel is related to external γ- (and neutron-) irradiation.

Other types of radiation (α- and β-radiation) are hazardous mainly if the radioactive substances enter the human body, primarily via ingestion of food, including water, and inhalation. In the safety assessment of the first several thousand years, where doses are evaluated, it is assumed that humans are exposed to radioactive substances released from the repository, transported to near-surface groundwater bodies and further to watercourses above ground. The following internal exposure pathways are thus considered /Posiva 2006/:

- Use of contaminated water as household water.
- Use of contaminated water for irrigation of plants and for watering animals.
- Use of contaminated watercourses and relictions.

In the period from several thousand to several hundred thousand years, safety criteria are expressed in terms of release rate constraints, and exposure pathways need not be considered.

After several hundred thousand years, no rigorous quantitative safety assessment is required but measures to quantify the hazard and its decrease with time can form the basis of lines of argument for safety that are not judged against formal numerical criteria.

2.2 Spent fuel characteristics

Several different types of spent fuel are planned for disposal in the Olkiluoto repository. These different types originate from the various Finnish nuclear power plants:

- BWR spent fuel from the boiling water reactors at Olkiluoto 1&2.
- PWR spent fuel from the pressurised water reactor at Loviisa, which has two sub-types TVEL VVER-440 and BNFL VVER-400 fuel assemblies.
- The advanced reactor under construction at Olkiluoto will give rise to EPR spent fuel.

The initial activity arising from these spent fuel types and its decay over time after unloading from the reactor (i.e. cooling time) are compared in Figure 2-1 for fuel with an initial enrichment of between 3.6% and 4% and a burn-up of 40 MWd/tU /Anttila 2005/.

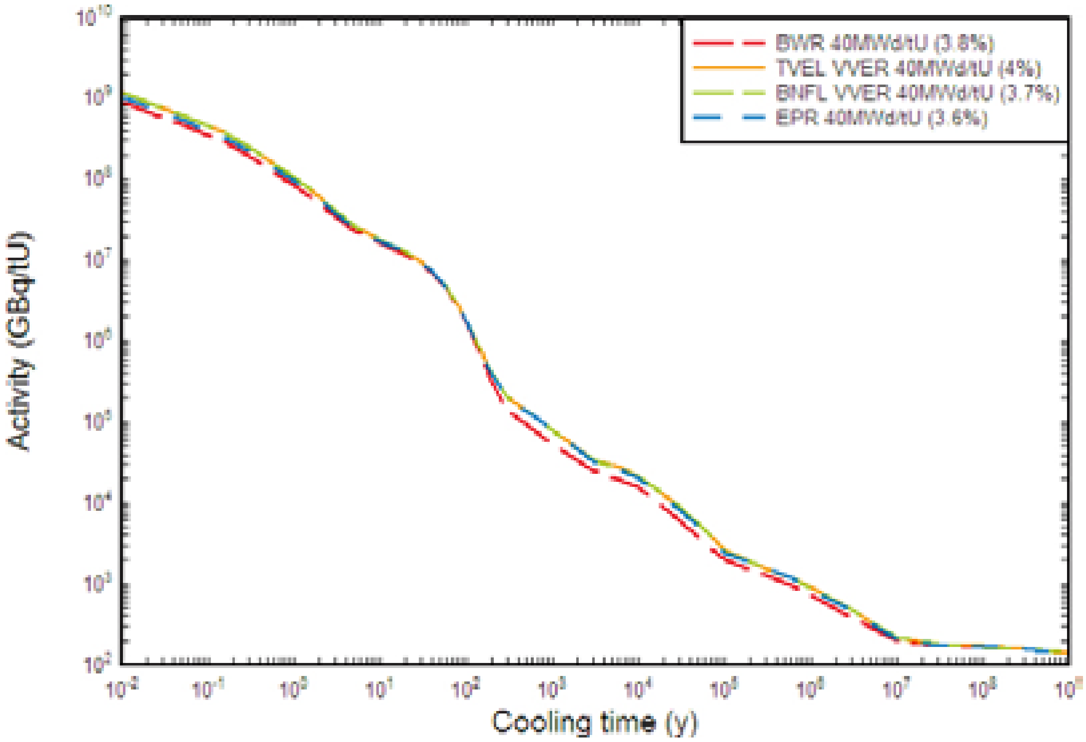

Figure 2-1. *Four different spent fuel types and their activity over time after unloading from the reactor. The burn-up was 40 MWd/tU for all types but enrichment varies slightly between 3.6 and 4.0% (data from /Anttila 2005/).*

The other major difference in spent fuel, especially that produced over a lengthy period when nuclear power plant operating conditions may change, is the burn-up. Generally, higher burn-up fuel has a higher activity on unloading from the reactor. Figure 2-2 shows the effect of increased burn-up on BWR spent fuel that had an initial enrichment of 3.8% /Anttila 2005/.

From these figures, it is clear that the range of spent fuel activity caused by fuel type, burn-up (and enrichment) is small compared to the change in activity even over the first year after unloading from the reactor.

BWR fuel with a burn-up of 40 MWd/tU is used as the reference spent fuel in the safety assessment calculations; Figures 2-1 and 2-2 suggest that the results for other fuel types are not expected to be very different.

2.3 Quantifying the hazard

Geological repositories provide safety by isolating the spent fuel from humans and their environment and containing the radionuclides associated with the spent fuel. The possibility of some eventual release of radioactivity cannot, however, be completely excluded. The harmful effects to the human body of any such releases are quantitatively expressed in terms of the annual effective dose to an adult individual. This is defined as the sum of the weighted dose equivalents in specific organs, integrated over 50 years, from the intake of activity into the body in one year, plus the sum of the weighted dose equivalents from external radiation in one year /Nagra 2002a/. For convenience, the term "annual individual dose", or simply "dose", is used in this report.

The radiological hazard associated with a given amount of radioactive material (whether contained in the repository or released to its surroundings) is sometimes expressed in terms of a "radiotoxicity index" or RTI(t) [-] /Nagra 2002a, Hedin 1997/, which is here defined as the hypothetical dose at time t resulting from ingestion of the activity A_j (t) [Bq] of radionuclide j, divided by 10^{-4} Sv (derived from the Finnish regulatory dose limit for the first several thousand years):

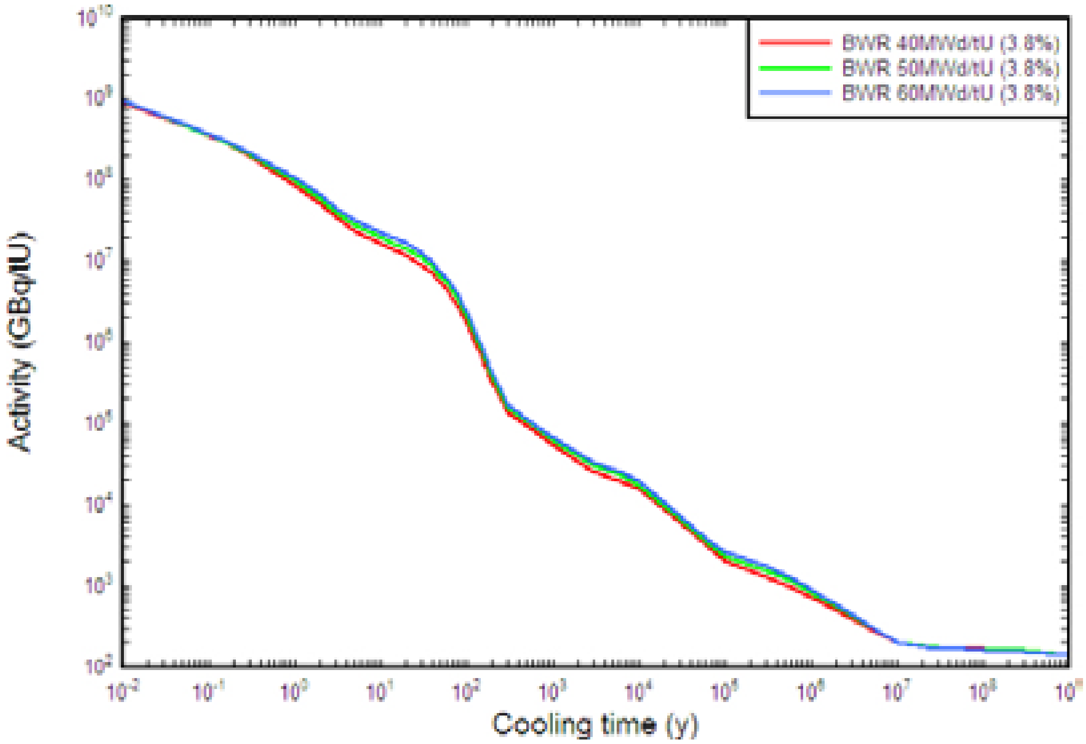

Figure 2-2. *The effect of varying burn-up on the activity of spent fuel. BWR fuel with an enrichment of 3.8% is used to illustrate the effect (data from /Anttila 2005/).*

$$RTI(t) = \frac{\sum_j A_j(t) D_j}{10^{-4}\,Sv} \qquad \text{(Eq. 2-1)}$$

where D_j [Sv/Bq] is the dose coefficient for ingestion.

The RTI allows a direct comparison of the radiological hazard through ingestion associated with radioactive waste, or radionuclides released from spent fuel, with that associated with different natural materials (Table 2-1).

An alternative indicator for hazard through ingestion is the "radiotoxicity flux" or RTF across a given interface (units: RTI/y, /Nagra 2002a/), which is defined by replacing the activity $A_j(t)$ in Eq. 2-1 by the annual activity flux $F_j(t)$ across that interface, see Eq. 2-2.

$$RTF(t) = \frac{\sum_j F_j(t) D_j}{10^{-4}\,Sv} \qquad \text{(Eq. 2-2)}$$

The RTF can be used for direct comparison of radiological hazard of activity fluxes from the repository with that from natural activity fluxes (Table 2-2). Appendix 3 in /Nagra 2002a/ provides a detailed evaluation of natural materials and fluxes that are relevant for a repository for spent fuel, high-level waste and intermediate-level waste in Switzerland. Similar calculations have been performed for the conditions relevant to a repository at Olkiluoto and the results are presented in Chapter 6 as part of the consideration of complementary safety indicators. These evaluations include trace elemental fluxes at the Olkiluoto site, prior to repository construction and operation, and groundwater fluxes, based on trace element data in /Pitkänen et al. 2003/.

Table 2-1. Natural materials which could be used to make comparisons of radiotoxicity with radioactive waste, or radionuclides released from spent fuel.

Radiotoxicity related to the repository	Natural material
Spent nuclear fuel	The uranium which was mined to produce the nuclear fuel
	Uranium ores of various grades, e.g. from Palmottu U ore deposit
	Olkiluoto host rock
Radionuclides that have migrated into the geosphere	Olkiluoto host rock
Radionuclides accumulated in the biosphere	Natural soil at Olkiluoto

Table 2-2. Possible comparisons of radionuclide fluxes originating from the repository with natural radionuclide fluxes (adapted from Table A3.10, /Nagra 2002a/).

Flux related to the repository	Natural flux
Flux of radionuclides from the repository into the geosphere	Flux of natural radionuclides over the repository area due to erosion
Flux of radionuclides from the geosphere into the biosphere	Flux of natural radionuclides dissolved in the groundwater within the biosphere aquifer
	Flux of natural radionuclides at the Olkiluoto site prior to repository construction
	Flux of natural radionuclides dissolved in water within the typical rivers
	Flux of natural radionuclides due to erosion

It should be noted that these measures of hazard relate only to ingestion (or, by a trivial modification, to inhalation). The repository also protects against the hazard associated with external irradiation, by shielding and by its deep underground location. External irradiation hazard is, however, relevant in the case of human intrusion scenarios (exposure to the intruder) and, in the very long term, if uplift and erosion have the potential to expose the spent fuel at the surface.

2.4 Evolution of radiological hazard over time

The radiotoxicity index (RTI, defined in Eq. 2-1) for 1 tonne and 5,500 tonnes (the repository inventory) of Finnish spent fuel to be disposed of in a KBS-3H (or KBS-3V) repository at Olkiluoto is shown in Figure 2-3 as a function of time after removal from the reactor (data from Appendix Table 2.1.1.1, /Anttila 2005/). This is compared with the natural radionuclides contained in 1 km^3 [(5)] of tonalite-granodiorite host rock at Olkiluoto[6] and with that of natural uranium ore corresponding to the volumes of the KBS-3H deposition drifts. The tonalitic-granodioritic-granitic-gneiss[6] is also used as an example to estimate the RTI of the volume of Olkiluoto host rock removed during excavation of the deposition drifts; this RTI is also shown in Figure 2-3.

5 1 km^3 is used as a reference volume since it is approximately the volume of rock between the repository at ~500 m depth and the surface, as the repository footprint is about 2 km^2 (the depth and area are 420 m and 1.6 km^2, respectively, in the current design; /Saanio et al. 2006/).

6 See Chapter 4 for a description of the rock types at the Olkiluoto site. The proportion of tonalitic-granodioritic-granitic gneisses (TGG) is about 8% in the drill cores studied so far at Olkiluoto /Andersson et al. 2007/, and thus the examples presented in this report are only indicative.

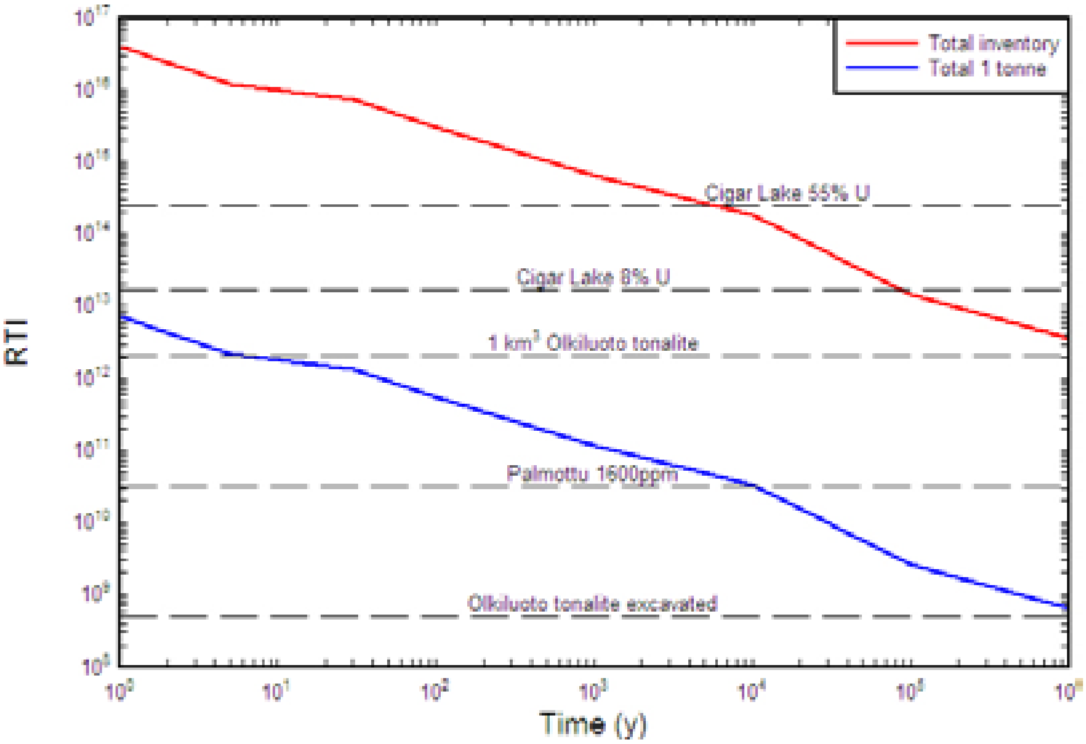

Figure 2-3. *Radiotoxicity index (RTI) of 1 tonne and 5,500 tonnes of Finnish spent fuel (red curve). Also shown are the RTI values for the volume of different grades of U which would fill the deposition drifts of the KBS-3H repository, as well as the RTI for the volume of Olkiluoto host rock that would be removed during excavation of the drifts (taking the Olkiluoto" tonalite-granodiorite" as representative of the TGG at the site, see Table B-6).*

For the natural uranium ore, different uranium concentrations (uranium ore grades) are considered: 8%, which is a representative concentration for the Cigar Lake uranium deposit in Canada /Cramer and Smellie 1994/; 55%, which is near the upper end of observed concentrations in uranium ore bodies; and 0.16% which is the average grade of the uranium mineralisation in the Finnish Palmottu U deposit /Ahonen et al. 2004/.

Figure 2-3 shows that the radiotoxicity of the spent fuel is substantially reduced over the course of time. After 1 million years, the radiotoxicity of the total inventory of spent fuel has dropped to below that of a volume of natural uranium ore (Cigar Lake 8% U) sufficient to fill the KBS-3H deposition drifts and is close to that of 1 km^3 Olkiluoto tonalite-granodiorite.

Another way of putting the potential hazard of spent fuel in perspective is to compare the activity contained in spent fuel with the activity in natural uranium that was used to produce the fuel /Hedin 1997/. Figure 2-4 shows the radiotoxicity index of 1 tonne of Finnish BWR spent fuel and the main contributing radionuclides as a function of time. This is compared to the RTI of 8 tons of natural uranium that was used to produce the fuel (from /Hedin 1997/). After about 100,000 years, the radiotoxicity of spent fuel has dropped to that of the natural uranium from which it was produced (assumed to be in equilibrium with its daughters).

Figure 2-5 shows the activity of Finnish BWR spent fuel with a burnup of 40 MWd/kgU as a function of time. The data in Figure 2-5 are normalised to the activity contained in an equivalent amount of mined uranium ore used to produce the spent fuel. As in the case of radiotoxicity, the activity of spent fuel is substantially reduced in the course of time. While fission and activation products dominate in the first 100 years, actinides and their daughter radionuclides dominate after roughly 500 years.

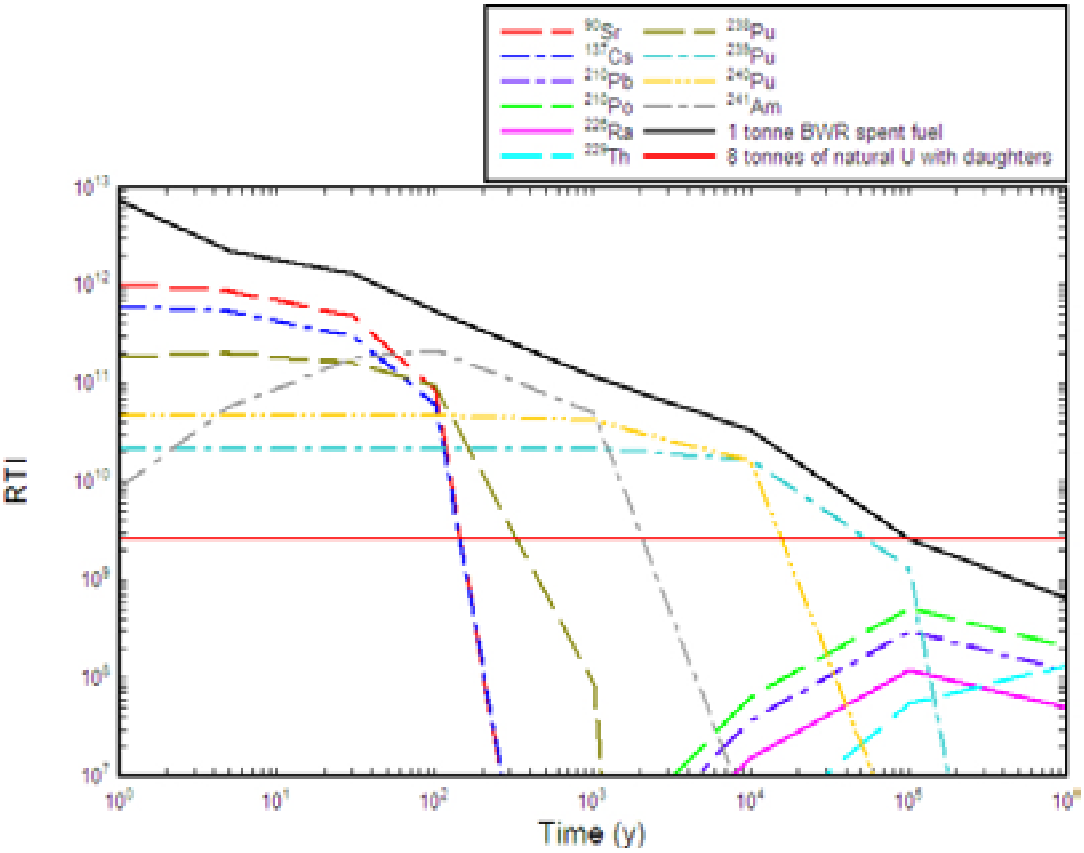

Figure 2-4. *Radiotoxicity index (RTI) of 1 tonne of Finnish BWR-type spent fuel with a burnup of 40 MWd / kgU and the major nuclides contributing to the RTI. Also shown is the RTI of 8 tonnes of natural uranium from which the precursor fuel was derived.*

After about 200,000 years, the activity of the spent fuel has declined to a level comparable with the natural uranium ore that was originally mined to produce the fuel.

Of all the radionuclides initially contained in spent fuel, basically what will remain in the far future are the long-lived uranium isotopes U-238 and U-235, with half-lives of 4.5×10^9 years and 7×10^8 years, respectively, together with their daughter radionuclides that are continuously formed by radioactive decay. Thus, the nuclear fuel cycle gives rise to a significantly increased activity during a period of roughly 1 million years. Thereafter, the total activity of the fuel cycle is roughly equal to the natural uranium minerals that were used to produce an equivalent amount of fuel.

As noted in /NEA 2007a/, such comparisons need to be used with caution. This is not only because the isotopic compositions of natural systems will differ from those of both the initial spent fuel and eventual repository releases but also because the assumption should not be made that natural situations are necessarily harmless. Furthermore, the comparisons do not address the hazard associated with external radiation (although, as noted previously, this hazard is very considerably removed by the location of the spent fuel within a deep geological repository), nor the different "accessibility" of the radiotoxic material when it is in its natural form compared to the spent nuclear fuel surrounded by the man-made barriers. Figures 2-3, 2-4 and 2-5 nevertheless indicate that the timescale over which spent fuel presents a hazard deserving special attention is in the order of one million years. This, however, does not imply a requirement for complete containment of radionuclides by all copper canisters in the repository for this period of time. The surrounding barriers (buffer and geosphere), being part of the isolation system, are expected to mitigate any potential radiological effects of an earlier than expected canister failure. Nor does it imply that the time period in excess of one million years can be ignored in the safety case. As also

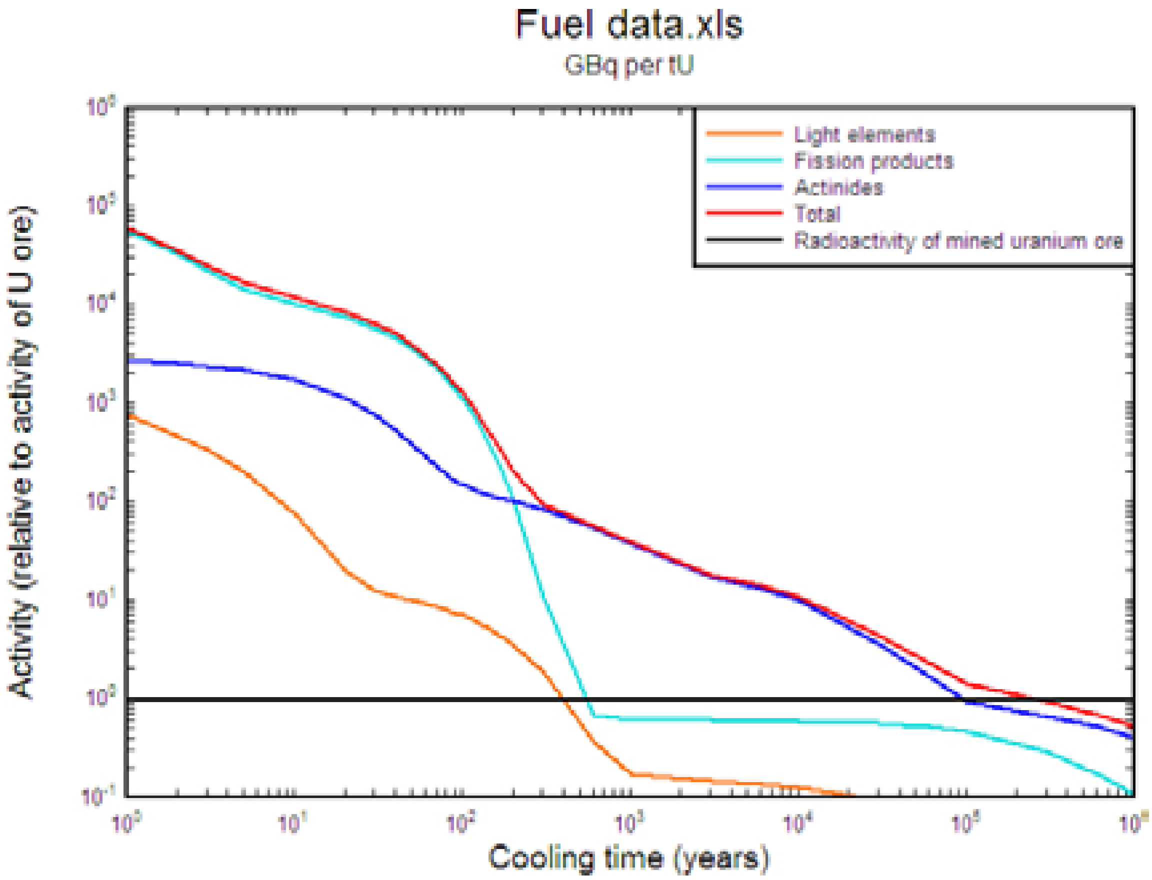

Figure 2-5. *Relative activity of uranium ore and spent nuclear fuel as a function of time. The total activity of one tonne of Finnish BWR-type spent fuel with a burnup of 40 MWd / kgU is shown (relative to the natural U ore activity of approximately 1.4TBq). The spent fuel activity is also shown separated by type of radionuclide. The light elements and fission products decay relatively quickly so that the main activity of the spent fuel after 1,000 years is due to the actinides and their daughters.*

noted in /NEA 2007a/, in addition to U-238 with a half-life of 4.5×10^9 years, other radionuclides (especially some of those created artificially in nuclear reactors) that could be important in terms of the hazard from external radiation persist out to one million years or longer. Thus, even though the hazard potential of spent fuel decreases markedly over time, spent fuel can never be said to be intrinsically harmless.

2.5 Chemotoxicity of releases from a spent fuel repository

The preceding sections considered the radiological hazard associated with the spent fuel in the KBS-3H repository but there are also risks associated with chemical toxicity of the variety of metals such as copper, antimony, cadmium, lead, uranium and plutonium and with other problematic elements like selenium and arsenic which are present in the repository and may be released into the groundwater.

Assessing the chemotoxicity of spent nuclear fuel presents significant challenges /Chapman and McCombie 2003/:

- Relatively limited data on the toxic effects of chemicals can be derived from toxicology experiments, epidemiological studies on exposed populations (e.g. occupationally exposed workers) or physiologically-based pharmacokinetic models.

- The largest body of data is based on experiments with animals and its applicability to humans is not straightforward.
- The diversity of toxins is much greater than that of radiation.
- As is the problem with radiation, chemical toxicity at low doses is difficult to assess.
- The cumulative effects of the presence of several chemical carcinogens in the same mixture and the interaction of chemotoxicity and radiotoxicity are unknown.

Extensive databases on the toxicity of different chemicals exist, an example is the Integrated Risk Information System (IRIS) database maintained by the U.S. Environmental Protection Agency (http://www.epa.gov/iris). This database provides reference oral doses and inhalation concentrations for chemicals; below these doses, no adverse effects are expected to occur. It also gives carcinogenic assessment for chemical agents. The World Health Organisation (WHO) also provides drinking water standards that are currently used to complement country-specific groundwater quality standards.

A few radioactive waste management programmes have attempted to evaluate the chemical toxicity of selected elements or carried out a preliminary assessment related to a deep geological repository for low, intermediate or high-level waste, including spent fuel (Canada, Finland, Sweden, UK, France and USA[7]). In this section, we describe the results of the most relevant study carried out by /Raiko and Nordman 1999/ for Posiva.

/Raiko and Nordman 1999/ applied the same type of approach used in the TILA-99 safety assessment: define the source term, define the possible transfer pathways and end points and compare with available drinking water standards to evaluate the overall safety. The models, data and results are the same as in TILA-99 and they apply to a KBS-3V repository. The elements expected to be of interest were identified based on the spent fuel composition. However, the repository also contains large amounts of Fe and Cu in the canisters which are not usually considered as part of the waste, per se. The annual releases from the repository were assumed to be diluted in 100,000 m^3 of well water, as in the calculation of radiological dose. This means that some elements can be discounted on the basis that, with this dilution factor, if there is less than 100 g of the element in the spent fuel canister, the concentration in the water will not exceed the lowest limit for any element that of mercury, set at 0.001 mg/L (assuming the failure of a single canister). The results are compared with drinking water standards set by WHO on health criteria or technical-aesthetic grounds[8] /WHO 1993, 1998/. The concentration limit for those elements for which there is no standard is assumed to be the same as that of mercury (0.001 mg/L). It was noted that the WHO standard for uranium in groundwater is 0.002 mg/L based on chemotoxicity, which is very low compared to the limit based on radiotoxicity of 14 mg/L, corresponding to an annual dose of 0.1 mSv (assuming 500 litres water consumption). Other transuranic elements were not treated because their chemotoxicity is considered to be so much lower than their radiotoxicity.

The calculation of maximum releases of elements was carried out with the code REPCOM and FTRANS (the same codes used in the current safety assessment for the near field and the far field, respectively. See Section 5.4). Material inventories are based on Olkiluoto 1&2 spent fuel with 3.3% enrichment and burnup of 36 MWd/kgU and Loviisa fuel with 3.6% enrichment and

7 Canada's OPG /Garisto et al. 2005/ presented an illustrative case of a chemotoxicity safety assessment for CANDU fuel; Posiva's study /Raiko and Nordman 1999/ is discussed further in the text; in Sweden, SKB only mentioned chemotoxicity in the FEP list of the interim main report of SR-Can /SKB 2004, Andra 2005b and AEA Technology 2001/ presented a preliminary assessment for the French and the UK low and intermediate level waste repositories, respectively; USDOE assessed the chemotoxicity of selected elements of concern outside the waste package at Yucca Mountain: Cr, Cu, Mn, Mo, Ni and V /USDOE 2002/.

8 Technical-aesthetical properties of water, such as colour, odour, turbidity, may not have a health impact but have a negative impact on the overall water quality.

the same burnup. Geosphere parameters[9] used are comparable to or slightly more pessimistic than those used in most of the cases of the KBS-3H radionuclide transport report /Smith et al. 2007b/. Maximum releases of elements in terms of concentration and the corresponding limits for drinking water results are shown in Table 2-3. Releases were converted to doses using the WELL-96 conversion factors (which very are similar to the WELL-2007 conversion factors used in the KBS-3H radionuclide transport calculations, see Section 5.5).

Although the results are specific to TILA-99, the values in Table 2-3 indicate that chemotoxicity is unlikely to be a more significant issue for the KBS-3H repository at Olkiluoto, for which calculated radionuclide releases are of similar magnitude to those in TILA-99 (see Section 5.6 for comparisons between TILA-99 and the KBS-3H safety assessment), since concentrations of spent fuel elements are at least four to six orders of magnitude lower than the limits.

Further evaluation of the chemotoxicity of spent fuel is beyond the scope of the KBS-3H safety studies as defined in Section 1.3.1. However, this is an issue of common interest for KBS-3V and KBS-3H.

Table 2-3. Maximum concentrations of elements released from a single TILA-99 spent fuel canister and their permissible concentrations in drinking water (modified from /Raiko and Nordman 1999/). The limit for drinking water for those elements that do not have a concentration limit is assumed to be equal to that of mercury (0.001 mg/L or 5 x 10^{-9} mol/L).

Element	Concentration in well water (mol/L)	Maximum permissible concentration in drinking water in Finland (mol/L)
Cs	$1.01 \cdot 10^{-12}$	
I	$9.33 \cdot 10^{-12}$	
Co	$2.27 \cdot 10^{-12}$	
Cr	$1.00 \cdot 10^{-11}$	$9.6 \cdot 10^{-7}$
Cu	$1.00 \cdot 10^{-10}$	$1.6 \cdot 10^{-5}$ (1) $3.2 \cdot 10^{-5}$ (2)
Pb	$1.00 \cdot 10^{-14}$	$4.8 \cdot 10^{-8}$
Mn	$1.00 \cdot 10^{-11}$	$9.1 \cdot 10^{-7}$ (1) $9.1 \cdot 10^{-6}$ (2)
Mo	$6.67 \cdot 10^{-11}$	$7.3 \cdot 10^{-7}$
Ni	$1.00 \cdot 10^{-11}$	$3.4 \cdot 10^{-7}$
Nb	$1.56 \cdot 10^{-11}$	
Pd	$1.00 \cdot 10^{-15}$	
Fe	$1.00 \cdot 10^{-11}$	$3.6 \cdot 10^{-6}$ (1)
Rb	$4.61 \cdot 10^{-12}$	
Sr	$1.10 \cdot 10^{-11}$	
Tc	$5.00 \cdot 10^{-15}$	
Te	$1.00 \cdot 10^{-13}$	
Sn	$1.00 \cdot 10^{-13}$	
Ti	$1.00 \cdot 10^{-15}$	
Th	$1.76 \cdot 10^{-14}$	
U	$3.00 \cdot 10^{-14}$	$8.4 \cdot 10^{-9}$ (2)
Bi	$1.00 \cdot 10^{-14}$	

[1] Based on technical-aesthetic grounds rather than health grounds.

[2] WHO recommendations based on health grounds.

9 Geosphere data used: WL/Q= 10,000 y m^{-1} corresponding to a fracture aperture of 0.5 mm, fracture width of 1 m, fracture length of 600 m and effective flow rate Q_f= 50 L y^{-1}. See sections 5.4 and 5.5 for further explanation.

3 The concept of geological disposal of spent fuel

3.1 The strength of geological disposal as a waste management option

The disposal of spent fuel needs to be managed in such a way as to ensure the safety of humans and to protect the environment for very long periods of time. According to current understanding, deep geological disposal is the only waste management option that offers long-term passive safety /NRC 1957, 2003, NEA 1999ab/.

In the Government Decision 1999/478, Finland selected geological disposal as the preferred long-term spent fuel management option. In the present report, no consideration is thus given either to aspects related to the (previously demonstrated) feasibility of geological disposal in Finland or to alternative long-term waste management options.

The strength of geological disposal is supported by:

- The existence of suitable rock formations in Finland – in 2001, the Finnish Parliament ratified the Government's favourable Decision in Principle (DiP) on Posiva's application to locate the repository at Olkiluoto. The suitability of the granitic host rock in Finland has been investigated by nearly two decades of site characterisation efforts, by the calculations in the TILA-99 safety assessment, which compared the radiological safety of four potential sites in Finland (including Olkiluoto) and by a demonstration of compliance with regulatory requirements. The existence of suitable rock formations in Finland has thus been demonstrated /e.g. McEwen and Äikäs 2000/ and the issue is not further discussed in this report.
- The strength of available repository concepts – in Finland and elsewhere, repository concepts have been developed and demonstrated to provide the required stability and longevity in different geological environments and for a range of geochemical conditions (Section 3.2).
- Observations of natural systems – indirect evidence for safe geological disposal is also provided by observations from natural systems, including the longevity of uranium ore deposits in many different geological environments (Section 3.3).
- Safety assessments conducted world-wide – the findings of integrated safety assessments conducted by numerous disposal organisations world-wide for a wide range of sites, host rocks and repository designs support the possibility of safe geological disposal (Section 3.4).

3.2 The strength of the process for repository design

Repository design is based on a set of principles which constitute a design philosophy. Table 3-1 gives the design principles for disposal and their implementation in the KBS-3H design as set forth by the Finnish regulator STUK in YVL 8.4 /STUK 2001/.

From these design principles, two points are of particular importance with respect to the strength of the repository design process:

- The stepwise approach to design and implementation to make use of better information as it becomes available.
- Design for robustness and safety to minimise uncertainties or detrimental processes.

Table 3-1. Principles for disposal and implementation in KBS-3H. Structure according to design principles set forth in YVL 8.4, Section 3.

Design principles for disposal	Implementation in KBS-3H
	Method of implementation and timing of disposal
Primacy of safety	The disposal concept will be implemented with due regard to safety. Safety is considered to have priority over other aspects related to implementation (stakeholder interests, timing, costs).
Use of best available technology and scientific knowledge	Utilisation of best available technology or techniques with good prospects to become available in a reasonable time frame.
	Acquisition of sound scientific knowledge (experimental investigations, expert elicitation, modelling capabilities, evidence from natural analogues, etc).
Optimising the timing of disposal	Credit will be taken for the decrease of the activity and radiogenic heat load of spent fuel during interim storage.
	Unnecessary delay in repository implementation will be avoided, to limit the hazard and other burdens to future generations.
Step-wise implementation	At each stage of repository implementation, the information basis will be sufficient to reliably characterise the system components at the required level of detail.
	Stepwise implementation allows for the involvement of stakeholders, the opportunity of feedback and the possibility of including modifications to the design at various stages of repository implementation.
	A possibility for monitoring and retrievability is ensured for a limited time period (see below).
No requirement for monitoring	Disposal will be planned so that no monitoring of the repository is required for ensuring long-term safety. The design should, however, allow for monitoring if desired.
Possibility of retrievability	Disposal will be planned so that retrievability of the spent fuel canisters is possible for a limited time period after spent fuel emplacement.
	Barrier system
Redundant, passive barriers	Multiple, passive barriers ensure long-term safety: Copper-iron canister, buffer around canister, distance block, drift end plug, granitic host rock. The technical barriers are designed in such a way that a deficiency in one barrier or a predictable geological change does not jeopardise long-term safety. No active maintenance of barriers will be required after repository closure. The technical barriers will effectively hinder the release of radionuclides during the first several thousand years, should any radionuclides be released in this time period, and at later times.
Design for robustness and safety	Multiple phenomena will contribute to the safety functions [(1)]. Uncertainties and detrimental phenomena will either be avoided, or their effects will be mitigated by suitable design measures.
	Robustness is enhanced by minimising potentially detrimental interactions between barriers and components.
	Disposal site and repository
Favourable repository site features	The long-term stability of the disposal site is favoured by the stable geological setting in Finland. Major fracture zones will be avoided in the layout of the deposition drifts. Safety should not depend only on the features of the site but the EBS should work within the site to ensure the safety of the repository.
Suitable repository depth	The impacts of events, human actions and environmental changes at the surface on long-term safety will be mitigated by choosing a suitable repository depth. Also, the disposal at depth will render inadvertent human intrusion into the repository difficult.

[1] For a discussion of safety functions, see the KBS-3H Evolution Report /Smith et al. 2007a/.

3.2.1 The stepwise approach to design and implementation

Designing a repository for radioactive waste is not a simple process akin to designing a structure where the key requirements, design factors and the site are known at the outset. For a repository, the overall requirements in terms of regulatory targets for operational and long-term safety, project duration and key milestones, budgets etc are known. However, the way in which the repository design process will achieve these requirements, beginning with incomplete site information, is a developmental process which makes use of information as it becomes available to minimise detrimental features or processes, or to take advantage of favourable ones. This requires a flexible repository design which can be developed iteratively throughout the repository implementation programme, allowing feedback from site characterisation and safety assessment – the stepwise approach. This approach is the basis of both Posiva and SKB design philosophies and its implementation is illustrated in the continuous updating of the Olkiluoto repository layout /e.g. Saanio et al. 2006/ to reflect a better site understanding /Posiva 2005/. In particular, leaving open key decisions, such as repository depth (or indeed, the choice of KBS-3H or -3V), until such time as there is sufficient information to make a fully informed decision which can be justified to regulatory authorities and other stakeholders, is part of the process. Assuming that the plans for the construction and operation of the repository for KBS-3V apply also to the KBS-3H design, the repository will be constructed and operated in nine phases /Saanio et al. 2006/. The duration of the operational period would be approximately 100 years.

Some repository design features can be used to address uncertainties about the geological environment. For example, the properties and long-term behaviour of fracture zones may be uncertain but not all such features can be avoided in siting the repository. The fracture zones occurring at the repository site can be classified into:

- Those which are sufficiently major to influence layout, i.e. the fractures that are avoided by adjusting the repository layout so that disposal panels lie wholly within host rock volumes bounded by fracture zones. Fracture zones of this magnitude would normally be detected by site investigations before construction of the repository begins.
- Smaller, more frequent and unavoidable features which influence spent fuel emplacement, i.e. drift sections intercepted by these features would not be used for emplacement of spent fuel packages, which would be replaced in KBS-3H by additional filling blocks or isolated by steel compartment seals, depending on the hydraulic properties of the fracture zone. The properties of these zones may not be apparent until excavation of the drifts, thus decisions about the necessity to avoid them, and the method for dealing with their presence in the deposition drifts, may be taken quite late in the planning.

For example, in the second stage of the planning for a KBS-3V repository at Olkiluoto /Saanio et al. 2006/, fracture zones in the bedrock at repository level (–420 m) with an average transmissivity of $>10^{-5}$ m^2/s should not be penetrated by any tunnels; these are equivalent to the layout determining features. According to the report smaller fracture zones and regions of rock with transmissivity of 10^{-5} to 10^{-7} m^2/s can be penetrated by deposition tunnels but no canisters will be located at those positions. Aso according to the same report structures with transmissivity below 10^{-7} m^2/s will not influence deposition hole positions. However, these criteria will be updated and specified by the Rock Suitability Criteria (RSC) programme set up by Posiva to develop a classification scheme to be applied for KBS-3V type repository layout, defining suitable rock volumes for repository panels, assessing whether disposal tunnels or sections of them are suitable for deposition holes and deciding on whether a deposition hole is acceptable for disposal.

In KBS-3H safety studies, it is assumed that drift sections having an inflow higher than 0.1 litres per minute (summed over discrete inflow points within an approximately 10 m-long drift section) will be excluded as distance block or supercontainer emplacement locations. The 0.1 litre per minute inflow criterion is roughly equivalent to a maximum fracture transmissivity

$T = 3 \times 10^{-9}\ m^2\ s^{-1}$ assumption if most of the inflow to a drift section is conveyed by a single fracture (see Section 7.1.4 and Appendix B.2 of /Smith et al. 2007a/). Filling blocks will be installed in zones with an upper inflow limit of 1 litre per minute after grouting (as the inflow limit for the installation of filling blocks is based on the requirement to avoid erosion of the blocks during the period of saturation (Section 5.5.6 and Appendix L of /Autio et al. 2007/). Inflow rates above this limit will require the installation of a compartment plug (see Figure 1-3).

The maximum allowable inflow of 1 litre per minute is thus higher in the case of filling blocks compared with the 0.1 litres per minute allowed for distance blocks. This is because of the different functions of these two components. The distance blocks should prevent significant water flow by piping between adjacent supercontainer drift sections during saturation of the drift, which could otherwise lead to buffer erosion, as described in Section 4.5.2 of the Process Report /Gribi et al. 2007/. The limit of 0.1 litres per minute is related to this requirement. The filling blocks, on the other hand, are not used to separate adjacent supercontainers and so the prevention of piping is not a primary consideration in deciding where they can be emplaced. There is, however, a requirement to avoid erosion of these blocks by water flowing around the drift through intersecting transmissive fractures and erosion. The relevant inflow criterion is expected to be higher, although the present choice of 1 liter per minute is preliminary and somewhat arbitrary value that may be updated in view of future studies and possible design changes.

According to the discrete fracture network modelling carried out by /Lanyon and Marschall 2006/ and to hydrogeological considerations at the Olkiluoto site, a 300 m-long KBS-3H drift will contain, on average, 17 to 18 supercontainers for the reference fuel type (BWR from Olkiluoto 1&2) or 22 to 23 supercontainers for all Finnish fuel types, one drift section (30 m long) with unfavourable hydraulic features, which is isolated from the rest of the drift by compartment plugs, and 3–4 filling blocks (each 10 m). On average, 17% of the drift will be unusable due to water inflow exceeding the 0.1 L/m criterion. A higher figure of 25% is tentatively and conservatively assumed in the layout adaptation /Johansson et al. 2007/, see also Section 2.2.7 of the KBS-3H Evolution Report; /Smith et al. 2007a/. This figure is based on a drift separation of 25 m and consideration of all the different Finnish fuel types (canister pitch ranging from 9.1 to 11 m). It also takes into account the possibility that some relatively tight fractures, which have the potential to undergo shear movements sufficiently large to damage the canisters, will be identified and avoided.

As part of this on-going process of fitting the repository into the site, a Host Rock Classification (HRC) system has been developed for the Olkiluoto site for the purpose of identifying suitable volumes of rock for the disposal of spent nuclear fuel /Hagros 2006/. The HRC-system is, however, specific to KBS-3V and is not directly applicable to KBS-3H, although it may well be that the host rock defined as suitable for a KBS-3V repository is equally suitable for a KBS-3H repository. The HRC-system has been applied in the KBS-3H layout study in order to define the respect distances to lay-out determining fracture zones, with some minimal modifications. A continuation of the HRC is the Rock Suitability Criteria (RSC) programme which has been set up to develop a classification scheme to be applied for KBS-3V type repository layout, defining suitable rock volumes for repository panels, assessing whether disposal tunnels or sections of them are suitable for deposition holes and deciding on whether a deposition hole is acceptable for disposal.

This stepwise process illustrates how layout adaptation and emplacement flexibility allow less favourable aspects of the host rock and geological environment to be avoided. Other features of the design concept seek to minimise disturbance to the host rock:

- The plans to excavate deposition drifts simultaneously with emplacement operations in other areas of the repository means that deposition drifts are open for the minimum amount of time, reducing potential for hydrogeological, mechanical and geochemical perturbations around the tunnel.

- Excavation techniques can be adapted to minimise the extent of the excavation disturbed/damaged zone (EDZ), especially around the deposition drifts. This may mean that different excavation techniques are used depending on the type of construction (access tunnels or deposition drifts) and sensitivity to development of an EDZ. The extended period of excavation of the repository may also mean that advances in excavation technology will be available for later stages and these will be taken up where advantageous as part of the commitment by Posiva to use of the best available technology, as required in the YVL 8.4 regulations /STUK 2001/ (see below).

STUK Regulation YVL 8.4 (as well as Swedish regulations) mentions under "Design principles for disposal" the use of "best available technology" (BAT): "*In accordance with Section 7 of the Government Decision, the implementation of disposal, as a whole, shall be planned with due regard to safety. The planning shall take account of the decrease of the activity of spent fuel by interim storage and the utilisation of best available technology and scientific knowledge. However, the implementation of disposal shall not be unnecessarily delayed*."

The meaning of BAT is subject to considerable subjectivity. The meaning of "best" is also subjective, considering the complex technical, economical, societal factors to be taken into account in a repository programme. Furthermore, the difference between BAT and "optimisation" and the implications of BAT in a repository programme are still unclear. Verification of compliance with BAT is also difficult from a regulatory perspective.

The issue of BAT is therefore under discussion in Finland and in Sweden as well as in other countries in which BAT is, or could become, a regulatory requirement. In general, the use of BAT cannot be claimed until the construction license application and most likely even later, when significant operational experience will have been gathered. Section 13.3.4 of the SR-Can Main report /SKB 2006a/ presents a preliminary assessment of BAT concerning the material and dimensions of the canister, buffer and backfill as well as the repository layout. The same section also includes a discussion of BAT and optimisation.

In summary, the strength of the stepwise approach is that it allows for changes as new knowledge is gathered, even during the construction/operations phase, but it must be noted that proper regulatory oversight is involved to ensure that the basis for any changes is in agreement with the design principles for disposal set forth by the regulatory authorities and also any license agreement under which the work is authorised.

3.2.2 Design for robustness and safety

A disposal concept based on a system of passive barriers which provide multiple safety functions through processes and properties that are well understood and that is not unduly affected by residual uncertainties can be said to be robust.

Since the early days of repository disposal concepts for long-lived, high-level radioactive waste, there has been considerable consensus on the materials used for the engineered barriers: materials considered for the overpack or canister have been steel (usually carbon steel) and copper, with titanium as a possible third option, and the buffer material based on a natural swelling clay, usually montmorillonite in the form of bentonite /e.g. KBS 1978, KBS 1983, Nagra 1985/. This was based on the recognition that demonstration of stability of the engineered barriers for very long periods of time would be required to assure long-term safety, thus the use of few materials with well known and understood behaviour, such as copper and steel, would simplify meeting this requirement.

In addition, the use of naturally-occurring materials, such as bentonite, meant that natural analogues could be used to demonstrate stability over geological timescales under conditions relevant to deep geological disposal. The disposal concepts described in Table 3-2 from a number of national radioactive waste programmes, illustrate how, despite differences in repository concept and site characteristics, the essential EBS materials remain the same. Although repository concepts have developed over the years, the materials used for the EBS (e.g. copper, steel, bentonite) have been the same from the outset of the various national programmes, for example the Canadian Environmental Impact Assessment of 1994 /Goodwin et al. 1994, AECL 1994/ and the Second Case study /Wikjord et al. 1996/, the Japanese H3 concept /PNC 1992/ and the Swiss Project Gewähr /Nagra 1985/. Other repository design concepts for different types of host rock (i.e. clay formations, e.g. Opalinus Clay in Switzerland /Nagra 2002a/, Boom Clay in Belgium /Ondraf/Niras 2001/ and Callovo-Oxfordian mudstones in France /Andra 2005b/, and tuff, e.g. Yucca Mountain Project /USDOE 1999/) are not discussed here except to note that these also use the same tried and tested materials.

On the other hand, materials with potentially detrimental interactions with other barriers or properties that are difficult to predict over the lifetime of the repository have been avoided as far as possible, although complete avoidance is not always possible; the potentially detrimental interactions between bentonite and corrosion products from the steel supercontainer in KBS-3H being a case in point /see Smith et al. 2007a, Gribi et al. 2007/. In this case, and also when considering the use of highly alkaline, OPC-based[10] materials, it has to be shown that the certain benefits of the use of a component or material (e.g. for operational safety) outweigh the possible detriment /Vieno et al. 2003/. For some applications, alternatives to OPC have been developed based on low pH cements and Silica Sol (colloidal silica) for grouts /e.g. Ahokas et al. 2006, Bodén and Sievänen 2005/ since practical and safe construction of a repository will require use of such materials for groundwater control and engineered structures[11].

The avoidance of materials with potentially detrimental behaviour is extended even to use of materials in the repository which are not part of the engineered barriers /Posiva 2003c, Juhola 2005/. Monitoring is carried out of all materials involved in the construction and operation of the Onkalo underground rock laboratory /Juhola 2005, Vuorio 2006/ and this will be extended to the repository itself. The purpose of this monitoring is to ensure that the number and amounts of foreign materials introduced is as low as possible to minimise:

- The perturbation of the repository near-field conditions – the short-term conditions, which could affect site characterisation, as well as the long-term, safety-relevant conditions.
- The potential for chemical interactions between these materials and the EBS or host rock, which could affect their long-term behaviour.
- Introduction of materials which could enhance radionuclide transport, e.g. organic materials which could give rise to complexants[12] with high environmental mobility.

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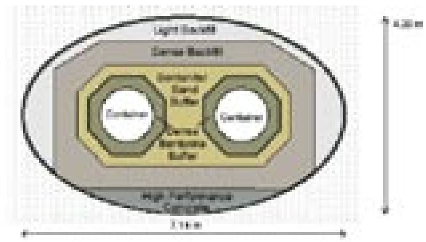
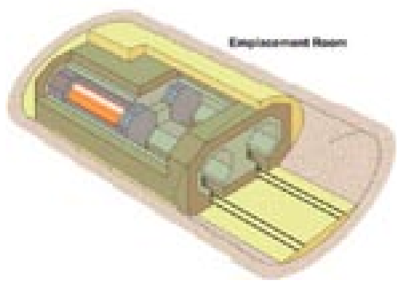
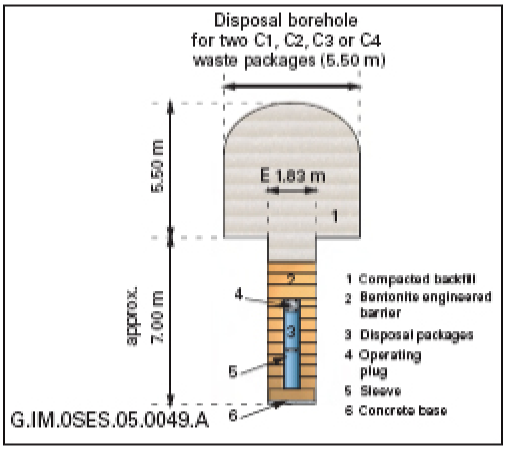
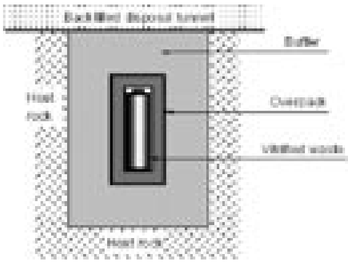
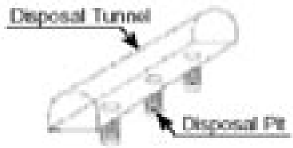
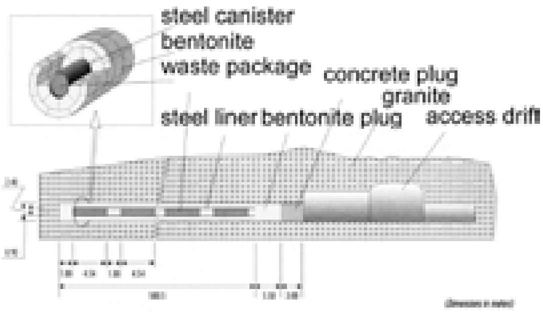
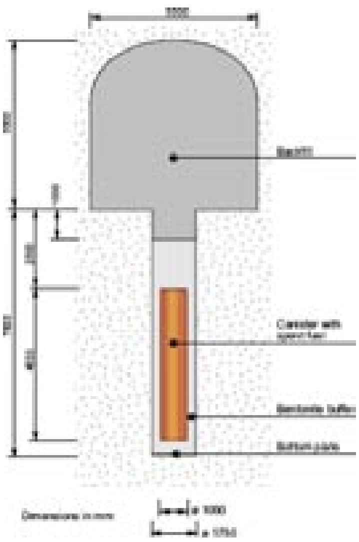
[10] Ordinary Portland Cement (OPC) is the basis of much concrete and mortar used in engineering applications.

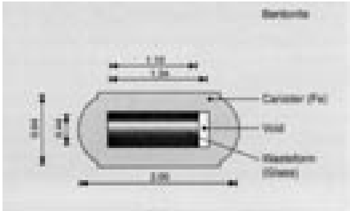
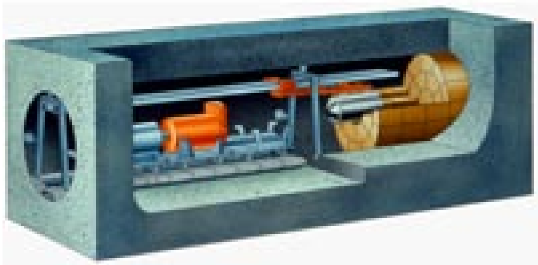
[11] In Appendix F of the KBS-3H Process report /Gribi et al. 2007/ it is estimated, based on current excavation/construction experience at Onkalo that the average amount of low pH cement per drift will be of the order of 2.6 to 3.9 tonnes, compared to more than 1,000 tonnes of bentonite. Mass balance calculations indicate that no significant consequences are expected from the use of these cement-based materials in the drift.

[12] These organic substances and their association with some metals could be particularly relevant when assessing the chemical risks associated with the repository system. The impact of these substances on the radiological of the repository remains an issue for future safety assessments.

Table 3-2. Repository design concepts for granitic host rocks in other national programmes. (SF: spent fuel; HLW: vitrified high-level waste; WP: waste package).

Programme	Waste	Overpack	Geometry/layout	Buffer
Canada OPG TCS /Gierszewski et al. 2004/	SF	Cu shell, steel inner support vessel	Horizontal emplacement of pairs of waste packages in oval-section[1] emplacement rooms.   	Buffer: 100% bentonite Backfill: 50:50 Dense: blocks Light: pellets 0.5 to 1.1 m radial thickness
France Dossier 2005 Granite /Andra 2005a/	HLW (SF)	Steel (Cu for SF)	2 or more (dependent on thermal characteristics) HLW canisters ("C wastes") emplaced in vertical boreholes. Steel sleeve inside the bentonite buffer sections allows for WP retrieval for a period.  KBS-3V design is envisaged if SF disposal is required.	Bentonite 100% 0.6 m radial thickness

Japan[2] H12 /JAEA 2000/	HLW	Carbon-steel	Vertical (pit) emplacement of single waste packages or horizontal (tunnel) emplacements of multiple WPs. EBS dimensions per WP are the same in both cases.   	Bentonite/ sand 70:30 mixture 0.7 m radial thickness
Spain /Enresa 1998/	SF	Carbon steel	Waste packages emplaced horizontally along axis of tunnel within steel liner surrounded by buffer. 	Bentonite 100% 0.75 m radial thick-ness
Sweden SR-Can /SKB 2006a/ (Also Finland /Vieno and Nordman 1999/)	SF	Cu with cast iron insert	Single waste package emplaced vertically in a borehole (KBS-3V design). 	Bentonite 100% 0.35 m radial thick-ness

Switzerland Kristallin-I /Nagra 1994/	HLW + SF	Carbon steel	Waste packages emplaced horizontally along axis of tunnel surrounded by buffer.  	Bentonite 100% 0.7 m radial thickness

[1] Strongly non-circular cross section is required in tunnels in response to marked stress anisotropy.

[2] The generic studies H3 and H12 considered crystalline "hard rock" which could be other than granitic, e.g. gabbro, basalt. However, groundwater flow in a fracture network was assumed.

3.3 Support for the concept from natural and anthropogenic analogues

Observations from natural systems or anthropogenic artefacts can provide supporting evidence for the safety of geological disposal concepts with respect to the long-term performance of the components of the repository. Although neither natural nor anthropogenic analogues represent any planned geological repository in its entirety, analogues can be found which represent many of the materials and processes of relevance to repositories.

Natural analogues may be used at a qualitative level, for example, to demonstrate some aspects of how a repository system is expected to evolve or perform over long periods of time, or at a more quantitative level to provide detailed data for improving the understanding or representation of processes.

Natural and archaeological analogue studies also serve to illustrate processes and interactions between components of the repository system, both in the near field (e.g. corrosion of iron and copper, degradation of bentonite, radionuclide retention in corrosion products and clays) and in the far field (e.g. radionuclide release, transport and retention, migration of an alkaline plume from cement, etc). The information below summarises some of the natural analogue studies from around the world and how they are used as supporting evidence for the concept and safety of deep geological repositories. Table 3-3 provides a summary and main references for these analogue studies. This list is not exhaustive; fuller coverage of this large and interesting subject is given in /Miller et al. 1994, 2000 and CSN 2005/.

3.3.1 Main conclusions from natural and anthropogenic analogues

Analogues for processes in repository systems

The natural uranium deposit at Cigar Lake (Canada) is the most complete analogue yet found to support the concept of geological disposal because of the multiple similarities to proposed geological repositories (see below). Confidence in the role of the geosphere, whether clay formation or crystalline rock, as the ultimate barrier preventing or retarding the release of radionuclides to the biosphere has been developed from natural analogue studies on this and other uranium deposits.

Table 3-3. Selected references for natural analogue studies based on their relevance for the KBS-3H/Olkiluoto safety assessment providing support for the performance of the repository system, components and safety-relevant processes. Fuller coverage of natural analogue studies is given in /Miller et al. 1994, 2000/ and /CSN 2005/.

Repository analogy	Natural analogue	Information available
	Repository system	
	Cigar Lake, Canada	/Cramer and Smellie 1994/.
	Oklo, Gabon	/Berzero and d'Alessandro 1990, Brookins 1990/
Transport processes	Poços de Caldas, Brazil	/Chapman et al. 1992/
	Palmottu, Finland	/Blomqvist et al. 2000/
	Koongarra, Australia	/Duerden 1990/
	Copper canister	
	Hyrkkölä, Finland	/Marcos 2002/
	Kronan cannon, Sweden	/Neretnieks 1986, Hallberg et al. 1987/
	Iron / steel components	
	Inchtuthil, UK	/Miller et al. 2000/
	Bentonite clay	
Transport barrier	Dunarobba, Italy	/Miller at al. 1994/
	Loch Lomond, Scotland	/Miller at al. 1994/
	Opalinus clay, Switzerland	/Nagra 2002a/
Longevity	Götland, Sweden and Sardinia, Italy	/Pusch and Karnland 1988/
	Kinnekulle, Sweden	/Pusch et al. 1998/
Chemical stability	Wyoming, USA	/Smellie 2001/
	Gulf of Mexico	/Eberl and Hower 1976, Roberts and Lahann 1981/
– Diagenetic illitisation	Kinnekulle, Sweden, Busachi, Italy and Cigar Lake, Canada	/SKB 2006c/
– Cement/bentonite interaction	Maqarin (Jordan)	/Alexander and Smellie 1998/
– Iron/bentonite interaction	Serrata de Nijar, Spain	/Marcos 2003, 2004/

Natural uranium deposits, such as those at Cigar Lake, Palmottu (Finland), Oklo (Gabon), Poços de Caldas (Brazil) and Koongarra (Australia), have been used to justify the assumption of a low rate of dissolution of spent fuel in safety assessment by comparing the UO_2 of the spent fuel to the naturally-occurring uraninite. These and other uranium deposits have also been useful to study other long-term processes such as:

- Mineral evolution and corrosion/dissolution processes of the uraninite (and pitchblende where present) as analogues of spent fuel.
- Redox processes and their role in radionuclide mobilisation and retardation.
- Radionuclide speciation and solubility, including the formation and behaviour of colloids, in a wide variety of groundwater conditions.
- Retardation processes affecting mobilised radionuclides, including sorption on fracture minerals and diffusion in the matrix, in a range of different rock types.
- Radionuclide mobility facilitated by colloids and microbial populations.

A brief description of the most relevant analogues in the context of the KBS-3H safety studies follows.

Cigar Lake, Canada

The Cigar Lake uranium deposit /Cramer and Smellie 1994/ is particularly interesting as it provides a large-scale analogue for a generic geological repository in hard, fractured formations. It is particularly relevant to spent nuclear fuel repositories because of the large amount of uranium, and very high uranium concentrations present, up to around 55% /Cramer and Smellie 1994/. It is also located at a depth similar to that of the Olkiluoto repository (–450 m). The uranium ore body, surrounded by a clay-rich halo, is located on the unconformity between old (Proterozoic) sandstones and underlying crystalline basement rocks, in an environment with active groundwater circulation.

The Cigar Lake uranium orebody was formed about 1,300 million years ago but even after this length of time, there are no geochemical signatures at the ground surface, illustrating the isolation capacity of the clay surround to the ore body and the host rock. The Cigar Lake analogue provides:

- Support for spent fuel stability under reducing conditions.
- Support for the conceptual model of hydraulic isolation and filtering of colloids by bentonite, as represented by the clay surround to the ore body.
- Support for the conceptual model of irreversible nuclide sorption on colloids.
- Support for the development of more highly elaborated radiolytic models.
- Quantitative data on the solubility of trace elements.

Cigar Lake does, however, not provide a good analogue for spent fuel dissolution. Reported uranium concentrations in the sampling points at the core of the Cigar Lake ore deposit are in the range 3 x 10^{-8} to 1 x 10^{-7} M, while the measured $H_2(g)$ was 0.04 cm^3/dm^3 /Cramer and Smellie 1994/. According to the spent fuel model in SR-Can fuel and canister report /SKB 2006f/, the solubility of $UO_2(s)$ under reducing conditions in presence of Fe and H_2 is 6.3 x 10^{-10} M. This apparent discrepancy in UO_2 solubilities can be explained considering that the Cigar Lake ore deposit comprises uraninite ($UO_{2.2}$) which is crystallographically the same as crystalline UO_2 but includes U(VI) atoms in its lattice. Due to the presence of U(VI) in the lattice, the solubility of uraninite is somewhat higher than that of UO_2 in spent fuel. According to the difference analysis approach presented in Chapter 1 of the KBS-3H Process report /Gribi et al. 2007/, the spent fuel model used in the KBS-3H safety studies was the same as that used in SR-Can. It is acknowledged, however, that alternative spent fuel dissolution models have been proposed and this is an issue for further study for both KBS-3H and -3V.

Oklo, Gabon

The 1.8 billion year old natural nuclear reactors at Oklo provide an excellent analogue for assessing the behaviour of fission products, actinides and actinide daughters in a fractured host rock.

The deposits are very rich U ore bodies (up to several tens of% U oxides) formed in the fracture system in sandstones and shales by the reduction of U-bearing solutions leached from lower grade ore deposits.

The data from Oklo support the view that very little of the radionuclide inventory migrated away from the sites of reaction during the 0.8 million years of reactor operation, despite a significant hydrothermal circulation, due to the low oxygen content (reducing conditions) of the hydrothermal fluid. Also, in the 1.8 billion years since the end of the reactor's activity, most of the radionuclides have either not migrated at all or have moved only a few metres. The exceptions are noble gases, halides (e.g. iodine) and alkali elements (e.g. Cs), which are more mobile, in agreement with performance assessment calculations for the repository system.

Poços de Caldas, Brazil

The volcanic caldera in Poços de Caldas hosts a deposit of uranium along with other metals. In addition to the type of information provided by Cigar Lake, Poços de Caldas provides evidence of radionuclide immobilisation by co-precipitation and sorption in oxyhydroxides of iron. Although these retardation and immobilisation processes cannot be quantified in a manner appropriate for application to radionuclide release calculations, this analogue and others confirm that they occur in nature. Radionuclides are also retained in the rock matrix causing both a decrease in radionuclide transport rate and in the maximum activity reaching the biosphere.

Palmottu, Finland

The Palmottu U-Th mineralisation is an interesting analogue in a geological context somewhat similar to Olkiluoto. The Palmottu Natural Analogue study /Blomqvist et al. 2000/ is based on a small, low grade, U-Th orebody in the form of U-bearing pegmatites and veins. Of particular interest is the very low U concentration in deep reducing groundwaters around the orebody of <10 ppb that contrasts with high concentrations of up to 500 ppb above the redox front (i.e. in oxidising conditions). The Natural Analogue Study focused on processes that may affect radionuclide migration and retardation in fractured metamorphic rocks similar to those which will host Finnish and Swedish repositories. The study examined:

- Stability and longevity of U minerals.
- Radionuclide transport by colloids.
- Redox processes.
- Radionuclide retardation by matrix diffusion.
- Blind predictive geochemical modelling.

One of the important parameters for matrix diffusion is the depth of the rock adjacent to a fracture that is affected by the radionuclide migration through the fracture (matrix diffusion depth). At the Palmottu uranium orebody, the matrix diffusion depth is about 25 mm, which indicates qualitatively the limited extent of migration of radionuclides inside the rock matrix. However, it should be noted that the matrix diffusion depth depends on several parameters and this value cannot be used directly in a quantitative assessment of the matrix diffusion process in the repository geosphere.

The Palmottu uranium deposit also provides evidence to support the stability of spent fuel in spite of the various alteration processes which occur over long times and confirms the potentially good isolation properties of the granitic bedrock.

Koongarra, Australia

The Koongarra uranium ore deposit provides a good analogue for the behaviour of the spent fuel in case of loss of canister retention and leaching of the uraninite under oxidising conditions, as well as evidence for radionuclide migration/retention and the movement of weathering fronts. The information gathered thus far provides support for the following:

- Immobilisation by precipitation, co-precipitation and sorption of radionuclides on secondary minerals (phosphates, clay minerals and oxyhydroxides of iron).
- Confirmation of the limited extent of matrix diffusion.

In the wider context of understanding processes which could affect repository evolution, an interesting scenario project was carried out as part of the larger analogue study at Koongarra. This project applied repository safety assessment scenario development methods to the evolution of the Koongarra site /Skagius and Wingefors 1992/. The aim was to facilitate the exchange of information between different scientific disciplines and the modellers, an issue of great importance to safety assessment. It was also expected that this approach could lead to better focussing of the Koongarra study evaluation on issues relevant to safety assessment.

Analogues for processes involving copper

The examples of the long-term durability of native copper in relevant conditions are illustrative evidence for the long-term stability of copper canisters in the repository environment. Several examples of copper occurrences have corroborated the expectation that the sub-surface conditions in the repository will preserve the copper canisters. Indeed, elemental copper has persisted for millions of years in several geological environments /e.g. Marcos 1989/ such as:

- In sedimentary rocks: Keweenaw Peninsula, Lake Superior region, Michigan, U.S.; Corocoro, Bolivia; south Devon, United Kingdom.
- In basaltic lavas: Keweenaw Peninsula; Appalachian States from central Virginia to southern Pennsylvania, U.S.; Coppermine River area, NWT, Canada; Dalane, Norway.
- In granitic rocks: Hyrkkölä and Askola, Finland.
- In the oxidised zones of sulphide deposits (many places in the world, including Finland; the eposits in Chile may be the best known) and in swamps.

At a very simple illustrative level, native copper in the surface of a boulder containing sulphides found near Outokumpu, Finland, shows only minor surface alteration to cuprite. Although the conditions of formation are not exactly known, its minimum age is about 10,000 years, i.e. since the last glaciation.

Hyrkkölä and Askola, Finland

Copper mineralisation at both Hyrkkölä and Askola is associated with the occurrence of uranium, extending the analogy with copper canisters for spent fuel in a repository. At the 1,700 My old U-Cu mineralisation at Hyrkkölä, near the Palmottu analogue site in Southern Finland, native copper and Cu sulphides occur in open fractures in crystalline rocks. This has allowed the study of sulphidisation and corrosion processes under conditions somewhat analogous to a repository. At Hyrkkölä, the copper is also associated with smectite, the main component of bentonite in the buffer, providing a further interesting feature in this analogue.

The Hyrkkölä analogue shows that, in the past, corrosion in both oxidising and reducing conditions has occurred. Corrosion in reducing conditions by sulphide ions (sulphidation) is identified as the key canister corrosion process during the post-closure evolution of the repository. Although sulphidation is no longer occurring, the mineral assemblage gives insights to the conditions in which it occurred. Most of the copper remained in its native state, indicating a slow copper corrosion rate, although the duration of sulphidation is not known. Oxidation in contact with groundwater containing dissolved oxygen (0.4 to 4 mg/L) is still occurring. The process started at least 10,000 to 100,000 years ago (based on indirect observations) and has been shown to proceed at a very slow rate, possibly due to passivation mechanisms. Studies of the behaviour of the Cu sulphide (djurleite) in the current highly oxidizing conditions, show that the djurleite has the ability to immobilise migrating uranium, most probably by electron exchange between Cu^+ in the external layers of copper sulphide and U (VI) sorbed as uranyl (UO_2^{2+}) on the mineral surface: Cu (I) oxidizes to Cu (II), whereas U (VI) is reduced to U (IV). No interaction of copper oxides with uranium has been observed /Marcos and Ahonen 1999, Marcos 2002/.

The persistence of native copper despite exposure to sulphide-containing groundwaters, which have been established for the last 300,000 years, as well as the current oxidising conditions, supports the expected durability of Cu canisters in the repository even though future conditions cannot be predicted.

Littleham Mudstone, U.K.

/Milodowski et al. 2002/ present an analysis of the corrosion of native copper plates that have survived in the Littleham Mudstone (UK) for more than 176 million years. Although the native copper is affected by corrosion, the study shows that a significant proportion (30–80% of the original thickness) of the copper plates is preserved in the water-saturated compacted clay environment of the mudstone. Apart from the recent weathering effects due to exposure at outcrop, petrographical studies demonstrate that most of the observed corrosion and alteration of the native copper is geologically old (i.e. predating the main sediment compaction) and also occurred before the end of the Lower Jurassic. This analogue demonstrates that the native copper can remain stable in a saturated and compacted clay environment for geological timescales well in excess of the timescales considered for performance assessment of a deep geological repository for spent nuclear fuel.

The Kronan cannon, Sweden

The Kronan bronze cannon from the Swedish man-o-war "Kronan" was partially buried muzzle down in shallow marine clay when the ship sank in 1676. This archaeological analogue illustrates the corrosion resistance and longevity of copper in oxidising conditions. The Kronan cannon has a very high copper content (96.5%, with 3.3% tin and 0.5% iron). The marine clay in which the cannon was almost totally buried may be considered as an analogue of bentonite. The head of the cannon protruded from the clay and was in direct contact with seawater. The average corrosion rate (0.15 microns per year over 300 years) of the buried part of the cannon was determined based on the time since the warship sunk. The oxidation products identified (CuO_2, Fe_3O_4) are in agreement with models for corrosion in moderately oxygenated water – rather different to the expected repository environment but a useful indicator for the possible case of oxygenated groundwaters reaching repository depth due to glaciation. The corrosion rate is very low despite the fact that the conditions would be expected to be favourable for copper corrosion.

In summary, natural and archaeological analogues provide evidence to support the long-term durability of the copper canisters in a variety of hydrogeochemical conditions, both oxidising and reducing.

Iron/steel components

Inchtuthil, Scotland

A hoard of more than 1 million iron nails at Inchtuthil, Scotland, was excavated from a 5 m deep pit covered with 3 m of compacted earth where they were buried as the fort was abandoned by the Romans in 87AD. The outer and shallower nails were severely corroded but the inner ones showed minimal corrosion, limited to the formation of a thin passivating layer on the surfaces. This was interpreted as due to the strong redox buffering provided by the outer layer of nails, which conditioned infiltrating groundwater so that it was chemically reducing by the time it came into contact with the nails at the centre of the hoard. This is analogous to the redox buffering by the steel supercontainer and iron inserts within the repository EBS which is expected to provide reducing conditions in the eventuality of oxidising water reaching the repository level or radionuclides being released from the spent fuel.

Corrosion of iron/steel components in contact with bentonite and their impact on bentonite are discussed further under analogues of processes in bentonite.

Analogues of processes in bentonite

The best studied natural analogues for processes in bentonite are briefly described below.

Dunarobba, Italy

The preservation of wood for over 2 million years by a clay formation covering the Dunarobba forest illustrates the isolating capacity of clay and its ability to limit microbial degradation of organic materials. The 1.5 My old trees are still in an upright position and enveloped in lacustrine clay above which there are permeable sand deposits with oxidising groundwater. The trees are still composed of wood that has been protected from decay by the clay barrier. Further information can be found in /Miller et al. 2000/.

Loch Lomond, Scotland

At present, Loch Lomond is freshwater and landlocked but a marine incursion into the Firth of Clyde between about 6,900 and 5,400 years ago left a band of marine sediment approximately one metre thick. These sediments are clay-rich, up to 80% in some horizons. The chemistry of the porewaters in the sediments identifies them as marine, with higher concentrations of chloride, bromide and iodide than the surrounding fresh groundwaters. Migration of these anions into the freshwater sediments above and below the marine band records a history of diffusive transport. The Loch Lomond analogue illustrates the expected slow diffusion of radionuclides through a clay barrier analogous to the bentonite buffer.

Opalinus clay, Switzerland

The Opalinus clay (and the overlying rock unit) was found to show a similar diffusive gradient of porewater stable isotope ratios (δO-18 and δH-2), albeit over much larger time and distance scales than the Loch Lomond example, and this was used to test the diffusive transport barrier function of the Opalinus clay. For further information see /Nagra 2002a/. As with the Loch Lomond analogue, the Opalinus Clay analogue illustrates the expected slow diffusion of radionuclides through a clay barrier analogous to the bentonite buffer.

Götland, Kinnekulle, Sweden and Sardinia, Italy

The Kinnekulle and Sardinia bentonites are similar enough in mineralogy to the bentonite used in the buffer to provide good analogues for illustrating its longevity and thermal stability. Despite the thermal degradation of smectite at some of these sites (e.g. the maximum temperature was 160ºC at Kinnekulle) to illite and the presence of cementation by silica, the essential properties required of the bentonite barrier were preserved. The results of these studies /e.g. Pusch and Karnland 1988/ help to bound the conditions required for bentonite alteration, particularly with respect to thermal alteration. The observation of the temperature-dependency of silica cementation and alteration to beidellite, both of which could affect the favourable swelling and sealing properties of bentonite in the EBS, supports the general design guideline to keep the temperature ≤ 100 °C in the KBS-3H repository.

Wyoming "MX-80" bentonites, USA

The type of bentonite proposed as buffer material for KBS-3H is MX-80, which is the commercial name of a type of bentonite mined from the Clay Spur bed at the top of the Cretaceous Mowry Formation in NE Wyoming, USA. According to the study by /Smellie 2001/, the Wyoming bentonites have been in long-term contact with reducing waters of brackish to saline character and thus provide an interesting analogue for repository bentonite under similar conditions.

Bentonites are the product of pyroclastic fall deposits thought to be generated by the type of explosive, sub-aerial volcanic activity characteristic of Plinian eruptive systems. In Wyoming, the ash clouds were carried to high altitudes and eastwards by the prevailing westerly winds before falling over the shallow Mowry Sea and forming thin but widespread and continuous horizons above sea floor muds and sands. The Mowry Sea environment included mineral-rich groundwaters which were brackish and partially reducing.

Under these initial aqueous conditions, the newly-formed bentonite appears to have been at equilibrium. Subsequent rapid deposition of impervious mud/silt has served to isolate the bentonite from alteration during the continued palaeo-evolution of the Mowry Sea basin. Based on available evidence, it would appear that in general most of the Wyoming bentonites studied have undergone no major post-depositional alteration unless exposed to surface/near-surface weathering processes. However, because of their physico-chemical isolation since deposition, it is not possible to study the effects of post-formational alteration of the bentonites under varying hydro-geochemical conditions during the palaeo-evolution of the Mowry Sea basin in Cretaceous times. Thus, while the bentonites provide a good analogue of long-term stability in a closed system, there is insufficient information to evaluate their long-term behaviour in an open system in contact with brackish to saline waters.

Gulf of Mexico, Busachi, Italy, Kinnekulle, Sweden, and Cigar Lake, Canada

These studies of diagenetic illitisation of montmorillonite clays in sediments from the Gulf of Mexico are not entirely applicable to the repository situation because the duration of heating is far longer than expected for the repository thermal phase. However, the studies were useful to confirm that the rate of illitisation is strongly temperature-dependent and slower in the natural environment than predicted from kinetic studies, possibly due to the limited supply of potassium. The studies support the conclusion that illitisation of a significant proportion of the bentonite would require a period of some tens of millions of years.

Maqarin, Jordan

Despite intense interest in the behaviour of bentonite under hyperalkaline conditions, such as could occur due to groundwater interaction with the cement used as structural material, there are not yet any convincing natural analogue studies in this area. The Maqarin natural hyperalkaline groundwater system is often mentioned but it lacks in appropriate clay minerals in the marls which form the host rocks to the hyperalkaline system. For further information see /Alexander and Smellie 1998/.

Serrata de Nijar, Spain

The interaction between bentonite and corroding iron is a topic of interest to the KBS-3H design, in particular. However, the study by /Marcos 2004/ of bentonite samples from outcrop at Serrata de Nijar unfortunately found no systematic variation in the Fe-content of the samples that could be explained as a function of the weathering depth of the samples. No information could be derived which allowed interpretations relevant to a repository environment. Currently, there is no good analogue to provide information in this area.

Other studies

/Wersin et al. 2007/ summarised the most relevant information on bentonite-iron interaction: Fe-rich saponites are formed at higher temperatures under hydrothermal conditions and Fe-phyllosilicates such as berthierine form in presence of ironstones. Berthierine acts as a precursor to chlorite but the transition to the latter form occurs at temperatures of 70–130 °C. Direct chloritisation of smectites occurs at even higher temperatures (150–200 °C). Meteorites rich in metallic iron (chondritic meteorites) embedded in silicate spherulitic matrix could be useful natural analogues but more information is needed.

In summary, natural analogues can provide good evidence for the isolation, retardation and retention properties of the buffer although, in most cases, the analogue materials do not closely represent the conditions expected in the repository, especially with respect to the high degree of compaction of the buffer bentonite. Regarding stability, the existence of bentonite in nature in a wide variety of environments and over very long timescales is in itself the most valuable natural analogue for the buffer stability. Unfortunately, the current database of natural analogues does not provide any relevant information on iron/bentonite or cement/bentonite interaction processes which are of interest to a KBS-3H type repository. Information from numerous other analogue studies is summarised in the SR-Can report on buffer and backfill processes /SKB 2006c/.

3.4 Support for the concept from other safety assessments

Safety assessments carried out worldwide for a large number of different host rocks and waste types over a number of years have resulted in the development of internationally accepted practices for the making of safety cases for geological disposal of radioactive wastes. The activities of international bodies, such as the NEA, have enhanced this process by bringing together groups of experts in different fields to document their experience /e.g. NEA 1997, NEA 2004a, and NEA 2007ab/. This means that recent safety assessments can be reviewed by teams of international specialists, as well as national regulatory authorities, to ensure that they comply with current "best practice" methodology, including the best available scientific data, models and calculational codes. In particular, safety cases and assessments for spent fuel disposal concepts in different host rocks, including granite (or granitic crystalline rocks) and sediments, which have received international reviews (e.g. the NEA peer review groups) as well as, where it was required, regulatory approval, include those of:

- SKI, Sweden (SITE-94, /SKI 1996/) – the first ever NEA review.
- Ontario Power Generation, Canada (OPG-TCS, /Gierszewski et al. 2004/).
- Andra, France (Dossier 2005 Granite, /Andra 2005a/ – mainly for vitrified HLW).
- Enresa, Spain /Enresa 1998/.
- Posiva (TILA-99, /Vieno and Nordman 1999/).
- SKB, Sweden (SR-97, /SKB 1999/, SR-Can, /SKB 2006a/).
- Ondraf/Niras, Belgium (Safir 2, /Ondraf/Niras 2001/ – mainly for vitrified HLW).
- Nagra, Switzerland (Project Opalinus, /Nagra 2002a/).

Although these safety assessments have been made for different disposal concepts, host rocks and geological settings, there are large overlaps in the arguments and analyses required to demonstrate the feasibility and long-term safety of geological disposal of spent fuel in the specific situations. The existence of such a body of scientific and technical experience from successfully-made safety assessments, upon which other repository programmes can draw, adds considerable support to the case being made for safe and feasible geological disposal using KBS-3H even though the specific design, using supercontainers and distance blocks, is relatively novel.

The more relevant safety assessments addressing KBS-3V specifically (i.e. TILA-99 and SR-Can) are described in more detail in Chapter 5, where differences and similarities between them and the KBS-3H safety studies are discussed in order to assess, for example, how the changes from the vertical to the horizontal geometry, the influence of the supercontainer and the implications arising from improved site-specific data are carried over into the KBS-3H safety studies.

4 Understanding of the Olkiluoto site

The location of the Olkiluoto site is shown in Figure 4-1 along with that of the Swedish candidate sites at Forsmark and Laxemar that were the subject of SR-Can. The purpose of the present report is not to describe in detail the Olkiluoto site or its evolution. The following sections provide a brief description of the site to enable the reader to understand the information related to the site in the following chapters. Section 4.1 provides the current status of knowledge of the site and the main uncertainties are discussed in Section 4.1.6.

Current knowledge of the site is built on surface-based investigations carried out over the last 20 years giving the baseline conditions at the site, as well as surface-based and underground monitoring over several years. This knowledge will be strengthened as the research programme at the underground rock characterisation facility (Onkalo) and the Olkiluoto site characterisation programme proceed. The monitoring programme is briefly summarised in Section 4.2. The emphasis is on the significance of the observations from Onkalo (still under construction), in particular those made at repository depth (–420 m), with respect to long-term safety. It is expected that Onkalo excavations will reach repository depth around 2009 /Posiva 2006/.

The characteristics of the site are being perturbed by the construction of Onkalo and will continue to be perturbed by the construction and operation of the repository itself. Understanding these disturbances is essential to understanding the site and also the repository evolution. The summary of the hydrological and hydrogeochemical disturbances is presented in Section 4.3.

Finally, complementary lines of evidence on site suitability are presented in Section 4.4, based on considerations of long-term geological stability of the site, the absence of natural resources and comparison of the site characteristics to those discussed in other safety assessments based on crystalline host rocks.

4.1 Summary of the current knowledge on the Olkiluoto site

A comprehensive overview of site information gathered over almost 20 years is given in the Olkiluoto Site Description report /Andersson et al. 2007/. A description of the evolution of the Olkiluoto site for the KBS-3V design alternative is presented in /Pastina and Hellä 2006/. On the basis of this report, and following the approach of the SR-Can Main report /SKB 2006a/, the evolution of a KBS-3H repository is presented in /Smith et al. 2007a/. The following is a simplified summary of the information in these reports.

4.1.1 Geological setting

Topography and hydrological setting

Olkiluoto is a relatively flat island with an average height of 5 m above sea level and the highest point 18 m above sea level. The island is covered by forest and shoreline vegetation. The sea around the island is shallow with a depth mainly less than 12 m within 2 km of the current shoreline. The elevations relative to sea level are continuously changing since the apparent rate of uplift is significant at 6 mm per year, mainly due to isostatic adjustment of the bedrock (see Section 4.1.5).

The overburden, both onshore and offshore, is mostly till. The other terrestrial sediment types are, in order of abundance, fine sand, sand and silt, with the thickness of the overburden mostly being between 2 and 4 m, although deposits up to 12 to 16 m in thickness have been observed in rock surface depressions /Lahdenperä et al. 2005, Posiva 2005/. The groundwater table follows the surface topography and is mainly 0 to 2 m below the surface. Since surface waters flow directly into the sea /Lahdenperä et al. 2005, Posiva 2005/, Olkiluoto Island forms its own hydrological unit.

Figure 4-1. The location of the Olkiluoto site, which is the subject of the present study, and the Forsmark and Laxemar sites, which were the subject of SR-Can.

Infiltration rate of surface water is currently being investigated; this parameter has to be measured indirectly and supported by modelling. So far, hydraulic connections have been found to be weak between the overburden and bedrock. Current (and provisional) estimates are approximately 1–2% of the annual precipitation. The evolution of surface conditions and ecosystem has been described in more detail in the biosphere assessment report /Ikonen 2006 and references therein/.

Geology

The bedrock at Olkiluoto belongs to the Svecofennian domain of Southern Finland and comprises a range of high-grade metamorphic rocks and igneous rocks. The metamorphic rocks include various migmatitic gneisses and homogeneous, banded or only weakly migmatised gneisses, such as mica gneisses, quartz gneisses, mafic gneisses and tonalitic-granodioritic-granitic gneisses. The igneous rocks comprise abundant pegmatitic gneisses and sporadic narrow diabase dykes /Paulamäki et al. 2006/.

At Olkiluoto, three different alteration episodes can be identified which have affected the chemical composition and mineralogical character of the altered rocks and, as a consequence, the physical properties of the bedrock /Andersson et al. 2007/:

- A *retrograde phase of metamorphism*, which affected the bedrock during the Svecofennian Orogeny about 1,900 to 1,800 million years ago.
- *Hydrothermal alteration processes*, which are estimated to have taken place at temperatures from 50ºC to slightly over 300ºC and are thought to be related to the late stages of metamorphism, to the emplacement of rapakivi granites 1,580 to 1,570 million years ago and to the intrusion of olivine diabase dykes 1,270 to 1,250 million years ago.

- *Surface weathering*, which probably dates back some tens of millions of years and is still currently active.

The bedrock was deformed in a ductile manner during the Svecofennian Orogeny and was subsequently affected by several tectonic events that resulted in brittle deformation.

The location of the site in the Fennoscandian shield, and especially in Finland, is advantageous with respect to the stability of the geosphere /Marcos et al. 2007/. Recorded earthquakes in Northern Europe since 1375 are shown in Figure 4-2. The figure shows that the density and magnitude of earthquakes in Finland is lower than in other areas. Earthquake magnitudes in Finland have never reached 5 on the Richter scale since records began in the 1880s /Marcos et al. 2007 and references therein/. At a regional level, seismic activity in the Olkiluoto region is currently low /see e.g. La Pointe and Hermanson 2002, Enescu et al. 2003, Saari 2006/. Seismic studies at Olkiluoto show negligible rock movements /Andersson et al. 2007/ but seismic activity in the future cannot be excluded. Major seismic activity is likely to occur with the greatest frequency following glaciations, although infrequent but significant seismic events during inter-glacial periods are also possible.

4.1.2 Rock fracturing and groundwater flow

Polyphase deformation has resulted in a network of fractures and fracture zones with different scales within the Olkiluoto bedrock. The frequency, spatial distribution, size distribution, shape and orientation of these structures affect both the hydraulic and mechanical properties of the rock. The fracture zones often constitute dominant paths for groundwater flow and their size also determines the size of rock shear movements taking place in the zone. As noted above, hydrothermal alteration has occurred in certain domains in the rock mass and this also affects its strength and transport properties.

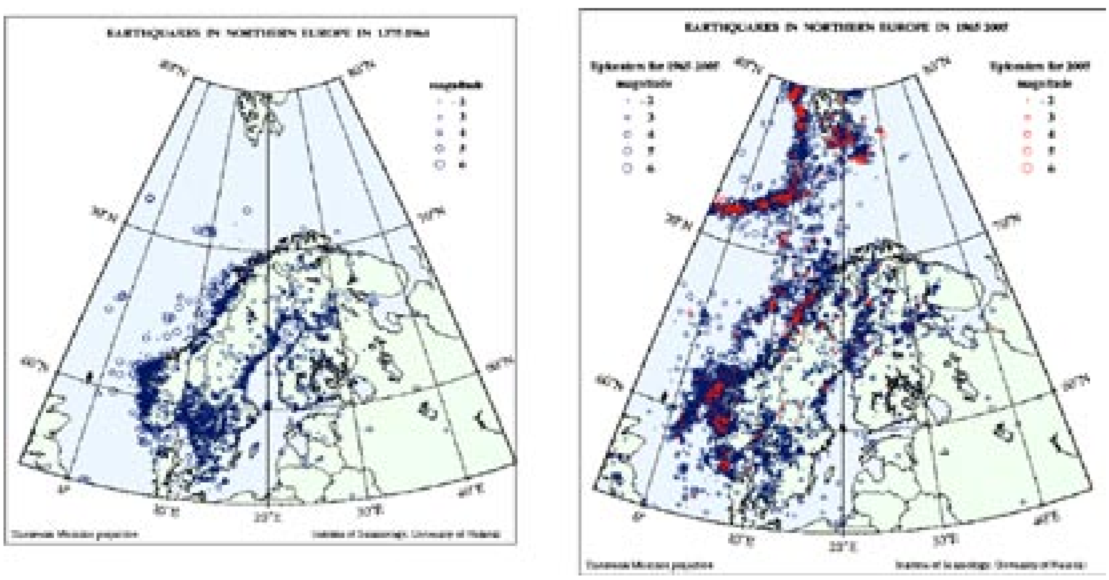


Figure 4-2. *Earthquakes in Northern Europe in 1375–1964 (left) and in 1965–2005 (right). Note that the density and magnitude of earthquakes in Finland is lower than in other areas in Northern Europe. (Source: University of Helsinki[13]).*

[13] Source for left figure: http: //www.seismo.Helsinki.fi/bulletin/list/catalog/histomap.html.
Source for right figure: http: //www.seismo.Helsinki.fi/bulletin/list/catalog/instrumap.html.

Information has been gathered on the occurrence, frequency and orientation of transmissive fractures at Olkiluoto. To date, the focus of the hydrogeological modelling has been on identifying and characterising the major hydraulically active deformation zones, whereas the rock masses between these zones have been given average hydraulic properties. However, for assessing the evolution of the engineered barrier systems and for modelling the performance of the geosphere transport barrier, it is also necessary to describe the flow system at the scale of individual fractures. Detailed data on the fractures exist but so far the analyses of these data have focused on those fractures reflecting likely conditions in the deposition drifts /Hellä et al. 2006/.

Groundwater flow at Olkiluoto is concentrated in the near-surface part of the rock and in transmissive fractures at depth. The latter are likely to intersect the deposition drifts at various locations and will lead to water inflow and saturation of gas-filled voids during the early, transient phase. Transmissive fractures at relevant depths, especially those with transmissivities higher than 10^{-8} m^2/s, are concentrated mainly in local zones of abundant fracturing. Fractures with lower transmissivities occur outside these zones, but also tend to form clusters /Hellä et al. 2006/. The rock matrix between fractures has an average porosity of 0.14% /Autio et al. 2003/ and a low hydraulic conductivity so that water fluxes through it are negligible compared to those through fractures.

Based on the analysis of available hydrogeological (borehole) data in /Hellä et al. 2006/, flow conditions in a deposition drift have been estimated:

- The total leakage into a compartment may be up to 10 litres per minute[14]. (Locations with worse conditions may be avoided for spent fuel deposition).
- The average frequency with which fractures with transmissivities greater than 10^{-9} m^2/s intersect a drift is 4 per 100 m (10^{-9} m^2/s is the detection limit – fractures of this transmissivity would be expected each to give rise to an initial inflow of about 0.04 litres per minute into the drift).
- The initial inflow into 5 m drift intervals (corresponding to the length of a KBS-3H supercontainer) is less than 0.1 litres per minute over more than 90% percent of the drift length.
- The initial inflow into 10 m drift intervals (corresponding to the length of a KBS-3H "supercontainer unit" comprising one supercontainer and one distance block) is less than 0.1 litres per minute over about 85% percent of the drift length.
- There are long sections (100 m or more) of the drift that are intersected by no fractures with transmissivities greater than 10^{-8} m^2 /s.

The estimates of /Hellä et al. 2006/ may be revised as a result of the ongoing detailed site characterisation work at Onkalo and the associated modelling.

The fractures at Olkiluoto are coated or filled by minerals; in particular, calcite and a range of clay minerals (illite, smectite, kaolinite, vermiculite and chlorite) make up most of the fracture filling. Pyrite coatings in fractures are also abundant, mainly as coatings on calcite grains. Pyrite has been observed in all boreholes studied so far at the site. These fracture fillings play an important role in the hydrogeochemical conditions at Olkiluoto and their evolution. Locally, rock matrix minerals may also be exposed on fracture surfaces /Luukkonen et al. 2004, Pitkänen et al. 2004/. Trace element data from fracture fillings are scarce. Some results are presented in the fracture calcite study of /Gehör et al. 2002/ during the EQUIP project, although this study did not analyse trace elements precipitated in calcite. A qualitative estimate of heavy metal concentrations within gouge minerals was carried out by /Gehör 2007/. Trace elemental monitoring (in addition to U) to establish the elemental cycling baseline is currently under planning as part of the site characterisation activities.

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[14] From Figures 16 and 17 in /Hellä et al. 2006/. It has, however, been observed at Äspö and in the interim storage facilities for low-level waste at Loviisa and Olkiluoto (VLJ repositories) that inflows have a tendency to decrease over time, possibly due to mineral precipitation in fractures /see Hagros and Öhberg 2007/. See also Section 7.5 in the Process Report /Gribi et al. 2007/.

4.1.3 Groundwater composition

The geochemistry of the Olkiluoto site has been extensively investigated by geochemical and mineralogical analysis of a number of deep boreholes. /Posiva 2003a, Posiva 2005, Andersson et al. 2007 and Pitkänen et al. 2004/ give a comprehensive picture of the hydrogeochemical conditions at Olkiluoto. Generally, chemical conditions in the groundwater are stable at depth at Olkiluoto; reactions and transport processes proceed slowly but will be perturbed by the presence of the repository and by external events occurring in the far future, such as major climate change. The groundwater composition over the depth range 0 to 1,000 m at Olkiluoto is characterised by a significant range in salinity (see, for example, Figure 11-8 in /Andersson et al. 2007/). Fresh groundwater with low total dissolved solids (TDS less than about 1 g/L) is found only at shallow depths, in the uppermost tens of metres. Brackish groundwater, with TDS up to 10 g/L dominates at depths between 30 m and about 400 m. Saline groundwaters (TDS > 10 g/L) dominate at still greater depths. The current salinity of groundwater at repository depth (400 to 500 m below ground) ranges from 10 to 20 g/L TDS /Andersson et al. 2007/. Chloride is normally the dominant anion in all bedrock groundwaters. Near-surface groundwater is also rich in dissolved carbonate and groundwater at depths between about 100 and 300 m is characterised by high sulphate concentrations. Both carbonate and sulphate concentrations decrease significantly at greater depths. Sodium and calcium dominate as main cations in all groundwaters and magnesium is notably enriched in sulphate-rich waters.

The ions dominating in different groundwater types reflect the origins of their salinity. In crystalline rocks, high dissolved carbonate content is typical of meteoric groundwaters that have infiltrated through organic soil layers. High sulphate content indicates a marine origin in crystalline rocks without sulphate mineral phases. More generally, the wide groundwater salinity variations at Olkiluoto can be interpreted in terms of varying degrees of mixing of certain reference water types, together with a range of water/rock interactions that buffer pH and redox conditions and stabilise groundwater composition. The reference water types are present-day Baltic seawater and four different groundwater types, termed, in order of decreasing age, brine reference, glacial reference, Littorina (Sea) reference and meteoric water. The groundwater flow and composition near to the surface are characterised by a dynamic hydraulic regime and a significant imprint of young meteoric waters. Below about 300 m depth, studies of methane indicate that the deep stable groundwater system has not been disturbed by glacial and post-glacial transients and that neither oxidising glacial meltwater nor marine water have mixed in this deeper system.

The redox conditions are illustrated in Figure 4-3 (taken from Pitkänen et al. 2004). Microbially-driven sulphate reduction is the dominant redox reaction between 100 and 400 m depth, whereas methanogenesis accompanied by high hydrogen levels predominates below this. Isotopic data suggest that at around 400 m concomitant sulphate reduction and methane oxidation occur. In this zone, sulphide is enriched (a few mg/L but up to 12 mg/L has been measured), whereas Fe concentrations are low. Below 400 m, sulphide concentrations drop to insignificant levels while Fe concentrations show an increase with depth, reaching several mg /L in the brine-type waters.

The formation and accumulation of CH_4 in groundwater are evaluated in /Pitkänen and Partamies 2007/. Methane in the groundwater is thought to have two primary sources. Thermal abiogenic hydrocarbons (a crustal inorganic carbon source without biogenic processes) dominate at greater depth where the highest CH_4 contents are observed. At repository level, biogenic CH_4 seems to dominate; total contents are smaller and far from saturation but the CH_4 mass is clearly higher than the mass of sulphate in sea water. Relative (in addition to absolute) decrease of CH_4 upwards in the methanic groundwater layer indicates that mixing and dilution of saline groundwater may have been more rapid than the accumulation of CH_4 (either by diffusion of abiogenic methane or biogenic production) at repository depths.

Chemical and isotopic data for saline groundwater, however, indicate that mixing of groundwater has been a slow process. Abiogenic CH_4 is not believed to derive from the Earth's mantle. More probably this CH_4 was generated in the bedrock under hydrothermal conditions (at least 200–300°C). Fluid inclusions in quartz grains have very high CH_4 contents, possibly indicating a very slow diffusion from the rock /Pitkänen and Partamies 2007/.

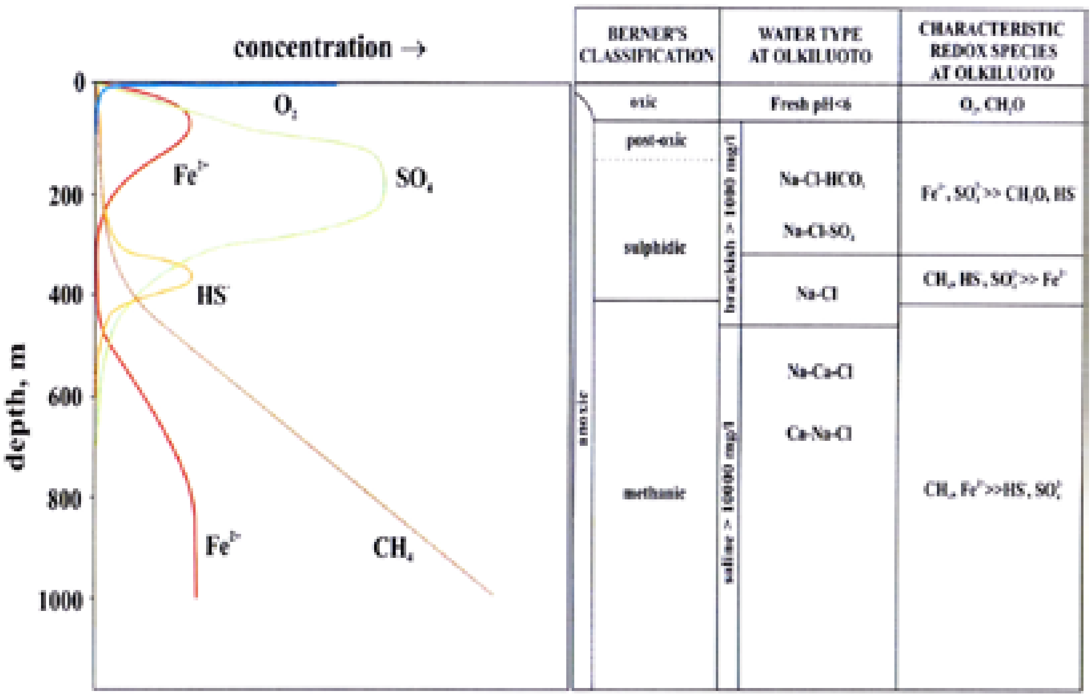

***Figure 4-3.** Redox zones at Olkiluoto as function of depth /from Pitkänen et al. 2004/.*

Microbial studies have been carried out on a few samples from deep boreholes /Haveman et al. 1998, 2000/ and recently from the shallow boreholes and overburden /Pedersen 2006, 2007/. Microbial activity was found to be low, which is typical for nutrient-poor deep crystalline environments. Sulphate-reducing bacteria were generally the most abundant species but iron reducers were also detected. The presence of methanogens and acetogens was noted but, because of the low amount of cultivatable cells, interpretation of these data is hampered. The presence of autotrophic methanogens (which use inorganic carbon together with hydrogen) in deep saline samples, together with high amounts of dissolved hydrogen and methane gases, indicate that carbonate reduction is an important process at great depths. Deep waters are rich in both CH_4 and H_2 but low in bicarbonate (i.e. CO_2 and dissolved inorganic carbon). According to carbon isotopic analyses results, reduced carbon species (i.e. methane) in deeper layers of groundwater are of old origin and formed via an abiogenic pathway. At the present, biogenic production in deeper water layers is hindered by the lack of dissolved bicarbonate /Pitkänen and Partamies 2007/.

The pH conditions in the deep aquifer system at Olkiluoto are well buffered by the presence of abundant carbonate and clay minerals found in fracture fillings. The pH values at relevant depths are generally in the range 7.5–8.5 /Pitkänen et al. 2004/. The redox conditions are also well buffered by the presence of iron sulphides and microbially-mediated redox processes. The estimated Eh values in the sulphidic zone are in the range of –200 to –250 mV (vs. SHE, Standard Hydrogen Electrode). In the methanogenic (carbonate reduction) zone, the Eh is estimated to be about –300 mV.

There are indications at Olkiluoto that groundwater at repository depth includes a small component, less than 20%, of glacial meltwater /Andersson et al. 2007/. Glacial meltwater could be from the latest glaciation or also from earlier ones. This glacial meltwater component is thought to be the result of slow mixing of water layers rather than by direct intrusion from the surface. The migration of oxygen dissolved in glacial meltwater to repository depth is, however, unlikely due to possible microbial activity and interaction with minerals in the rock. The recent interpretation of the hydrogeochemical site data in the Site Description 2006 report /Andersson et al. 2007/ and, particularly, gas isotopic data from Olkiluoto by /Pitkänen and Partamies 2007/ show no evidence of direct oxidising meltwater intrusion into the deeper groundwater system at

Olkiluoto. Nevertheless, although the glacial meltwater component is not presently very large, it may have been larger in the past and the possibility of future infiltration of glacial meltwater cannot be ruled out.

4.1.4 Rock stress

The stress state at the Olkiluoto site has been determined for different depths /Johansson et al. 2002a, Posiva 2005, Andersson et al. 2007/. According to the repository layout principles /Johansson et al. 2007/, the repository drifts will be aligned as much as possible with the direction of the maximal horizontal stress for reasons of mechanical stability. Regional data indicate that the mean orientation of the maximum horizontal stress is roughly E-W but the data display a large scatter, so that it is currently uncertain whether the stress orientation at the site differs from the mean regional orientation. At 500 m, the maximal horizontal stress is estimated to be between 15 and 31 MPa and the minimum horizontal stress[15] is estimated to be in the range 10 to 18 MPa. The vertical stress is estimated to be between 7 and 15 MPa at 500 m. The major principal stress is sub-horizontally orientated, and is thus slightly larger in magnitude than the maximum horizontal stress. The other two principal stress components vary significantly in magnitude and orientation between the different measurement locations, indicating the need to relate the stress field to geological structure and to conduct associated numerical analyses.

4.1.5 Post-glacial adjustment

During the last glacial maximum, 17,000–22,000 years ago, the Fennoscandian ice sheet reached as far south as northern Germany. The thickness of the ice sheet at that time is thought to have been about 2 km over Finland /Lambeck et al. 1998/. The weight of the ice mass acting on the viscous mantle caused the Earth's crust to sink some hundreds of metres. As the ice sheet started to melt about 13,500–10,300 years ago, the crust started to rise. The crust is currently still in the process of returning to its position of isostatic equilibrium.

The most obvious consequences of postglacial adjustment in Fennoscandia are the land uplift along both sides of the northern part of the Baltic Sea and the concomitant retreat of the shoreline. Olkiluoto Island began to emerge from the Baltic Sea about 3,000 –2,500 years ago. Currently, the rate of isostatic post-glacial uplift at the site is estimated to be 6.8 mm per year /Johansson et al. 2002b, Kahma et al. 2001, Eronen et al. 1995/. The apparent uplift rate, which is the rate of isostatic uplift minus the eustatic component due to sea-level change associated with the changing shapes of the sea basins, is 6 mm per year (this does not include the impact of global sea level change). The land uplift rate is expected to vary little over the next few centuries but will decrease significantly within the next few thousand years /Ruosteenoja 2003/.

4.1.6 Summary of site knowledge and remaining uncertainties

The current state of knowledge about the site, as summarised in this chapter, leads to the following statements about the Olkiluoto site (based on /Andersson et al. 2007/ and /Pastina and Hellä 2006/):

- Surface conditions and geology are well understood although some uncertainties remain.
- Rock mechanical properties and the status of in situ stress are fairly well understood but the long-term rock mechanical evolution presents uncertainties.
- The rate of local groundwater flow at the planned repository depth is low.
- Geochemical conditions at repository level are favourable to the engineered barrier system: reducing conditions, pH above 6 and below 11, low concentration of corrosive agents (e.g. sulphide, ammonia), low content of organics, moderate salinity levels of about 10–20 g/L.

[15] The minimal principal rock stress is also relevant to the early evolution of the repository in that it may affect, for example, the maximum gas pressures that can develop around the repository as a result of gas generation by the corrosion of steel components (Section 5.5).

- The number of major, fast transport pathways is low and their characteristics are known.
- The impact of the Onkalo construction on the site thus far is understood and engineering measures to minimise any negative effects are being or have been identified.

The modelling carried out to describe the hydrogeological and hydrogeochemical status of the site involves uncertainties. These are described in the latest Olkiluoto Site Description 2006 /Andersson et al. 2007/. Assessing the level of confidence in the modelling results is essential. Protocols similar to those developed by SKB /e.g. SKB 2006a/ have been applied to assess data, conceptual models, general understanding of the site and its evolution, and the need for alternative models, as well as to check the consistency between models used in different disciplines and between different model versions /Andersson et al. 2007/.

Olkiluoto site characterisation activities have been ongoing for over 20 years and there is an increasing level of confidence in the Olkiluoto site description; the main remaining challenge of the site characterisation work is to properly assess the confidence in the description outside the well characterised Onkalo volume. Other uncertainty issues concern aspects of the characterisation of the rock at the detailed scale; plans and actions have been identified for further work in this area. In short, the remaining development needs are the following /Andersson et al. 2007/:

- There is basically a very good understanding of the rock volumes near Onkalo, whereas the data density is still much less to the east of the site. It will be essential to enhance transferability of findings from Onkalo in order to make confident predictions of the conditions to the east.
- There is significant progress in the integration between the hydrogeological and geological modelling and further advances are expected in Site Report 2008.
- There are abundant data concerning connectivity and transmissivity of fracture networks but they are not yet fully evaluated – although the analyses by /Lanyon and Marschall 2006/ are a good starting point.
- The origin and evolution of groundwater composition are generally well understood, as illustrated by the overall consistency between the hydrogeological and hydrogeochemical descriptions and data, but further actions may still be needed to ensure confidence in predicting the potential for infiltration of dissolved oxygen or very dilute groundwater.
- The rock stress state is generally well understood but details of its spatial variability may still be critical for understanding e.g. the potential for spalling.

4.1.7 Summary of site evolution

The main events related to the site evolution from the beginning of the repository operation up to the end of the next glacial cycle are described below. More details are provided in /Smith et al. 2007a/ and in /Pastina and Hellä 2006/. The main hydraulic and hydrogeochemical events considered during the evolution are:

- Drawdown of the water table, surface water infiltration and up-coning of saline groundwater from the deeper layers of the bedrock during the operational phase due to the open excavations.
- Recovery of the flow and salinity field during the early post-closure thermal phase.
- Saline water retreat due to land uplift in the rest of the post-closure thermal phase.
- A stagnant flow regime during the permafrost and, in the case of an advancing ice sheet, possible up-coning of saline waters.
- Enhanced surface flow with potential glacial meltwater intrusion during the glacial melting period.
- Possible intrusion of seawater (fresh/brackish) during the submerged period.
- Crustal depression and land uplift during and after the ice sheet period.

Two climate scenarios have been selected for site-related climatic effects: the Weichselian-R scenario, based on the repetition of the latest glacial cycle, and the Emissions-M scenario (also called the Greenhouse variant in SR-Can), based on climate changes due to moderate levels of emission of anthropogenic CO_2. The evolution of the site according to the two climate scenarios is the same except for the time scale at which climate-related events occur. The differences appear only after the onset of the first permafrost period in the Weichselian-R, at approximately 15,000 years AP.

The operational and the early post-closure phases are the most eventful for the geosphere until the glacial melting phase. The thermal effects on the host rock from the spent fuel decay heat are significant only several hundred years after closure of the repository and last about 15,000 years. Thermal gradients also accelerate geochemical reactions, such as oxygen consumption at repository depth, reactions of engineering and stray materials that were introduced during the construction of the repository and microbial activity.

The glacial phase is relatively uneventful except for the glacial melting periods (two are expected in the next glaciation, according to the Weichselian-R scenario). The glacial melting periods may generate higher groundwater flow because of the pressure differential between the ice-covered and ice-free areas. Up-coning of deep saline ground water and seawater intrusion are also possible due to these pressure differentials. Furthermore, intrusion of diluted glacial meltwater to repository depth cannot be ruled out. Glacial meltwater, because of its low ionic strength, could cause chemical erosion of the buffer and the backfill and enhance canister corrosion, particularly if oxygen is dissolved in the meltwater. The probability, extent and consequences of glacial meltwater intrusion are currently being evaluated in the framework of KBS-3V work.

Post-glacial earthquakes are also possible following the retreat of an ice sheet. The expected number of canisters that could potentially be damaged by rock shear in the event of a large earthquake is 16 out of 3,000, as calculated in KBS-3H process report /Gribi et al. 2007/ and evolution report /Smith et al. 2007a/. This is a preliminary result since there are significant uncertainties that could lead to either an underestimate or an overestimate of the actual likelihood of canister damage. The fracture network data used for this estimate is based on work by /La Pointe and Hermanson 2002 and Poteri 2001/; these data are likely to be revised in the course of future studies. One of the key assumptions used in the estimate of canisters damaged by a rock shear movement is that fractures with even moderate amounts of cohesion and friction will not slip as a result of earthquakes, as demonstrated by means of a sensitivity study by /La Pointe et al. 2000/. Thus, La Pointe and Hermanson adjusted the fracture intensity (P32) of the discrete fracture network model based on the assumption that only open fractures have the potential to slip. Cohesion and friction between the surfaces of fractures classified as filled or tight was assumed to prevent these fractures from slipping. Cohesive fractures (sealed or partly open) have also been excluded in the data used in the KBS-3H safety studies. In contrast, for SR-Can, a fracture model was used that included both sealed and open fractures when computing intersection probabilities. If the assumption by /La Pointe et al. 2000/ were not to hold, the number of canisters that could be damaged by rock shear would have been underestimated for the KBS-3H case. Other key assumptions and sources of uncertainties, e.g. related to the application of Hedin's /Hedin 2005/ model to estimate the probability of canister/fracture intersection in a KBS-3H repository, are described in the Section 7.4.5 of the Evolution report /Smith et al. 2007a/.

Throughout the ensuing interglacial, the Olkiluoto site will be submerged. No natural gradient or topographical feature giving rise to elevated groundwater flows rates is present. The Olkiluoto site is expected to emerge from the water in about 120,000 years, according to the Weichselian-R scenario, and begin a new glacial cycle.

In the Emissions-M climate scenario, surface temperatures will continue to rise during the post-closure phase, peaking at about 20,000 years AP, and effectively leading to snowless and frostless winters. By 50,000 years AP, the land will have risen to about 70 m higher than today's level. A permafrost stage occurs at 170,000 years AP but no ice sheet is formed before 350,000 years AP. It is assumed that the climate evolution in the Emissions-M scenario will be similar to the Weichselian-R from the development of the ice sheet onwards. The main differences compared to the Weichselian-R scenario are the timing of climatic changes (permafrost, ice sheet, glacial melting) and the effects of the prolonged land uplift and prolonged infiltration of surface water. Rock temperatures at repository depth are also higher for the Emissions-M scenario after the occurrence of the first permafrost period in Weichselian R-scenario (around 15,000 years). For example, at 20,000 years AP and at –420 m, the rock temperature is estimated to be roughly 10 °C higher that the current ambient rock temperature /Pastina and Hellä 2006, Smith et al. 2007a/.

The KBS-3H and KBS-3V Evolution Reports for Olkiluoto describe mainly the evolution of the site up to the end of the next glacial cycle (125,000 years AP for the Weichselian-R scenario and 450,000 years AP for the Emissions-M scenario). Although it is not explicitly required by the Finnish regulations, the site evolution will be followed up to one million years in future updates of the Evolution report because this period reflects the time within which the peak doses will be released (see Chapter 5 of the Evolution Report; /Smith et al. 2007a/). Within one million years, eight more glacial cycles are expected according to the Weichselian-R scenario and about two according to the Emissions-M scenario. Further studies will investigate the effect of the additional glacial cycles expected with the next million years on the barrier system.

4.2 Observations from Onkalo monitoring and their implications

The local hydrogeological and hydrogeochemical conditions described above are the baseline conditions and have already been disturbed by the construction of the underground characterisation facility, Onkalo. These conditions will also be further perturbed by the construction of the repository, as described below. In anticipation of such perturbations, a monitoring programme is being carried out during the construction of Onkalo in order to determine the perturbations caused and enhance site understanding.

4.2.1 Assessment of disturbances caused by construction and operation

In advance of actual observations from the Onkalo excavations, an assessment of potential disturbances caused by the presence of the Onkalo tunnels was undertaken by /Vieno et al. 2003/. Further assessments incorporating the more recent site data and hydrogeological models, as well as observations made during the on-going excavations, have been carried out in /Löfman and Mészáros 2005, Ahokas et al. 2006, Pastina and Hellä 2006 and Alexander and Neall 2007/ in addition to the Prediction-Outcome studies, an integral part of the Site Description (see Chapter 9 of /Andersson et al. 2007/). The R20 programme launched by Posiva is working on the strategy to handle groundwater inflow from the major hydrogeological zone HZ20 onwards.

4.2.2 Onkalo monitoring programme

In the construction phase of Onkalo, the monitoring activities aim primarily to /Posiva 2003b/:

- Observe possible changes in the repository host rock that could be of importance for the long-term performance, and its assessment, of the repository.
- Obtain data that can help in understanding the features and processes in the repository host rock and the surface environment.
- Obtain information on the response of the host rock to the construction activities that can be used in the further planning of the construction and operational activities, as well as for planning the final closure of the facility.
- Collect observations that can be compared with the predicted environmental impact of the facility.

In planning the monitoring programme, a study was conducted to identify all the potential perturbations to the repository near field caused by the excavation and operation of Onkalo /Miller et al. 2002/. The potential perturbations were assessed qualitatively to determine those which could have a significant impact on the long-term performance of the repository and/or influence the understanding of the site. The most important potential perturbations identified were then used to focus the monitoring programme. In practise, this means that monitoring activities have been carried out in the areas of:

- Rock mechanics.
- Hydrology and hydrogeology.
- Geochemistry.
- Foreign materials.

These are described briefly in the following sections. In addition, there is a programme of surface environmental monitoring which will not be described further here /see Posiva 2003b/.

It is important to emphasise that the long-term safety of the repository at Olkiluoto will not depend on the existence, or the results, of any monitoring programme. Also, the rates of many processes within the repository that are important in long-term safety are so slow that it would not be possible to detect any sensible changes within any likely institutional control period /Posiva 2003b/.

The current design for Onkalo is shown in Figure 4-4. Onkalo will consist of a system of exploratory tunnels accessed by a main tunnel and, according to the current design, by two or three shafts /Posiva 2003c/. The main characterisation level will be located at the planned repository depth (–420 m) with a lower characterisation level at –520 m. During the operational phase of the repository, Onkalo will serve as an access route to the repository and the first disposal tunnels are planned to be adjacent to the main characterisation level. Demonstrations and testing activities will be carried out predominantly at the main characterisation level.

The total underground volume of Onkalo will be approximately 365,000 m^3 and the combined length of tunnels and shafts approximately 8,500 m /Posiva 2003c/. Construction of Onkalo started in 2004 and the construction and installation period is planned to last until 2014. Investigations in Onkalo have already started and they will be carried out throughout the construction phase and possibly beyond 2014.

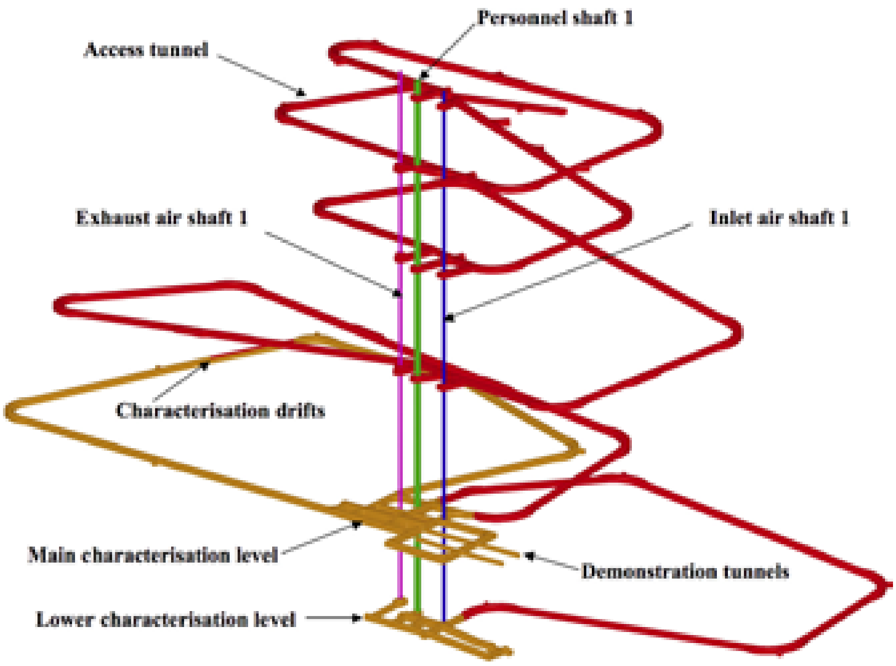

***Figure 4-4.** General layout of Onkalo according to the design based on the situation at Autumn 2007.*

4.2.3 Rock mechanics

As in all significant rock engineering projects, Onkalo includes a rock monitoring programme to ensure safety during construction and operation and to check the validity of the assumptions, models and rock mass properties used in design analyses. Furthermore, in the case of a repository, an important barrier to radionuclide transport is the bedrock itself; its quality has to be confirmed for disposal purposes and one of means of doing this is by monitoring during the construction and operational phases. In addition, the rock mechanics monitoring programme allows the evaluation of the effect of repository construction on the mechanical stability of the rock mass as well as the use of the results in safety analyses /Posiva 2003b/.

Onkalo represents a section of the future repository and it was therefore important to launch the rock mechanics monitoring programme as early as possible. Monitoring from the early phases of construction will be more helpful in terms of design, whereas continuous monitoring using the same instruments will allow the longer-term stability of the rock mass to be investigated.

Monitoring activities have been, and will continue to be, carried out in the following areas:

- Mechanical impact of construction.
- Microseismicity.
- Tectonic deformation and isostatic uplift.
- Rock movements.
- Stress changes.
- Rock damage.
- Loads in rock support structures.

Temperature measurements will also be added for interpretation of mechanical data.

4.2.4 Hydrology and hydrogeology

Large underground excavations act as a "sink" drawing water from all directions. This affects the hydrogeology and the hydrogeochemistry within a radius of a few kilometres from the site. As a result of this "sink" effect and inflow to the tunnels, the groundwater table is drawn down, surface/meteoric water may reach greater depths and saline water from deeper zones of the bedrock may be brought up toward the excavated areas (this effect is called "up-coning"). Inflow into the open spaces takes place mainly through water-conducting fracture zones but also from sparsely fractured rock (although in the latter case, at a much lower rate).

The potential hydrological and hydrogeochemical disturbances caused by the open tunnel system of Onkalo (and/or of the repository) are subject to considerable uncertainty. These disturbances were assessed by /Vieno et al. 2003, Löfman and Mészáros 2005, Ahokas et al. 2006 and Alexander and Neall 2007/. In the Site Description 2006 (see Chapter 9 of /Andersson et al. 2007/), these effects were re-evaluated using the most up-to-date hydrogeological model, as well as knowledge of baseline conditions, data from monitoring during the excavation of the Onkalo access tunnel (which is still on-going) and numerical flow modelling. These flow modelling results indicate that Onkalo and repository construction will have a pronounced effect on the flow and salinity at repository depth.

Groundwater monitoring will be performed during the construction of Onkalo (see summary of ongoing and planned activities in /Ahokas et al. 2006/). The monitoring programme for hydrology and hydrogeology is extensive, covering a wide range of activities at the surface, in boreholes and underground in the Onkalo excavations. Results from baseline studies carried out over the last 13 years have provided ample evidence of the range of natural fluctuations, for example, caused by dry and wet periods and as a result of changes in sea level. The considerable impacts anticipated due to the construction of Onkalo should therefore be easy to detect and separate from these natural perturbations. Small and unexpected changes further from Onkalo may be important for characterisation purposes, thus accurate time-series analyses are needed to determine if such changes are due to natural fluctuation or to the impact of Onkalo itself.

The drawdown of the groundwater table and the inflows into Onkalo will change the salinity in the bedrock. The extent of any such general up-coning and the magnitude of the changes in salinity in individual fractures or fracture zones will be determined by the requirement for pumping from Onkalo and are difficult to predict. The modelled changes in salinity /Löfman 2005/ are used as a basis for estimating the magnitude of the possible changes.

Saline water has good electrical conductivity and the changes in the saline water interface can be measured by electrical or electromagnetic surveys, either from boreholes or from the ground surface. Geophysical monitoring is based on the fact that only the salinity of the water increases, thereby increasing its conductivity, whilst the other electrical properties of the bedrock remain unchanged.

Monitoring of the rate and location of groundwater ingress and the rate of removal of water due to pumping and other activities is required for the analysis and understanding of the monitoring results described above. The rate of water abstraction is determined by the water balance, where the amount of water removed by pumping and by the ventilation system is monitored, taking into account the use of water for construction, washing of walls etc On the basis of the experience from other underground facilities, such as Äspö, the total ingress of groundwater is likely to decrease gradually following the period of construction and locally (especially near the surface) during the construction phase.

Hydrology and hydrogeological activities include monitoring of /Posiva 2006/:

- Meteorological conditions.
- Infiltration and surface run-off.
- Evolution of the groundwater table and hydraulic head.
- Evolution of the hydraulic network.
- Temperature.
- Groundwater flow rates and directions.
- Evolution of the saline water interface.
- Inflow and water balance in Onkalo.

4.2.5 Geochemistry

Monitoring the geochemical stability and changes in response to the excavation of Onkalo is an important part of the continuous monitoring programme. The principal characteristics to be monitored are the compositions of the surface waters and groundwaters, both in the immediate vicinity of the excavations and in the surrounding areas at Olkiluoto. Changes that may occur in minerals, mainly those found in fractures and fault zones near to the excavations, are less likely to be observed because of the relatively slow rates of reaction of these minerals.

The monitoring of groundwater composition and changes due to the inflow of groundwater into the Onkalo, the drawdown of the groundwater table and its effect on the salinity distribution caused by the excavation of the Onkalo is partly an extension of the site characterisation programme which determined the baseline conditions. Further information will be obtained during the monitoring programme and this, in turn, will provide more information on baseline conditions and on the characteristics of the site /Posiva 2003b, Andersson et al. 2007, Klockars et al. 2007/.

Monitoring focuses on following the eventual intrusion of oxygen-rich water and on the migration of the redox front as well as on the changes in the interfaces between water types.

Monitoring from the surface

The surface-based geochemical monitoring programme consists of:

- The analyses of water samples from selected permanently packed-off sections in a number of deep boreholes. A suite of parameters (such as major ions, electrical conductivity (EC), redox, pH, isotopes of H and O and some redox species and dissolved gases) will be evaluated periodically, while a comprehensive characterisation will only be carried out about once every year.
- Monitoring electrical conductivity at a number of locations to trace the migration of saline or dilute groundwater (a key factor for assessing the performance of the geological system for disposal). Changes in salinity indicate changes in the water composition and can be used to trigger a more comprehensive sampling programme. Depending on the properties of the conductive structures, these measurements may provide indications of either local, possibly rapid, changes in salinity or slower processes, such as up-coning of saline waters or drawdown of superficial meteoric or seawater, on a broader scale.
- Sampling shallow groundwater from overburden observation tubes and the bedrock to track the potential evolution of near-surface groundwaters, changes in recharge conditions, potential changes in deep groundwater discharge and to provide input to the monitoring of the migration of the redox front (see below).
- Monitoring the migration of the redox front and change in pH by analysing the content of dissolved oxygen, dissolved CO_2, dissolved inorganic carbon (DIC), pH and Eh, and redox parameters (Fe^{2+}/Fe_{total}, U, S_{total}, SO_4, S^{2-}). Additional data, such as δS-34 of sulphur species, may be determined once a year from selected sampling points. Organic matter and microbial activity may also be determined, as part of redox monitoring, and mineral sampling (e.g. calcite, pyrite) may take place at locations where changes are predicted from modelling.
- Monitoring drainage from excavated rock spoil heaps and the possible recharge of this water to the groundwater flow system. Specific parameters determined are Cl, NO_3, NO_2 and dissolved organic carbon (DOC), which would track contamination by rock salts and explosive residues that readily leach from the rock spoil.

Monitoring in Onkalo

During construction the geochemical monitoring in Onkalo will consist of:

- Sampling and analysis of the groundwater flowing from leaking structures and fractures into the tunnel at a selection of inflow points in the tunnel and, in conjunction with hydraulic tests, in boreholes drilled from the tunnel.
- Sampling and analysis of groundwater accessed by boreholes from Onkalo.
- Measurement of the gas content in groundwater samples taken in the boreholes drilled from the tunnels.
- Monitoring the migration of the redox front and the change in pH from selected boreholes in the tunnel, as described above.
- Monitoring the quantity and composition of water flowing into the tunnel before it is pumped out to sea.
- Detailed analyses of parameters that are related to construction and stray materials introduced into the facility, such as analyses of explosive residues, nitrogen compounds, microbes, (hyper)alkaline-plumes etc.
- Analyses of microbes and biofilms on the surfaces of rock and fracture infillings as well as in the groundwater, e.g. sulphate reducing bacteria, iron reducing bacteria, methanogens, and monitoring the end-products of the microbial processes, such as organics, mineral-coatings, iron compounds etc.

4.2.6 Foreign materials

Foreign materials are those materials introduced into Onkalo and the future repository during construction and operations, either for use in the excavation and construction (also termed engineering materials) or as a by-product of activities (also called stray materials, e.g. rubber from vehicle tyre wear or impurities in ventilation air), but which are not part of the engineered barrier system. This means, for example, that impurities in bentonite, although unavoidable, are classed as foreign materials whereas the bentonite itself is part of the EBS.

The use of foreign material is restricted and regulated in the Onkalo and a list of appropriate materials that can be used in Onkalo has been prepared /Vuorio 2006/. The constructor must keep a record of potentially detrimental materials introduced and retrieved from Onkalo. The unauthorised introduction of materials into the Onkalo is prevented by means of quality control and security arrangements. The amount of foreign materials, such as cement, concrete, labelled water etc will also be monitored by random inspections and the results compared with the recorded amounts.

In order to limit the input of organic matter to Onkalo and for reasons of hygiene, lavatories will be provided underground and the potential leakage from such installations will be monitored.

4.2.7 Use of the results from Onkalo activities

The most extensive use of the information collected from the monitoring system is for the further characterisation and understanding of the Olkiluoto site /Posiva 2003b/. New information can lead to changes in existing geological, hydrogeological, geochemical or rock-mechanical models of the site and, should these be sufficiently important, changes in design or construction methods may also be considered. To facilitate this utilisation, the data will be regularly assessed and compared with parameters derived from existing models. In this process of assessment, attention will be paid to natural fluctuations in the ambient conditions as well as to measurement uncertainties. To make comparison easier, baseline characterisation data are used to set bounding values for the natural ranges of variation of the most important parameters. These boundary values will be considered as “action levels” and values outside these ranges will trigger a more thorough analysis of the monitoring data, which may result in modifications to the existing models and designs.

“Specific action procedures” are defined /Vieno et al. 2003/ and developed for cases where observations are made that could have direct relevance to operational or long-term safety.

With regard to operational safety, such observations may be related to, for example:

- Rock instability.
- High inflows of groundwater.
- Hazardous gases.

As to the preservation of the properties of the rock mass and the associated hydrogeological system that are needed to assure long-term safety, observations that may warrant specific action include:

- Data that indicate fast pathways between the intended area for repository development and rock closer to the surface.
- Data that indicate anomalous chemical characteristics of the groundwater.
- Data that indicate efficient hydraulic connections between the excavated areas and deeper high-salinity groundwater domains.
- Data that indicate anomalous stress-strength conditions in the bedrock.

The basis for assessing the significance of the observations is discussed in /Vieno et al. 2003/. The most important aspects from the point of view of long-term safety relate to inflows of groundwater to the tunnels and foreign materials in the repository area.

Over the 10 years or so of monitoring activities at Onkalo, the main objective is to supply data to improve and increase confidence in the site description and associated models that will support the safety case for the license application for the Olkiluoto repository, which will be made in 2012.

Given the current schedule for Onkalo construction, the first data from observation niches built in Onkalo at repository depth will be available at the earliest in 2009–2010. According to Posiva's license application schedule, these data cannot be incorporated into the interim report on licensing preparedness, due in 2009 /Posiva 2006/. A wealth of information from Onkalo will be available, however, at the end of 2010 or early 2011 and in the following years. These data will be evaluated and presented in the final Site Report prepared in support of the Complementary Evaluations report for the KBS-3V design alternative (due in 2011), itself in support of the 2012 license application. This Site Report will also be relevant to the KBS-3H safety assessment as it mainly concerns site properties and not design features.

Onkalo monitoring will not end when the repository goes ahead (if the construction license is granted), indeed, Onkalo may provide many important data at a very late stage, possibly even after the beginning of repository construction. It is likely that there will still be open questions and possibly significant uncertainties with respect to some aspects of the site that only further data from repository depth can resolve. These late data will feed into the next generation of models, along with information provided during repository construction, in support of future safety assessments required at key programme milestones.

4.3 Complementary lines of evidence on site suitability

4.3.1 Geological stability

Uplift and erosion

An important requirement for the siting of a deep repository is the long-term geological stability with low rates of uplift and erosion. Even if there is no significant tectonic uplift going on at the present day, the understanding of the site and its evolution must show that this is also the expected situation over the next million years or more. Such a demonstration is based on an understanding of the tectonic environment of the site, its geological history and future development.

Tectonic uplift of the crust takes place in areas of active tectonic plate movements and may be associated with mountain building, as in the Alps or Japan. In such environments, the uplift may continue for millions of years and, combined with erosion, result in deeply buried rocks being eventually exposed at the surface. The very long periods of time over which the repository-host rock system must retain and retard the radionuclides from the spent fuel means that in regions affected by tectonic uplift, the uplift rate becomes relevant to the long-term safety of the repository.

Olkiluoto, sited on the 1,800–1,900 million years old, thick, shield area of southern Finland, is not affected by tectonic uplift because it is remote from active plate margins and current mountain building; present-day tectonic influence takes the form of mild regional compression arising from the mid-Atlantic Ridge "push" /Lambeck and Purcell 2003/. After an "exciting" early history, which gave rise to the metamorphic and igneous rocks described earlier, southern Finland underwent a period of erosion and peneplanation of the Proterozoic bedrock. This was followed by a long quiescent period with some Late Vendian to early Palaeozoic (i.e. before about 500 My) sediment deposition in the shallow sea that covered large parts of the Fennoscandian shield at the time. Subsequent erosion of these sediments by the late Phanerozoic (before 1.64 My) gave rise to the present-day topography /Paulamäki et al. 2002/. Thus there is evidence of regional stability over millions of years with no suggestion that this situation will be disrupted by changes in plate tectonics in the next few million years.

Tectonic uplift is different from sea-level change due to glacial rebound /Mäkiaho 2005/ which is limited by the readjustment necessary before the crust comes back into isostatic equilibrium after the loss of the extra weight of ice – in total, a matter of some tens of metres of relative uplift of the land is envisaged over the next few thousand years /Ruosteenoja 2003/.

It is also interesting to consider the potential rates of erosion for the Olkiluoto area. In making an assessment of likely weathering rates for southern Sweden in the SR-Can study, SKB came to average figures of 1–10 metres per million years for the low relief terrain and cool temperate climate /SKB 2006d, Påsse 2004/. This was based on studies of long-term erosion in Sweden /Lidmar-Bergström et al. 1997/ as well as the effects of erosion due to glaciation /Lidmar-Bergström 1996/. These erosion rates also apply to Finland. Such low erosion rates, especially in combination with evidence for long-term stability, ensure the continued isolation of the repository over the required period of time.

Geomechanical stability

Finland is located in a geomechanically stable area. The density and magnitude of earthquakes in Finland is lower than in other sites in Northern Europe (see Section 4.1.1). This is confirmed also by the very low seismicity measured at the Olkiluoto site.

Hydrogeochemical stability

The layering of the different groundwater types and their mixed interfaces seems to recover quite fast from (or resist being affected by) disturbances related to glaciation, such as infiltration of meltwater (see Section 4.1.3), suggesting considerable hydrogeochemical stability. Palaeohydrogeological arguments provide convincing support for expectations concerning long-term flow system evolution, as the current groundwater chemistry is the result of flow, transport and water rock interactions driven by past and current climate.

4.3.2 Absence of exploitable natural resources

The existence of mineral resources, such as metal ore deposits, or other exploitable resources, such as geothermal energy, in a repository siting area means that there is an increased likelihood of inadvertent intrusion into the repository during, for example, exploratory drilling. This could be a problem in the future, particularly if records of the repository location have been lost. Consequently, to avoid such accidental intrusion, spent fuel disposal sites cannot be located where there are exploitable resources.

The geological environment at the Olkiluoto site has no economic potential for hydrocarbons (although there is a notable content of methane and some higher hydrocarbons in the groundwaters at depth, it is not significant enough to constitute a commercial resource) or coal, the low geothermal gradient makes geothermal energy exploitation unlikely[16] and there is no evidence for metalliferous or other industrial mineral deposits locally that might be considered commercially viable in future. Thus the site lacks the natural resources that might attract exploratory drilling and inadvertent human intrusion in the future.

[16] The increasing use of ground sourced heat pumps for domestic and small building heating applications suggests that drilling of boreholes could occur even though there is no potential for larger-scale commercial geothermal plants. However, it should be noted that these small-scale heat pumps exploit only the upper tens of metres of the geology rather than boreholes several hundreds of metres deep.

The presence of substantial amounts of spent fuel and high-quality copper (canisters) could, however, also be seen as a valuable future resource and attract deliberate human intrusion. In this case, knowledge of the presence of copper and spent fuel in the repository implies that the intruders will also be aware of the dangers, as well as the difficulties, inherent in its exploitation. Future exploitation, or deliberate intrusion for other reasons, cannot be prevented, indeed it may be necessary to future society, but in this case, the responsibility for the potential consequences must lie with those carrying out the task /Grimwood and Thegerström 1990, NEA 1995/.

4.3.3 Comparison of Olkiluoto geosphere characteristics with those of other granitic repository sites

The Nuclear Energy Agency lists the following favourable characteristics for a generic repository environment /NEA 2004a/. The environment should be:

- Unlikely to be affected by major tectonic movements, volcanic events or other geological phenomena that could give rise to rapid or sudden changes in geological or geochemical conditions.
- Largely decoupled from events and processes occurring near the surface, including the effects of climate change.
- Lacking in natural resources that might attract exploratory drilling thus minimising the possibility of inadvertent human intrusion in the future when the location of the repository may no longer be known.

Other repository programmes consider or have considered crystalline bedrock as host rock for a geological repository (see Table 3-1): Sweden, Canada, France (Dossier 2005 Granite), Japan (H-3 and H-12), Switzerland (Kristallin-I), and Spain. The general favourable attributes of a granitic host rock cited by these programmes /e.g. Andra 2005a/ are typically:

- Long-term geological stability.
- Low erosion rates.
- Low permeability of the host rock matrix.
- Good thermal conductivity.
- Good mechanical properties of granite (favouring repository construction).

The existence of fractures, which are potential routes for water circulation, constitutes one of the main issues for repository design in a crystalline medium. The design solution typically adopted is to distribute the repository modules in rock blocks that are only slightly or not fractured and observe respect distances from the major water-conducting faults. This solution is also adopted in the Finnish and Swedish repository designs.

The hydrogeochemical environment can be very different among granitic or crystalline bedrock sites. The hydrogeochemical conditions and their evolution greatly affect the performance of the engineered barrier system. In particular, it is important to show the absence or the presence of only a limited amount, of agents detrimental to the engineered barrier system performance (e.g. sulphides, chlorides, oxygen) and to understand their evolution with time. This is also the approach undertaken in the KBS-3H safety studies.

The arguments used in other safety assessments on the suitability of their sites show that Olkiluoto fulfils the general criteria for site suitability recognised internationally. This comparison can be considered a complementary argument for the suitability of the Olkiluoto site from the point of view of long-term safety.

5 Safety assessment – support for approach and key assumptions

The aim of this chapter is to provide support for the KBS-3H safety assessment by comparing what is done in this assessment with what has been done elsewhere, specifically in the most relevant KBS-3V safety assessments, TILA-99 and SR-Can[17]. The purpose is not to compare every calculational case, model or dataset exhaustively but to illustrate how the KBS-3H safety assessment builds on the preceding work and to demonstrate that assumptions and uncertainties, some of which will have different significance for, or potential impact on, KBS-3H are treated in a rigorous a way, despite less experience with this concept.

5.1 Scope of the safety assessment of a KBS-3H repository at Olkiluoto

In the KBS-3H, as in the KBS-3V, the Base Scenario assumes (as required by Finnish regulations) that the performance targets defined for each barrier are met. This is interpreted as meaning that each barrier fulfils the safety functions assigned to it in the safety concept for a period extending to a million years or more. In this case, no canister failure occurs before one million years. There are, however, uncertainties in the evolution of a repository whereby one or more canister failures lead to radionuclide release and transport and exposure of humans and other biota to released radionuclides, in a one million year time frame. A wide range of cases addressing various uncertainties in the evolution of a KBS-3H repository is defined and analysed in radionuclide release and transport calculations in the KBS-3H assessment (see Section 5.3.1 and the summary in Table 5-1). The emphasis is on uncertainties relating to the evolution of the near-field conditions due to the KBS-3H-specific components, such as the effect of the supercontainer on the transport barrier provided by the buffer. Some uncertainties are, however, considered not to fall within the main focus of the assessment. In particular, uncertainties that are not considered relevant in discriminating between the performance of KBS-3V and KBS-3H repositories are either not addressed or are analysed in less detail than others. These include uncertainties in the transport barrier provided by the geosphere, biosphere uncertainties and uncertainties related to future human actions. Thus, for example, variability and associated uncertainty in the geosphere transport barrier as a function of time in response to climate change, isostatic rebound and changes in groundwater composition are not generally addressed. The impact of different assumptions regarding groundwater salinity on the transport barrier functions of both the buffer and the geosphere is, however, considered. The possibility that an influx of glacial meltwater could give rise to chemical erosion of the buffer, thus increased fluxes of sulphide to the canister surface and early canister failure due to copper corrosion, is also addressed. In this case, it is assumed that the barrier function of the buffer is degraded with respect to both the transport of sulphide to the canister surface and the transport of released radionuclides. Overall, however, the range of cases considered is more limited than that considered, for example, in TILA-99 or SR-Can (see Sections 5.2, 5.3.2 and 5.3.3, below).

[17] An updated safety assessment for a KBS-3V repository at Olkiluoto is being produced for Posiva at the time of writing.

5.2 Relevant safety cases for the disposal of spent fuel

The main content of this section is a comparison between the KBS-3H assessment at Olkiluoto, the earlier TILA-99 assessment /Vieno and Nordman 1999/ and SR-Can /SKB 2006a/. However, as noted above, it is important to understand that the scope and objectives of the three assessments are somewhat different, which is reflected particularly in the assessment cases treated.

5.2.1 Scope and objectives of TILA-99

Detailed site investigations were carried out at four sites in Finland (Hästholmen, Kivetty, Olkiluoto, Romuvaara) with the aim of selecting one of them as the site for the spent fuel repository by the end of 2000. TILA-99 is the post-closure safety assessment for a spent fuel repository at the four candidate sites submitted in support of the application for the Decision in Principle.

TILA-99 is a continuation and update of the previous TVO-92 /Vieno et al. 1992/ and TILA-96 /Vieno and Nordman 1996/ assessments and focuses on the normal evolution of the repository at the candidate sites and on the potential release and transport of radionuclides from the repository into the geosphere and biosphere. TILA-99 deals with a KBS-3 reference disposal method of spent fuel assemblies emplaced in copper-iron canisters in a KBS-3V-type repository excavated at a depth of about 500 metres in crystalline bedrock (Note: the version of KBS-3 used in TILA-99 is here referred to as the TILA-99 concept). Alternative canister and repository designs were assessed by /Autio et al. 1996/ and alternative spent fuel management options and disposal concepts are discussed in the EIA report[18] /Posiva 1999/.

The aims of TILA-99 were to provide a robust and transparent safety assessment of the candidate sites in which data and assumptions were fully traceable and the results reproducible from the information and data presented in the report. With the exception of some realistic scenarios, conservative assumptions, models and data, and deterministic modelling were used throughout the analysis. The models employed in the release and transport analyses of radionuclides were relatively simple. Also, a single exposure pathway, from drinking contaminated well water, was used in place of more complex, multi-pathway biospheres for the different sites.

5.2.2 Scope and objectives of SR-Can

The SR-Can project is a preparatory stage for the SR-Site assessment, the report which will be used in support of SKB's application for a final repository at either the Forsmark or the Laxemar site. The purposes of the SR-Can safety assessment are:

1. To make a first assessment of the safety of potential KBS-3 repositories (i.e. KBS-3V type) at Forsmark and Laxemar to dispose of spent fuel.
2. To provide feedback to design development, to SKB's R&D programme, to further site investigations and to future safety assessment projects.
3. To foster a dialogue with the authorities that oversee SKB's activities, i.e. the Swedish Nuclear Power Inspectorate (SKI) and the Swedish Radiation Protection Authority (SSI), regarding interpretation of applicable regulations, as a preparation for the SR-Site project.

The objective of the SR-Can report is to investigate whether the KBS-3 disposal method has the potential to fulfil regulatory safety criteria, given the host rock conditions at the sites insofar as they are known after the initial site investigation phase. The intention of the SR-Can report is not to fully establish the suitability of the studied sites – this will be done in SR-Site. The intention is also not to finally establish the technical system for disposal – but rather to investigate the safety of the system as it is specified at this stage, and to give feedback for further developments to that specification.

[18] Operational safety, nonradiological environmental impacts as well as social and financial impacts was also evaluated in the separate Environmental Impact Assessment (EIA) report /Posiva 1999/.

The Swedish Radiation Protection Institute's Regulations concerning the Protection of Human Health and the Environment in connection with the Final Management of Spent Nuclear Fuel or Nuclear Waste (SSI FS 1998:1) includes the requirement that protection of human health shall be demonstrated by compliance with a risk criterion that "*the annual risk of harmful effects after closure does not exceed* 10^{-6} *for a representative individual in the group exposed to the greatest risk*" /SSI 1998/. Harmful effects refer to cancer and hereditary effects. As a result of this risk criterion, the SR-Can assessment is required to make greater use of probabilistic methods than is the case in Finland with either TILA-99 and or the KBS-3H safety studies.

Although most of the calculations in SR-Can are deterministic, SR-Can also uses probabilistic calculations as a means of handling data uncertainty and spatial variability in modelling radionuclide transport and dose for a wide range of calculation cases. The use of probabilistic evaluations of calculation cases followed by sensitivity analysis results is the approach used in SR-Can to maintain realism in calculation cases while capturing the uncertainty. Conditional risks are calculated for each scenario and variants and these are then weighed together using the probability for each scenario/variant. Furthermore, each variant, represented by a specific calculation case, is to be evaluated probabilistically to determine the mean exposure given the data uncertainties for the particular variant. The approach of calculating risk as a weighted sum over a number of scenarios constrains the way in which scenarios are selected and defined.

The aim of providing feedback to the design specification for KBS-3V has meant a great emphasis on understanding and treating as quantitatively as possible all the processes which could occur and affect the repository evolution and thus eventual performance. This represents a move away from the use of very pessimistic parameter values to (hopefully) bound the effect of detrimental processes, which has been the approach taken in earlier and more preliminary performance assessments in many national programmes, but reflects the increasing experience of SKB with the KBS-3V repository system, greatly improving databases and the development of assessment methodology.

5.3 Assessment cases

5.3.1 Assessment cases for KBS-3H

An overview of features and processes with different significance for, or potential impact on, KBS-3H compared with KBS-3V is given in Table 5-1. Summary descriptions of their relevance to radionuclide release and transport are also given in the table, along with an indication of major uncertainties, the evaluation of impact on canister failure mode and timing and on radionuclide transport. Table 5-2 provides an overview of assessment cases calculated in the Radionuclide Transport Report /Smith et al. 2007b/.

The cases to be analysed are divided into a 2-level hierarchy of groups. At the first level of the hierarchy are groups of cases addressing the following potential canister failure modes:

- An initial penetrating defect.
- Failure due to copper corrosion.
- Rupture due to rock shear.

At the second level of the hierarchy are sub-groups of cases or individual cases addressing particular areas of uncertainty, such as uncertainties in the evolution of the spent fuel or in the evolution of the buffer. For each of the potential canister failure modes, a base case is defined and a number of variant cases of the second level type. Perturbations to radionuclide release and transport caused, for example, by the steel and cementitious components of the KBS-3H repository external to the canisters are assumed to be negligible in the base cases (even though this may be non-conservative) but are considered in variant cases. The majority of calculation cases addressing uncertainties not specifically related to a canister failure mode are assigned to the group dealing with an initial penetrating defect.

Table 5-1. Features and processes with different significance for, or potential impact on, KBS-3H compared to KBS-3V; summary descriptions of their relevance to radionuclide release and transport are given, along with major uncertainties, evaluation of impact on canister failure mode and timing and on radionuclide transport. PR: Process Report /Gribi et al. 2007/, ER: Evolution Report /Smith et al. 2007a/, RNT: Radionuclide Transport Report /Smith et al. 2007b/.

Feature/process	Relevance to RN release and transport	Major uncertainties	Evaluation of impact Impact on canister failure mode / timing	Impact on radionuclide transport
Piping and erosion during the operational phase and during saturation (cf. PR Section 4.5.2; ER Section 5.5.6)	May locally perturb buffer density and increase rate of diffusion of corrosive agents to canister surface and rate of radionuclide diffusion from failed canister	Likelihood of occurrence; amount of bentonite conveyed by piped water; degree of homogenisation after piping/erosion cease	Scoping calculations in ER App. B.7	Illustration of impact of increased radionuclide diffusion rates in buffer in assessment case PD-HIDIFF
Processes due to the presence of steel components (external to canister) and their corrosion products (cf. PR Section 4.7.1; ER Sections 5.4.2; 5.6.4; 6.5.3)	May result in chemical alteration of buffer and consequent changes to physical properties; may perturb mass transfer at buffer-rock interface	Degree and spatial extent of perturbation	Scoping calculations in ER App. B.7 (impact on capacity of buffer to protect canister in the event of rock shear movements < 10 cm assumed to be negligible)	Illustration of impact of increased radionuclide mass transfer at buffer-rock interface and mixing in outer part of buffer in assessment cases PD-FEBENT1; PD-FEBENT2; PD-FEBENT3
	May provide sorbing surfaces for radionuclides; Fe(II) may compete for sorption sites on buffer	Quantitative understanding of impact; possibility of release of sorbed radionuclides in the event of change in groundwater chemistry	None expected	Impact on sorption not assessed (remaining issue for further study); impact on change in groundwater chemistry on buffer as a whole illustrated in PD-GWMC
H_2 from corrosion of steel components (external to canister) (cf. ER Sections 5.3.1; 5.6.4; 5.7.4)	May participate in microbial reduction of sulphate to sulphide, which may subsequently corrode canister surface	Quantitative understanding of impact	Scoping calculations in ER App. B.7 (minor impact)	None expected
	May perturb groundwater flow and radionuclide transport in the geosphere for the first few thousand years	Quantitative understanding of impact	Minor impact on mass transfer of corrosive agents between geosphere and buffer (not quantitatively evaluated)	Impact on radionuclide transport for an initially defective canister not assessed (remaining issue for further study)
High-pH leachates from cementitious components (cf. ER Section 5.6.5)	May result in chemical alteration of buffer and consequent changes to physical properties; may perturb mass transfer at buffer-rock interface	Degree and spatial extent of perturbation	Scoping calculations in ER App. B.7	Illustration of impact of increased radionuclide mass transfer at buffer-rock interface and mixing in outer part of buffer in assessment cases PD-FEBENT1; PD-FEBENT2; PD-FEBENT3
KBS-3H drift and surrounding EDZ / rock spalling (cf. ER Sections 4.1.2; 5.4.5)	May perturb mass transfer at buffer-rock interface	EDZ hydraulic properties; impact of buffer swelling on rock spalling; transport characteristics of spalled zone	Scoping calculations in ER App. B.7.	Illustration of impact of rock spalling in assessment case PD-SPALL

Table 5-2. Overview of radionuclide release and transport assessment cases considered in the KBS-3H safety studies.

Case	Description
Cases assuming a single canister with an initial penetrating defect (PD-)	
Case	**Description**
PD-BC	Base case for initial penetrating defect in BWR-type canister
PD-VVER	Initial penetrating defect in VVER-440 PWR type canister
PD-EPR	Initial penetrating defect in EPR type canister
PD-HIFDR	Increased fuel dissolution rate
PD-LOFDR	Reduced fuel dissolution rate
PD-IRF	Evaluates transport only of radionuclides present in instant release fraction[a]
PD-BIGHOLE	Increased defect size
PD-HIDELAY	Increased delay until loss of defect transport resistance
PD-LODELAY	Decreased delay until loss of defect transport resistance
PD-BHLD	Increased defect size plus decreased delay until loss of defect transport resistance
PD-HIDIFF	Increased diffusion rate in buffer
PD-FEBENT1	Perturbed buffer-rock interface - high conductivity, narrow perturbed zone
PD-FEBENT2 PD-FEBENT3	Perturbed buffer-rock interface - more extensive perturbed zone (2 different thicknesses)
PD-SPALL	Perturbed buffer-rock interface – high conductivity, narrow perturbed zone, lower flow through intersecting fractures than that assumed in cases PD-FEBENT1 , 2 and 3
PD-EXPELL	Dissolved radionuclides expelled by gas from canister interior and across buffer to geosphere
PD-VOL-1 PD-VOL-2	C-14 transported in volatile form by gas generated by corrosion (2 rates of gas generation)
PD-BCN	Initial penetrating defect in BWR-type canister; Nb present in near field and geosphere in anionic form
PD-BCC	Initial penetrating defect in BWR-type canister; C-14 present in geosphere in anionic form (carbonate)
PD-VVERC	Initial penetrating defect in VVER-440 PWR type canister; C-14 present in geosphere in anionic form (carbonate)
PD-EPRC	Initial penetrating defect in EPR type canister; C-14 present in geosphere in anionic form (carbonate)
PD-NFSLV	Near-field solubilities varied according to uncertainties in redox conditions
PD-SAL	Brackish / saline water present at repository depth (all time)
PD-HISAL	Saline water present at repository depth (all time)
PD-GMW PD-GMWV	Change from reference (dilute / brackish) water to glacial meltwater[b] at 70,000 years (release also starts at 70,000 years – two alternative meltwater compositions)
PD-GMWC	Change from reference (dilute / brackish) water to glacial meltwater[b] at 70,000 years (release starts at 1,000 years, as in the reference case)
PD-HIFLOW	Increased flow at buffer-rock interface
PD-LOGEOR	Reduced geosphere transport resistance
PD-HIGEOR	Increased geosphere transport resistance
PD-HIFLOWR	Increased flow at buffer-rock interface and reduced geosphere transport resistance
Cases assuming a single canister failing due to copper corrosion (CC-)	
Case	**Description**
CC-BC	Base case for failure due to copper corrosion; buffer treated as mixing tank
CC-HIFDR	Increased fuel dissolution rate
CC-LOFDR	Reduced fuel dissolution rate
CC-GMW	Glacial meltwater present at repository depth (impact on near-field solubilities and geosphere retention parameters)
CC-LOGEOR	Reduced geosphere transport resistance
CC-LOGEORG	Reduced geosphere transport resistance, glacial meltwater[b]
CC-LOGEORS	Reduced geosphere transport resistance, saline groundwater[b]

Cases assuming a single canister failing due to rock shear (RS-)	
Case	**Description**
RS-BC	Base case for failure due to rock shear
RS-GMW	Glacial meltwater present at repository depth (impact on near-field solubilities and geosphere retention parameters)
Additional cases (hypothetical pulse release to geosphere) (MD-)	
Case	**Description**
MD-1	Variations in matrix diffusion depth (3 cases)
MD-2	
MD-3	

a Certain radionuclides are enriched at grain boundaries in the fuel, at pellet cracks and in the fuel / sheath gap as a result of thermally driven segregation during irradiation of the fuel in the reactor. These are assumed to enter solution rapidly once water contacts the fuel pellet surfaces, and are termed the instant release fraction (IRF).

b Glacial meltwater is a very dilute ice-melting water. Saline groundwater represents water with a Total Dissolved Solid (TDS) content of about 20 g/l. For detailed composition of the waters used in the assessment, see Appendix D of Radionuclide Transport report.

The evaluation of radionuclide release and transport in the hypothetical case of a canister with an initial penetrating defect is useful in illustrating the impact of a range of uncertainties affecting release and transport processes in the event of canister failure. Furthermore, assuming that the penetrating defect is present at the beginning of operations covers a variety of scenarios for canister failure, regardless of the cause or timing of such event. The assumption of an initial penetrating defect results in the earliest possible radiological impact, although not necessarily the largest impact, for each uncertainty considered.

Using the initial penetrating defect as a reference failure mode, although it is not the main scenario, provides a common basis for comparison with TILA-99 and is also the approach used in SR-Can (see Section 5.3.3).

The possibility that any welding defect will penetrate the copper shell completely or be sufficiently deep to have significant implications for the timing of failure due to copper corrosion is, however, considered to be low (Section 4.1.4 of /Smith et al. 2007a/). The defect size is considered to be the largest (non-penetrative) defect which could be conceivably missed by the inspection technique; a variant case with increased defect size is also considered.

In the base case for the initial penetrating defect failure mode (case PD-BC), the initial penetrating defect is assumed to affect a single canister of BWR fuel from the Olkiluoto 1&2 reactors. The reference spent fuel is assumed to have a burnup of 40 MWd/kgU and an enrichment of 4.2%, which are at the high end of the expected ranges. At the planned closure time of the repository (the year 2100), the average cooling time of the fuel will be well over 30 years, but, for the release and transport analyses, a conservative cooling time of 30 years has been assumed.

Groundwater conditions are assumed to be reducing and dilute/brackish. Of the various reference waters studied, this type is closest in terms of total dissolved solids (TDS) to the expected undisturbed conditions at repository depth in the period up to 10,000 years in the future /Pastina and Hellä 2006/.

In the majority of cases, the failed canister is assumed to contain fuel from the Olkiluoto 1&2 reactors (BWR fuel); Loviisa 1&2 (PWR VVER-440 fuel) and Olkiluoto 3 (EPR fuel) are considered in variant cases specifically addressing the differences between fuel/canister types.

Scoping calculations reported in Appendix B.5 of the Evolution Report /Smith et al. 2007a/ give the expected value of the number of canisters in the repository that could potentially be damaged by rock shear in the event of a large earthquake as 16 out of the total number of 3,000 canisters, although there are some significant uncertainties associated with these values that could lead to them giving either an underestimate or an overestimate of the actual likelihood of damage. The marginal probability of encountering an earthquake of sufficient magnitude to induce damaging movements on these fractures at Olkiluoto in a 100,000 year time frame has been estimated at 0.02 (Table 5-8 in /La Pointe and Hermanson 2002/). The expectation value of the number of canisters damaged by rock shear in a 100,000 year period is thus 16 × 0.02 = 0.32 (or 1 canister in 300,000 years).

5.3.2 Comparison with TILA-99

i) Overview

TILA-99 defines a "base case" in which conditions around the repository are assumed to be roughly similar to those of today and the copper canisters emplaced in the repository are assumed to be initially intact - i.e. they have no initial penetrating defects, or other defects that significantly affect canister lifetime. The evolution of the repository in this case is said in TILA-99 to correspond to a "best-estimate" of the expected behaviour of the system.

Variant assessment cases (or scenarios in the terminology of TILA-99) considered in TILA-99 are categorised as:

- Reference scenarios for the four sites considered in the assessment, in which the consequences of a single, initially defective or later "disappearing" canister are evaluated using site-specific data.
- Cases for sensitivity analysis, in which uncertainties in most of the key features of the assessment model chain are considered.
- Cases for "what if" analyses, which address some specific issues frequently discussed in Finland and abroad, as well as responding to some specific requests presented in STUK's review of TILA-96.

Table 5-3 shows a summary of relevant issues, indicating whether they are dealt with in TILA-99 and in the KBS-3H assessment by defining and analysing assessment cases or by qualitative argument, or whether they are considered as being outside the scope of the assessment.

As noted in Section 5.2, above, some issues addressed in TILA-99 are outside the scope of the KBS-3H assessment or are dealt with less thoroughly in the KBS-3H assessment compared with TILA-99. For example, issues relating to geosphere properties and evolution were of more interest in TILA-99 compared with the KBS-3H assessment because of the objective of comparing the candidate sites in TILA-99. Thus, although indicated as "covered by an assessment case" in both columns in Table 5-3, uncertainties in flow are dealt with only to a limited extent in the KBS-3H safety assessment, even though more site-specific data on Olkiluoto were available compared with TILA-99. In contrast to TILA-99, the focus of the KBS-3H safety studies is making an assessment of the horizontal variant of the KBS-3 disposal method so that a comparison can be made with KBS-3V. Since there are issues that arise from the use of the supercontainer and distance blocks, which have no equivalent in the vertical design, and from emplacement in tunnels rather than individual spent fuel package disposal holes, the KBS-3H assessment addresses some issues that were not included in TILA-99, for example, iron/bentonite interaction at the tunnel wall, which is irrelevant to the TILA-99 concept. However, some issues identified in SR-Can have been included in the KBS-3H assessment, e.g. rock spalling and buffer erosion/canister corrosion due to glacial meltwater, although they were not considered in TILA-99.

Table 5-3. Summary of relevant issues and how they are dealt with in TILA-99 and in the KBS-3H safety assessment.

Issue		TILA-99	KBS-3H
Groundwater properties	Uncertainties in flow	A	A
	Uncertainties in salinity	A	A
Far-field transport	Far-field dispersion	A	O/S
Biosphere	Dose conversion factors	A	O/S
Canister failure	Mode	A	A
	Timing	A	A
Source term	Uncertainties in instant release fraction	A	A
	Uncertainties in fuel dissolution rate	A	A
Solubilities	Uncertainties in values	A	A
Radiolysis	Impact on source term	A	Q
	Impact on near field redox conditions	A	Q
Transport through backfill or along buffer/rock interface	Location of nearest intersecting fracture	A	Q
	Impact of rock spalling	N	A
	Impact of iron/bentonite interaction	O/S	A
Buffer emplacement	Very poor bentonite	A	A
Gas	Displacement of contaminated water	A	A
	Transport of volatile species	Q	A
Glacial meltwater	Buffer erosion/canister corrosion	N	A
	Changes to redox conditions	A	Q
Post-glacial faulting	Physical damage to canister	A	A
	Physical damage to buffer	A	A
	Physical damage to rock	A	A
	Changes to redox conditions	A	Q

A	Covered by an assessment case
Q	Dealt with by qualitative argument
O/S	Defined as outside assessment scope or irrelevant to variant
N	Not addressed

The KBS-3H safety assessment also addresses the transport of volatile species quantitatively whereas this was addressed only by qualitative argument in TILA-99. However, the KBS-3H safety assessment addresses by qualitative argument some issues that were addressed by assessment cases in TILA-99, e.g. influx of oxygenated water, as better understanding of the processes involved means that there is no need for separate assessment cases.

In conclusion, there are no omissions in cases considered to be relevant for the purpose of the KBS-3H assessment when compared to TILA-99.

ii) Comparison with the TILA-99 reference scenarios

Summary of TILA-99 reference scenarios

A total of 22 reference scenarios was evaluated in TILA-99, as described in Chapter 11 of /Vieno and Nordman 1999/: 10 for the case of an initial small ("pinpoint"; 5 mm^2) hole though the copper shell (SH scenarios), a further 10 for the case of no physical containment ("disappearing canister") after 10,000 years (DC scenarios); and 2 for the case of an initial larger ("fingertip"; 1 cm^2) hole though the copper shell (LH scenarios).

The SH, DC and LH scenarios all considered Olkiluoto 1&2 type canisters in a KBS-3V-type repository in a geological environment in which either non-saline or saline groundwater conditions are assumed to persist for the entire time frame covered by the assessment. In addition, four scenarios consider Loviisa 1&2 type canisters in non-saline and saline conditions (Table 5-4).

In each case a range of different flow conditions was assumed around the canisters and in the host rock. These different flow conditions and the resulting transfer coefficients from the buffer to the rock (Q_F) and the transport resistance of the geosphere migration path (WL/Q) are summarised in Table 5-4 (transfers from the top of the deposition hole to the tunnel and from the tunnel to the geosphere were also considered in TILA-99, but are not relevant for the purposes of comparison with the KBS-3H safety assessment).

Table 5-4. The reference scenarios in TILA-99: canister types and failure modes assumed, salinity and flow conditions and the mass transport parameters (for neutral species). Parameter values from Table 11-19 of /Vieno and Nordman 1999/. SH=small hole, DC= disappearing canister, LH=large hole.

Scenario	Flow conditions	Canister type/ failure mode	Q_F [(1)] m^3/y	WL/Q [(2)] y/m
Non-saline groundwater				
SH-ns50	All sites: "median"	Olkiluoto/small hole	2×10^{-4}	5×10^{4}
DC-ns50		Olkiluoto/disappearing canister		
LH-ns50		Olkiluoto/large hole		
SH-ns50Lo		Loviisa/ small hole		
DC-ns50Lo		Loviisa/disappearing canister		
SH-R95=Of95	Romuvaara and future Olkiluoto "95th percentile"	Olkiluoto/small hole	10^{-3}	2×10^{4}
DC-R95=Of 95		Olkiluoto/disappearing canister		
SH-K95=Hf95	Kivetty and future Hästholmen "95th percentile"	Olkiluoto/small hole	2×10^{-3}	2×10^{4}
DC-K95=Hf95		Olkiluoto/disappearing canister		
SH-vhflowns	A "very wet" location	Olkiluoto/small hole	5×10^{-3}	5×10^{3}
DC-vhflowns		Olkiluoto/disappearing canister		
Saline groundwater				
SH-sal50	Present-day Hästholmen and Olkiluoto "median"	Olkiluoto/small hole	2×10^{-4}	5×10^{4}
DC-ns50		Olkiluoto/disappearing canister		
LH-sal50		Olkiluoto/large hole		
SH-sal50Lo		Loviisa/ small hole		
DC-sal50Lo		Loviisa/disappearing canister		
SH-Opd95	Present-day Olkiluoto "95th percentile"	Olkiluoto/small hole	6×10^{-4}	2×10^{4}
DC-Opd95		Olkiluoto/disappearing canister		
SH-Hpd95	Present-day Hästholmen "95th percentile"	Olkiluoto/small hole	1.5×10^{-3}	2×10^{4}
DC-Hpd95		Olkiluoto/disappearing canister		
SH-vhflowsal	A "very wet" location	Olkiluoto/small hole	3×10^{-3}	1×10^{4}
DC-vhflowsal		Olkiluoto/disappearing canister		

[1] Transfer coefficient from the buffer to the rock (see Section 5.4).

[2] Transport resistance of a geosphere migration path: a fracture of width *W* (m), length *L* (m) and flow rate *Q* (m^3/y).

Common features and differences compared to the KBS-3H safety assessment

The 22 reference scenarios in TILA-99 address:

- Site-specific groundwater flow conditions at the four sites considered in this assessment.
- Different fuel types and associated canister designs.
- Different canister failure modes.
- Uncertainty and variability in time in groundwater flow and salinity.

In the KBS-3H safety assessment, all analyses consider the selected Olkiluoto-specific hydrogeological and geochemical conditions. No alternative sites are addressed.

The TILA-99 reference scenarios considered both failed Loviisa canisters and failed Olkiluoto canisters. Thus, in the KBS-3H safety assessment, it is considered unnecessary to repeat this analysis and the focus in the majority of assessment cases is on a canister containing fuel from Olkiluoto, specifically from the Olkiluoto 1 or 2 reactors, with an assumed initial penetrating defect. There is, however, a set of variant cases that address failed canisters containing fuel from Loviisa 1&2 and Olkiluoto 3.

The majority of KBS-3H assessment cases consider a single, initial penetrating defect, with a radius of 0.5 mm, i.e. an area of about 0.75 mm^2, which is significantly smaller than the TILA-99 SH "pinpoint" defect size of 5 mm^2, and very much smaller that the LH "fingertip" 1 cm^2 defect. The 0.5 mm radius defect "*In the view of Posiva's canister experts, …corresponds roughly to the maximum defect size that might escape detection using current non-destructive testing (NDT) quality control techniques*" /Smith et al. 2007b/. While welding defects in the copper shell of the canisters are expected, the possibility that any will penetrate the copper shell completely, or be sufficiently deep to have significant implications for the timing of failure due to copper corrosion, is considered to be small. Nevertheless, the evaluation of radionuclide release and transport in the case of a canister with such a defect is useful in illustrating the impact of uncertainties such as, for example, the characteristics of the buffer-rock interface, which may also impact releases following other failure modes.

In TILA-99 the variability with time of groundwater flow and salinity explicitly is not explicitly modelled. A range of flow and salinity combinations is considered, covering the expected ranges of variability and uncertainty, each of which is hypothetically assumed to persist for the entire period covered by the assessment (Table 5-4). In the KBS-3H safety assessment, flow at the buffer/rock interface is increased with respect to the Base Case in cases PD-HIFLOW, with no assumed change to the geosphere transport resistance. In case PD-HIFLOWR, flow at the buffer/rock interface is increased with respect to the Base Case, and geosphere transport resistance is decreased with respect to the Base Case. In cases PD-HIFLOW and PD-HIFLOWR, Q_f, is set to a value of 6.3×10^{-4} m^3 y^{-1}. This represents an increase of a factor of $\sqrt{(10)}$ with respect to the Base-case value of 2×10^{-4} m^3 y^{-1}, and corresponds, for example, to an order of magnitude increase in either the transmissivity (T) or hydraulic gradient.

iii) Comparison with the TILA-99 sensitivity analyses

Summary of TILA-99 sensitivity analyses

Sensitivity analyses carried out in TILA-99 addressed the effects of uncertainties in most of the key features of the assessment models chain:

- Canister failure time.
- Source term models.
- Very high solubility estimates for reducing conditions.
- Redox conditions in the near field.

- Transport along the tunnel.
- Penetration depth of matrix diffusion.
- Route dispersion in the far field.
- Alternative dose conversion factors.

In general, the TILA-99 sensitivity analyses assume median or high flow and transport data for non-saline and saline conditions. They address a single failed canister containing spent fuel from the Olkiluoto 1&2 reactors.

Common features and differences compared to the KBS-3H safety assessment

The focus of the analyses carried out in the KBS-3H assessment is 3H-specific uncertainties and uncertainties that are likely to affect KBS-3V and KBS-3H differently. Uncertainties in the radionuclide transport characteristics of the geosphere or uncertainties in the biosphere do not, therefore, fall within the scope of the analyses. Unlike TILA-99, where the focus was on a comparison of potential sites, the KBS-3H safety assessment has no sensitivity analyses addressing route dispersion in the far-field (Peclet number[19]) and alternative dose conversion factors, which are considered beyond the scope of the KBS-3H assessment analyses. It does, however, include sensitivity analyses addressing the near field that can be compared to those carried out in TILA-99.

Canister failure time

A canister is considered to have failed once its copper shell is penetrated. The TILA-99 reference scenarios consider initial small and large penetrating defects (holes), in which canister failure in effect occurs prior to canister emplacement, and the case of a "disappearing canister" which ceases to provide any resistance to water ingress and radionuclide release at 10,000 years following emplacement. Sensitivity analyses are, however, carried out assuming that the canister disappears immediately after the sealing of the repository (the DC0 scenario), and at 10^3, 10^5 and 10^6 years (the DC3, DC5 and DC6 scenarios, respectively). These scenarios are hypothetical – there is no physical mechanism suggested that would lead to canister "disappearance" at the assumed times.

"Disappearing canister" scenarios are only considered in the KBS-3H safety assessment in the context of specific canister failure modes – i.e. failure due to copper corrosion and rupture due to rock shear – where the timing of loss of resistance to water ingress and radionuclide release is determined by the scientific understanding of the specific processes leading to these failure modes.

In TILA-99, the evolution of the small and large initial penetrating defects is not explicitly treated. The presence of a defect is assumed to allow the ingress of water and the release of radionuclides immediately after sealing of the repository. It is further assumed that the resistance to water ingress and radionuclide release remains constant for all time. The KBS-3H safety assessment, on the other hand, defines cases addressing uncertainties in the evolution of an initial penetrating defect. In all KBS-3H assessment cases, it is assumed that it takes 1,000 years after canister emplacement before a transport pathway is established between the fuel and the canister exterior, based on SR-Can (see discussion in Section 5.3.3(ii)). Thereafter, in the base case for the initial penetrating defect failure mode (PD-BC), it is assumed that the transport resistance provided by the limited initial size of the defect is lost after a further 9,000 years (i.e. a total of 10,000 years after deposition), also according to SR-Can's defect evolution model.

[19] The Peclet number is a dimensionless factor, relating rate of transport of a substance by advection to transport by diffusion, used to describe dispersion in the geosphere.

Source term models

In both TILA-99 and the KBS-3H safety assessment, the source term consists of four components:

- Instant release fractions[20].
- Fuel matrix.
- Activation products in zircaloy.
- Activation products in other metal parts.

A significant source of uncertainty is the size and, for some radionuclides, the existence of a segregated and rapidly released fraction. This uncertainty is considered in both TILA-99 and in the KBS-3H safety assessment. In the TILA-99 reference scenarios, the instant release fractions are conservative values taken from the review of /Johnson and Tait 1997/. A sensitivity analysis is, however, performed using "realistic" values from the same review. In the KBS-3H safety assessment, more recent estimates are used, taken from SR-Can (Appendix A-2 in /SKB 2006a/). The central values used in SR-Can are assumed in the majority of assessment cases. The pessimistic (upper) instant release fractions given in SR-Can are, however, considered in a variant case named PD-IRF (see Table 5-2).

In light of the "difference analysis" approach adopted in the KBS-3H safety studies (see Section 1.3.1), the same fuel matrix dissolution model was used in the KBS-3H safety assessment as in SR-Can. This is different to that used in TILA-99.

In TILA-99, it is assumed that the fuel matrix is degraded by the products of alpha radiolysis occurring in a thin layer of water on the fuel surfaces. The release rate of radionuclides from the fuel matrix is taken to be directly proportional to the (time-dependent) alpha activity of the fuel. This initial fractional degradation rate of the fuel is conservatively taken to be 10^{-4} per year, decreasing to a lower limit of 10^{-6} per year. A still more conservative model variant is considered in sensitivity analyses. This is termed the "Finnish instant coffee model", in that the initial fractional fuel degradation rate of 10^{-4} per year is assumed to remain unchanged until the fuel has completely degraded and its radionuclides released. In the KBS-3H safety assessment, a constant fractional fuel degradation rate is used, following SR-Can. This model is based on several recent experimental studies performed on alpha-doped UO_2 and spent fuel under anaerobic, reducing conditions in the presence of a hydrogen atmosphere and corroding iron /Werme et al. 2004/. Werme et al. however, propose degradation rate values two to four orders of magnitude lower than that used in the Finnish instant coffee model. In the majority of KBS-3H assessment cases, the fractional degradation rate of the fuel is set to 10^{-7} per year (the central value proposed in /Werme et al. 2004/), with rates of 10^{-6} per year and 10^{-8} per year considered as variants addressing this source of uncertainty. It is acknowledged that other spent fuel dissolution models have been proposed and this is an issue for further work for both KBS-3H and -3V.

The treatment of the release of activation products from zircaloy and from other metal parts is identical in all assessment cases in TILA-99 and, with the exception of the PD-EXPELL case treating expulsion of dissolved radionuclides from the canister by gas (see Section 5.3.3), also in the KBS-3H safety assessment. A fractional rate of 10^{-4} per year (i.e. complete release in 10,000 years) is used for the zircaloy and 10^{-3} per year for other metal parts, according to TILA-99 (p. 101 of /Vieno and Nordman 1999/).

The assumed fractional corrosion rate of zircaloy of 10^{-4} per year is somewhat higher than the expected rate of corrosion /see Johnson and McGinnes 2002/ and is conservative, since it will lead to higher than expected radionuclide release rates.

[20] Certain radionuclides are enriched at grain boundaries in the fuel, at pellet cracks and in the fuel/sheath gap as a result of thermally-driven segregation during irradiation of the fuel in the reactor. These radionuclides are assumed to enter solution rapidly once water contacts the fuel pellet surfaces, and are termed the instant release fraction (IRF).

This high corrosion rate leads to an inconsistency, however, in that sustaining it would require more water than will enter through a defect of the postulated size[21]. In spite of this inconsistency, the conservative zircaloy corrosion rate from TILA-99 has been used in the radionuclide transport report, although the possibility of using a more realistic corrosion rate may be considered in future studies.

Near-field radionuclide solubilities

The near-field radionuclide solubilities assumed in the TILA-99 reference scenarios were conservatively chosen. In addition, TILA-99 considers a "very high solubilities" scenario, in which highly conservative solubilities for reducing conditions are assumed to take into account uncertainties in geochemical conditions.

A different method of evaluating near-field solubilities is used in the KBS-3H safety assessment, compared to TILA-99. Solubility limits for the KBS-3H near field have been estimated by /Grivé et al. 2007/ for a range of groundwater types relevant to the Olkiluoto site. The updated method is considered both better supported and more realistic. There remain uncertainties in solubilities but these are not considered relevant to discriminating between the 3V and 3H designs.

Redox conditions in the near field

In the TILA-99 reference scenarios, reducing conditions are assumed to prevail throughout the near field, except in a thin film of water adjacent to fuel surfaces, where, as noted above, it was assumed (for the purposes of evaluating fuel matrix degradation and radionuclide releases) that radiolytic oxidants are present. In a sensitivity analysis, it was further assumed that oxidising conditions resulting from radiolysis spread throughout the near field.

In the KBS-3H safety assessment, however, due to the large amounts of hydrogen and iron present around the fuel, this is no longer regarded as a plausible scenario and reducing conditions are assumed to prevail throughout the near field in all assessment cases.

Transport along the tunnel

In the TILA-99 reference scenarios, radionuclides escape from the repository near field via three routes:

- Through the buffer around the failed canister and into a transmissive rock fracture intersecting the deposition hole.
- From the top of the deposition hole into the excavation damaged zone below the tunnel floor.
- From the tunnel backfill into the rock via a fracture intersecting the tunnel.

It is assumed that a fracture intersects the tunnel close to the deposition hole. In a sensitivity analysis, however, it is assumed that radionuclides must migrate for a significant distance along the tunnel before encountering an intersecting fracture and entering the rock.

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[21] According to the Process Report (Section 2.7 of /Gribi et al. 2007/), the corrosion of the approximately 5,000 moles of zircaloy in a canister at a rate of 10^{-4} per year will produce ~1 mole of H_2 per year (one mole of zircaloy produces two moles of H_2), consuming 1 mole of water. However, it is also shown in the Process Report that the water flow rate into a canister with a one millimetre diameter defect is only in the order of 0.004 and 0.04 litres per year, or 0.2 to 2 moles per year, based on a hydraulic conductivity of the bentonite at the mouth of the hole of 10^{-13} to 10^{-12} m s^{-1} and a pressure difference across the buffer is 4.2 MPa (corresponding to the hydrostatic pressure at the depth of 420 m, the lower end of the inclined drift). Furthermore, plugging of the hole with bentonite or corrosion products and the decrease of the hydraulic gradient over time due to gas pressure buildup will reduce the rate of water inflow into the canister.

In the KBS-3H safety studies, most assessment cases address the situation where radionuclides are transported from a defective canister via the buffer to transmissive fracture intersecting the deposition drift near the canister location (radionuclide transport pathways R_1 and R_2 in Figure 5-1). Since transport in the bulk of the buffer is slow, being diffusion dominated, significant transport along the drift in the axial direction is considered possible only if the buffer-rock interface is perturbed (pathways R_3 to R_6), in particular by thermally-induced rock spalling or by chemical interaction of the buffer with the iron of the supercontainer or with cement (the impact of these processes in considered to be negligible in the majority of cases). Three cases are considered:

- A narrow, continuous high-permeability zone at the buffer -rock interface.
- A narrow, discontinuous high-permeability zone at the interface.
- A thick, discontinuous high-permeability zone at the interface.

iv) Comparison with the TILA-99 "what if" analyses

Summary of TILA-99 "what if" analyses

The "what if" analyses carried out in TILA-99 address the effects of:

- A combination of very high flow of non-saline groundwater and saline water chemistry.
- Very poor bentonite.
- Displacement of contaminated water out of the canister due to gas generation.
- Glacial meltwater.
- Post-glacial faulting.

As with the TILA-99 sensitivity analyses, the majority (all but the first) assume either median or high flow and transport data for non-saline and saline conditions, and consider a single failed canister containing spent fuel from the Olkiluoto 1&2 reactors.

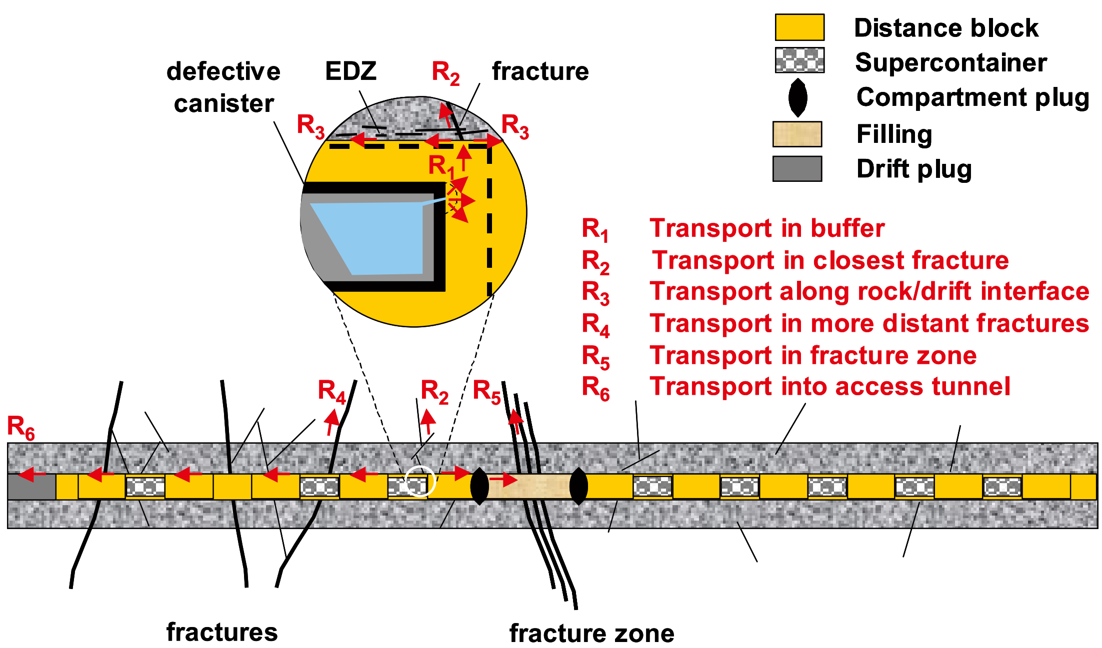

Figure 5-1. Radionuclide transport paths from a failed canister to the host rock in a KBS-3H repository.

Common features and differences compared to the KBS-3H safety assessment

Very high flow of non-saline groundwater and saline water chemistry

The "what if" combination of very high flow of non-saline groundwater and saline water chemistry is considered beyond the scope of the KBS-3H safety assessment, since it relates principally to geosphere uncertainty.

Very poor bentonite

The TILA-99 "what if very poor bentonite" analyses are intended to illustrate the effects of the bentonite buffer on radionuclide transport. They consider the hypothetical case of a deposition hole containing a failed canister (i.e. the canister has a small hole) being filled with the backfill mixture of crushed rock and bentonite, instead of compacted bentonite. The physical and chemical protection of the canisters afforded by the buffer is assumed to be unchanged, as is its capacity to filter any radionuclide-bearing colloids generated around the spent fuel. There are no corresponding assessment cases in the KBS-3H safety studies, although the possible adverse effects on the buffer of erosion due to penetration of dilute water to repository depth subsequent to glaciation, and due to post-glacial faulting are addressed in specific cases (see below). The effect of poor bentonite on corrosion of the copper canister is however assessed in the KBS-3H Evolution Report (see Appendix B case "D"; /Smith et al. 2007a/); even with very poor buffer, the canister lifetime is still of the order of 800,000 years.

Displacement of contaminated water out of the canister due to gas generation

Pressure due to gas generated by corrosion of the canister insert may expel water that has entered a failed canister along with dissolved radionuclides – principally those assigned to the instant release fraction. Although relevant to both the TILA-99 concept and KBS-3H, the likelihood of occurrence of contaminated water displacement by gas is higher for KBS-3H because of the greater likelihood that an initial penetrating defect occurs in an unfavourable position (i.e. a position on the underside of the canister). The potential effect of contaminated water displacement by gas is considered in both TILA-99 and the KBS-3H assessment. In addition, in the KBS-3H safety studies, an assessment case addresses the transport of radionuclides as volatile species by repository-generated gas – a potential process that is only discussed qualitatively in TILA-99 (pp. 205–207 of /Vieno and Nordman 1999/).

In TILA-99, it is assumed that a gas-driven water pulse begins at 100 years after canister deposition and lasts for a further 100 years. In the KBS-3H safety assessment, the fate of water/gas/vapour and radionuclides in the canister interior has been modelled in more detail, as described in the Process Report /Gribi et al. 2007/. On the basis of these modelling results, it is assumed that a gas-driven water pulse begins at 2,800 years after deposition and lasts for a further 1,300 years. In TILA-99, gas-driven water flow conveys the entire instant release fraction, plus 1% of the fuel matrix, 1% of the zircaloy and 10% of other metal parts, together with their associated radionuclides, which are assumed to enter solution. These inventories correspond to the releases from the waste forms during the first hundred years assuming that they have been in contact with water from the time of emplacement, and assuming TILA-99 dissolution rates. In the KBS-3H safety assessment, gas-driven water flow also conveys the entire instant release fraction and radionuclides released from the zircaloy/other metal parts but the dissolution of the fuel matrix is assumed to be negligible over the first few thousand years (note that the reference initial fuel dissolution rate is three orders of magnitude smaller in the KBS-3H safety assessment compared to TILA-99 - see the discussions of source term models, above).

Glacial meltwater

Glacial meltwater is of concern primarily because of its potential to give rise to low-ionic strength conditions at repository depth, which could result in buffer erosion. Furthermore, if dissolved oxygen were conveyed via fractures or crushed rock zones to repository depth, this could in principle affect both canister lifetime and the radionuclide retention properties of the buffer, backfill and the host rock. In TILA-99, it is argued that the potential impact on canister lifetime is limited, with a lifetime of around 10^5 years, even if oxic glacial meltwater is assumed to occupy the rock around the repository permanently. A scenario is considered in TILA-99 in which glacial melting brings about high flows in the geosphere and oxidising conditions in the buffer, backfill and geosphere. Glacial meltwater is not, however, considered to bring about oxidising conditions inside a canister with an assumed initial penetrating defect, due to the presence of large amounts of iron. Solubility limits appropriate to reducing conditions are thus applied within the canister.

In the KBS-3H safety assessment, it is argued that the migration of oxygen to repository depth is unlikely due to its interaction with minerals in the geosphere or possibly also due to microbial activity. Although glacial meltwater can be found at depth at Olkiluoto, there is no indication that oxygen-rich meltwater has penetrated to such depths in the past but it cannot be excluded for the future. The recent interpretation of the hydrogeochemical site data, and especially gas isotopic data, from Olkiluoto by /Pitkänen and Partamies 2007 and Andersson et al. 2007/ finds no evidence of oxidising meltwater intrusion into the deeper groundwater system at Olkiluoto (see Section 4.1.3). The case of oxidising conditions within and around the repository is therefore not addressed in TILA-99. However, in the KBS-3H safety assessment three assessment cases (PD-GMW, PD-GMWV and PD-GWMC) consider the penetration of glacial meltwater at repository depth in the case of a penetrating defect. Canister failure (water ingress through the penetrating defect) is also assumed to occur at 70,000 years in cases PD-GMW and PD-GMWV, with loss of transport resistance of the defect 9,000 years later. Since there is no radionuclide release in the first 70,000 years, the change in groundwater composition at 70,000 years is not explicitly modelled in these cases. In case PD-GWMC, there is assumed to be an initial penetrating defect that allows water ingress and radionuclide release after 1,000 years, as in the Base Case. The change in groundwater composition at 70,000 years is, therefore, explicitly modelled in this case. Oxidising conditions in the geosphere are considered with respect to transport of radionuclides from the repository, as in the case for TILA-99, but the conditions are reducing in the near field.

The possibility that low-ionic strength conditions will occur as a result of meltwater penetration, resulting in buffer erosion and eventually advective conditions in the buffer, reduced canister lifetime and increased radionuclide transport rates through the buffer subsequent to canister failure, is not considered in TILA-99. It is, however, addressed in the KBS-3H safety assessment in a group of cases addressing the canister failure mode due to copper corrosion (CC- cases in Table 5-2).

Post-glacial faulting

Large post-glacial earthquakes in the vicinity of a repository could lead to rock shear movements on fractures intersecting the TILA-99 concept boreholes or the KBS-3H repository drifts. Such movements could lead to changes in mass transport properties of the fractures and, if the shear movements are sufficiently large, could cause canister rupture. TILA-99 considers two variants of a scenario in which post-glacial faulting occurs at 30,000 years:

- A relatively mild variant in which a canister with an assumed initial penetrating defect experiences high flows in the geosphere and oxidising conditions in the buffer, backfill and geosphere (similar to the TILA-99 glacial meltwater scenario described above, except that oxidising conditions are assumed to prevail inside as well as outside the canister)
- A more severe variant in which an initially intact canister is assumed to "disappear" at 30,000 years, the surrounding buffer is assumed to be degraded and, again, high flows in the geosphere and oxidising conditions in both the near field and geosphere are assumed.

In TILA-99, no estimate is given of the likelihood of canister damage by post-glacial faulting whereas, in the KBS-3H safety assessment, the conservatively evaluated expectation of the number of canisters damaged by rock shear is given as 0.32 in a 100,000 year period (or 1 canister in 300,000 years). Thus a single failed canister at 100,000 years is postulated and, as in the severe TILA-99 variant where a canister disappears at 30,000 years, is modelled as "disappearing" for the purposes of radionuclide release and transport calculations in this assessment.

Also, as the shear movement that damages the canister occurs on a fracture intersecting the deposition drift at the canister location, it is assumed to cause displacement that reduces the minimum transport distance for radionuclides through the buffer by 15 cm. It is furthermore assumed that, due to the recent movement on the fracture, its transport resistance is reduced such that it makes no contribution to the transport resistance of the near field. The geosphere transport resistance is set to WL/Q = 1,000, which is a factor of 50 lower than in the Base Case for an initial penetrating defect (case PD-BC), in order to account for the possibility that the shear movement that damages the canister also has a detrimental impact on the geosphere. It is, however, assumed that reducing conditions are maintained in the near field, due to the large amount of iron present, and also in the geosphere. The overall effect of this KBS-3H rock shear (RS) case is then less pessimistic than the severe TILA-99 case since there is no assumption of oxidising conditions. Glacial meltwater and the assumption of oxidising conditions in the geosphere are, however, considered in the KBS-3H RS-GMW case.

5.3.3 Comparison with SR-Can

i) Overview

The KBS-3H safety assessment follows the SR-Can assessment in that, as noted in Section 5.3.1, the majority of calculational cases addressing uncertainties not specifically related to a canister failure mode are mainly assigned to the group dealing with an initial penetrating defect.

In SR-Can /SKB 2006a, Section 10.5.1/, the use of this failure mode is explained thus:

"This failure mode is not explicitly addressed in the reference evolution since the initial state of the canisters suggests that there will be no penetrating pinhole defects in the copper shell. An analysis of this failure mode is, however, relevant in addressing important aspects of the internal evolution of the canister. For the pinhole failure mode, the canister possesses no transport resistance, whereas the buffer and the geosphere have intact retention properties. It is, therefore, also a convenient case for demonstrating the retarding capacity of the buffer and the geosphere and for exploring uncertainties relating to these components of the repository. Furthermore, an initially small defect is eventually expected to evolve into a large defect, which resembles the case of a failure caused by general corrosion of the canister, when the buffer is still intact. Although the likelihood of this latter failure mode was found negligible in the reference evolution, it is of interest to understand its consequences."

Three alternative canister failure modes are also assessed in SR-Can with additional cases which permit other aspects of the system to be examined:

- Advection/canister corrosion failure (buffer erosion leading to advective conditions enhancing to canister corrosion rate).
- Shear movement failure (buffer thickness reduced but function retained).
- Isostatic load failure (buffer function retained).

The basis for the definition of the cases is the Reference Evolution and two variants of the reference evolution are analysed:

- A base variant in which the external conditions during the first 120,000 year glacial cycle are assumed to be similar to those experienced during the most recent cycle, the Weichselian.

Thereafter, seven repetitions of that cycle are assumed to cover the entire 1,000,000 year assessment period.

- A greenhouse variant in which the future climate, and hence external conditions, are assumed to be substantially influenced by anthropogenic greenhouse gas emissions.

Table 5-5 gives the assessment cases for SR-Can (for both Forsmark and Laxemar sites unless otherwise indicated) based on the reference evolution.

A more exhaustive treatment of uncertainties for all conceivable canister failure modes, going beyond the assumptions made in the reference evolution, is given in the evaluation of scenarios in SR-Can. This leads, for example, to a more pessimistic evaluation of the advection/corrosion failure mode /see Chapter 12, SKB 2006a/.

Table 5-6 shows a summary of relevant issues and indicates how they are dealt with in SR-Can and the KBS-3H safety assessment, whether by defining and analysing assessment cases or by qualitative arguments, or whether they are considered to be irrelevant to the design or outside the scope of the assessment.

The KBS-3H safety assessment considers just the single site at Olkiluoto in contrast to SR-Can in which most cases listed in Table 5-5 are calculated for both the Forsmark and Laxemar sites. The exceptions are the 2 cases for the growing pinhole failure (PH) which address alternative hydrogeological interpretations of the Forsmark site. The PH base case uses a discrete fracture network (DFN) model in which the size of fractures is fully correlated with their transmissivities. The alternative DFN for Forsmark used in the variant case considers only semi-correlation between fracture size and transmissivity. The second variant case for Forsmark uses the multi-component Continuum Porous Medium (CPM) model. The CPM model could be considered unrealistic, since it is only discrete fractures that are transmissive. However, the current hydrogeological DFN model may overestimate the connectivity of the fracture system. The CPM model is thus judged to be a fair representation of the spectrum of possible interpretations of current hydraulic data that show very low – if any – transmissivity at depth.

Based on the information in Site Description Model (SDM) version 1.2 of the Forsmark site /SKB 2005/, it is not possible to rule out any of these three representations as unlikely, or to state that any of them is a distinctly more plausible representation of the site than the others.

It should be noted that only a certain range of fracture sizes are considered in the DFN models, from a few metres up to 1,000 metres and it is the correlation for this limited set that is of interest when evaluating the representativeness of the DFN variants. Also the different hydraulic variants of the Forsmark site have radically different consequences in terms of canister failures and hence safety.

For the Laxemar site, only a semi-correlated DFN model is considered in SR-Can since confidence in the Laxemar site descriptive model is limited and recent data from the site suggest, in particular, that its hydraulic properties are more favourable than suggested by the site descriptive model version 1.2 /SKB 2006b/.

Although several types of fuel are to be deposited in the SR-Can repository, the assessment cases are carried out for a single reference fuel type, namely SVEA-96 BWR fuel with a burn-up of 38 MWd/kg U /SKB 2006e/. As with the KBS-3H safety assessment, previous assessments have included variant cases for different fuel types and it was considered unnecessary for the objectives of the SR-Can to repeat them.

ii) Comparison with SR-Can base cases for canister failure modes

While SR-Can and the KBS-3H safety assessment have 3 canister failure modes in common, as noted above, there are some differences in the way that that the barriers are treated after failure for the various failure modes.

Table 5-5. Overview of assessment cases for radionuclide release and transport in SR-Can.

Canister failure mode	Radionuclide release and transport cases
Growing pinhole failure	Base case for canister with initial penetrating pinhole (BWR fuel, burn-up 38 MWd/kg U, for all cases):
	– Forsmark: fully correlated[1] DFN geosphere
	– Laxemar: semi-correlated DFN geosphere
	Cases addressing alternative Forsmark hydrogeological interpretations:
	– Semi-correlated DFN geosphere
	– Continuous porous medium (CPM) geosphere
	Case addressing uncertainty about spalling in deposition holes (no spalling)
	Case addressing sensitivity to alternative properties of the EDZ
	Cases addressing sensitivity to less favourable backfill properties in the tunnels:
	– High hydraulic conductivity
	– Loss of swelling pressure in backfill material
	Case addressing gas expulsion of dissolved radionuclides from the canister
	Case addressing sensitivity to deposition hole acceptance/rejection criteria
	Case considering co-precipitation of radium in the canister
	Cases addressing altered climate conditions:
	– Permafrost
	– Ice margin
	– Ice margin + enhanced groundwater flow conditions
	– Greenhouse
Advection / canister corrosion failure	Base case for canister failure due to advection and corrosion:
	– Forsmark: semi-correlated DFN geosphere (10 canister failures)
	– Laxemar: semi-correlated DFN geosphere (50 canister failures)
	Cases addressing sensitivities in the base case:
	– Sensitivity to fuel dissolution rate
	– Sensitivity to concentration limits (i.e. no credit taken for concentration limits)
	Case addressing sensitivity to deposition hole acceptance/rejection criteria
Shear movement failure	Base case for canister failure by shear movement (with no radionuclide retention in the geosphere)
Isostatic load failure	Base case for canister failure by isostatic load failure
Additional cases	
	Cases addressing radionuclide transport in the gas phase (excluding advection/corrosion failure mode)
	Cases to illustrate barrier function (for Forsmark semi-correlated DFN):
	With base case geosphere:
	– An initial, large opening in all canisters
	– An initial absence of enough buffer to cause advective conditions for all deposition holes
	– A combination of the above two, i.e. an initial, large opening in all canisters and advective conditions due to loss of buffer for all deposition holes
	With loss of the radionuclide retention capability of the rock:
	– An initial, large opening in all canisters
	– An initial absence of enough buffer to cause advective conditions for all deposition holes
	– A combination of the above two, i.e. an initial, large opening in all canisters and advective conditions due to loss of buffer for all deposition holes

[1] Discrete fracture network (DFN) with full correlation between fracture sizes and transmissivities.

Table 5-6. Summary of relevant issues and how they are dealt with in SR-Can and in the KBS-3H safety assessment.

Issue		SR-Can	KBS-3H
Groundwater properties	Uncertainties in flow	A	A
	Uncertainties in salinity	A	A
Far-field transport	Far-field dispersion	A	O/S
Biosphere	Dose conversion factors	A	O/S
Canister failure	Mode	A	A
	Timing	A	A
Source term	Uncertainties in instant release fraction	A	A
	Uncertainties in fuel dissolution rate	A	A
Solubilities	Uncertainties in values	A	A
Radiolysis	Impact on source term	Q	Q
	Impact on near field redox conditions	Q	Q
Transport through backfill or along buffer/ rock interface	Location of nearest intersecting fracture	A	Q
	Impact of rock spalling	A	A
	Impact of iron/bentonite interaction	O/S	A
Buffer emplacement	Very poor bentonite	Q	A
Gas	Displacement of contaminated water	A	A
	Transport of volatile species	A	A
Glacial meltwater	Buffer erosion/canister corrosion	A	A
	Changes to redox conditions	Q	Q
Post-glacial faulting	Physical damage to canister	A	A
	Physical damage to buffer	A	A
	Physical damage to rock	A	A
	Changes to redox conditions	Q	Q

A	Covered by an assessment case
Q	Dealt with by qualitative argument
O/S	Defined as outside assessment scope or irrelevant to variant

Initial penetrating defect

In the case of the initial penetrating defect, in SR-Can, the pinhole is circular with an initial radius of 2 mm, i.e. an area of 12.5 mm^2. This is considerably larger than that considered in the KBS-3H PD base case (PD-BC) which has a radius of 0.5 mm, although a larger hole of radius 2 mm has been considered as a variant case (PD-BIGHOLE, see Table 5-2).

In the KBS-3H penetrating defect base case (PD-BC), the initial pinhole develops into a larger defect after 9,000 years at which point the residual transport resistance of the canister is neglected. In SR-Can it is assumed that a further 1,000 years (i.e. a total of 10,000 years after deposition) are required before a transport pathway is established between the fuel and the canister exterior. SR-Can's assumption is based on the slow water ingress rate, further decreased by the gradual build-up of an internal counter pressure due to hydrogen gas formation, as well as on the barrier functions of the cast iron insert and of the fuel cladding. Also, according to SR-Can, loss of transport resistance may occur at any time from 1,000 to 100,000 years after the transport pathway between the fuel and the canister exterior is first established. Additional assessment cases are therefore defined in which the delay before loss of transport resistance is either increased to 101,000 years or reduced to 2,000 years after canister deposition, in order to cover the range of uncertainty. According to Section 10.5.2 of /SKB 2006a/, 1,000 years can be regarded as pessimistic, since any one of these factors is likely to provide more than 1,000 years of delay.

The activation products incorporated within metal parts of the fuel elements are assumed to be released congruently with metal corrosion in the KBS-3H safety assessment but without any such delay in SR-Can – a more pessimistic assumption. In the penetrating defect base case in SR-Can, all deposition holes are assumed to be affected by spalling, unlike the disposal tunnel for KBS-3H PD-BC.

Failure due to copper corrosion

In SR-Can, modelling was carried out to predict the failure rate of canisters due to erosion of the buffer and the subsequent enhanced copper corrosion rate. The number of failed canisters is based, for both sites, on semi-correlated DFN model. For Forsmark, 10 canisters are estimated to fail between about 500,000 and 1 million years. For Laxemar, no failures occur during the initial 100,000 years and 50 canisters fail between 100,000 and 1 million years. The Forsmark fully correlated DFN model, assuming application of the FPI[22] (Full Perimeter Intersection) criterion when selecting deposition holes, yields no canister failures due to advection/corrosion during the one million year assessment period /SKB 2006c, Section 9.4.9/. Pessimistically, therefore, the semi-correlated DFN model for Forsmark was used as the base case for the advection/corrosion failure mode. Spalling was assumed when calculating the bentonite erosion rate but it has insignificant influence on canister corrosion and radionuclide transport since these phenomena are driven by advection.

In the KBS-3H safety assessment, a single canister failure was assessed, which is similar to the SR-Can deterministic variant case for Forsmark that also considers a single failure. Following canister failure, the same groundwater flow and geosphere transport resistance are assumed in the KBS-3H Copper Corrosion base case as in the PD-base case. The choice of transmissivity of the fracture intersecting the drift at the location of the failed canister and the geosphere transport resistance are considered to be moderately pessimistic. Pessimistically chosen parameters related to groundwater flow are also appropriate for this failure mode, especially because the canister positions most vulnerable to failure will be those associated with the highest groundwater flows at the buffer/rock interface.

In the SR-Can base case for advection/corrosion failure, there is a delay before release of radionuclides after failure of the canister due to corrosion of the insert. This delay was determined to vary between 1,000 and 100,000 years and, in the calculation, a triangular distribution of failure times with the peak at 100,000 years was used. However, given the long time before failure of a copper canister, the additional time for corrosion of the insert does not have significant effect on the results. The KBS-3H safety assessment conservatively takes no benefit for a delay between canister failure and release of radionuclides as the canister and insert both "disappear" at 100,000 years.

The inventory of activation products in the metal parts of the fuel is generally assigned to the instantaneously accessible fraction (IRF) in SR-Can, as it was considered unnecessary to develop a model for the metal parts since nuclides in these will be dispersed by the buffer in any case. However, in the advection/corrosion case, this simplification would lead to unrealistically high releases of Ni-59 and Nb-94 and it is therefore assumed that the radionuclides present in the metal parts will be released over a period of 1,000 years. This is judged to be a gross overestimate of the release rate as the metal parts consist of corrosion-resistant alloys.

The IRF gives rise to a pulse to the geosphere and biosphere. The dose conversion factors (termed Landscape Dose[23] Factors [LDFs], which are site-specific, in SR-Can), however, are calculated for constant releases over long periods. These long periods allow near steady-state

[22] FPI criterion (usually denoted FPC in SR-Can, /SKB 2006a/) leads to the rejection of a deposition hole if a fracture which intersects the whole tunnel perimeter is expected to also intersect the deposition hole.

[23] In the safety assessment, the annual effective dose to the most exposed individual calculated using the biosphere dose assessment model is termed the "annual landscape dose".

situations to develop and ensure that the effects of downstream accumulation are included in the dose calculation. For a pulse, a steady state does not develop for many radionuclides and downstream accumulation is very low compared to the initial release. Moreover, the annual average lifetime risk will be lower for a short release occurring over less than a lifetime, i.e. less than about 50 years. In SR-Can, a pulse release of X Bq to the biosphere is converted to a constant release occurring over a lifetime, assumed to be 50 years, i.e. to X/50 Bq/y, to which the LDF is applied. This approximation is considered to be a cautious way of estimating doses from pulse releases to the biosphere and results in an upper bound for doses arising from the IRF nuclides[24].

In both SR-Can and KBS-3H cases, solubility limits are not applicable in the canister as the high water flux reduces nuclide concentrations below elemental solubility limits. The one exception is uranium for which a concentration limit is still an effective constraint on release, due to the large amount of U-238 present in the fuel. This limits the near-field releases of the uranium isotopes but leads to increased releases of Th-230 (and Ra-226), Th-229 and Pa-231 generated by decay of the re-precipitated U-234, U-233 and U-235, respectively.

Rupture due to rock shear

The cases for rupture of the canister due to rock shear are treated very similarly in SR-Can and the KBS-3H assessment: in both cases, the failure of one canister at 100,000 years is assessed, with no transport resistance from the canister after failure and an assumed loss of 15cm of buffer thickness due to the shearing across the deposition hole/deposition drift. Uncertainty about the condition of the fracture zone after movement is treated by assuming an increased water flow (Q_{eq} – the equivalent flow rate at a deposition hole – increases from the (deterministic) pinhole base case value of around 5 x 10^{-6} m^3/y to 1 m^3/y in SR-Can; radionuclide concentrations in the boundary layer are assumed to be zero in the KBS-3H safety assessment) and by taking no credit for transport resistance in the near field or radionuclide retention in the geosphere.

Rupture due isostatic loading

This case is not treated in the KBS-3H safety assessment and is also considered to be extremely unlikely in SR-Can. The canister and insert are bypassed after failure but the buffer and geosphere are not affected, thus the case resembles that of the initial penetrating defect.

iii) Comparison to SR-Can sensitivity analyses

SR-Can uses a probabilistic assessment method that samples from a range of values, for which a distribution is defined, for some parameters (more details are given in the SR-Can Data Report, SKB 2007e) so that uncertainty in these parameters is taken into account. For example, for the initial penetrating defect case:

- Instant release fraction – triangular distribution for all nuclides except actinides, Sm-151 and Zr-93 (0% IRF) and Ca-41, Nb-94, Ni-59 and Ni-63 (100% IRF).
- Time between onset of nuclide transport and loss of transport resistance in canister – triangular distribution (0, 10^3, 10^5 years) used to illustrate uncertainties and risk dilution. In PD-BC the loss of transport resistance happens in 9,000 years.
- Fuel dissolution rate – log-triangular distribution (10^{-8} /y, 10^{-7} /y, 10^{-6} /y).
- Buffer porosities – anions: triangular (0.12, 0.17, 0.24); cations: constant (0.43).
- Buffer diffusivities – triangular distribution.
- Buffer sorption coefficients – log-triangular distribution.

[24] Note that the IRF is spread over a life time (50 years) **or** divided by the reduction factor obtained from the geosphere dispersion, τ, whichever yields the lowest dose /see SKB 2006a, p.443/.

SR-Can does however also consider several cases which examine the sensitivity of the results of the base case for a penetrating defect to uncertainties in the reference evolution:

- Alternative interpretations of the Forsmark hydrogeology.
- Effect of thermal spalling in the disposal holes.
- Extensive EDZ.
- High hydraulic conductivity in the deposition tunnel.
- Loss of swelling pressure in the deposition tunnel backfill.
- Gas in canister interior expelling contaminated water.
- Sensitivity to deposition hole rejection/acceptance criteria.
- Co-precipitation of radium in the canister.

Of these cases, many are not relevant to the KBS-3H assessment because of:

- The differences in the safety case objectives, e.g. alternative hydrogeological models.
- The -3H design, e.g.
  - the effect of spalling in deposition holes – although this is addressed for the deposition drift in the KBS-3H safety assessment,
  - disposal tunnel backfill conditions,
  - deposition hole acceptance/rejection criteria – although here there may be a equivalent in terms of the allowable fracture transmissivity or water inflow rate at a supercontainer emplacement position.
- The way in which the concept is modelled in the KBS-3H Radionuclide Transport Report /Smith et al. 2007b/, e.g. the EDZ is assumed not to be hydraulically significant because the "outlet" fracture is located at the point of canister failure in order to minimise the effect of transport distance to the geosphere (see Figure 5-2). Therefore, the EDZ does not affect the transport of radionuclides migrating in solution. It is, however, implicitly taken into account in cases addressing the release of volatile C-14 with repository-generated gases (PD-VOL-1 and PD-VOL-2).

Expulsion of contaminated water by gas

Of the remaining cases listed above, the KBS-3H safety assessment also considers the case of contaminated water expelled by gas from the canister but there are differences in the way that this case is treated between the two assessments. In SR-Can, no credit is taken for the 1,000 year, or more, delay caused by the time to generate sufficient gas to expel the water. Also, the IRF inventory is released instantaneously to a buffer compartment with a water volume of ~ 1 m^3. From this point, the transport processes are as in the pinhole (PH) base case.

The KBS-3H case is rather less pessimistic with respect to duration but more so with respect to the nuclides affected and buffer performance: it is assumed that a gas-driven water pulse, beginning at 2,800 years after deposition and lasting for a further 1,300 years, propels water from the canister interior through the buffer directly to the fracture. It is further assumed that all radionuclides in the zircaloy and other metal parts are combined with the IRF and, commencing 2,800 year after deposition, are released directly from the canister interior to the geosphere at a fractional rate of 7.7×10^{-4} per year (i.e. complete release in 1,300 years). For simplicity, the stability of the zircaloy, which may in reality retain some radionuclides for longer than this, is conservatively neglected.

The likelihood of this case and the possibility of gas expulsion that could occur from multiple canisters at similar times have not been evaluated in the KBS-3H safety assessment.

Alternative climatic conditions

A further set of cases in SR-Can considers altered climatic conditions for the initial penetrating defect failure mode. Such cases are outside the scope of the KBS-3H safety assessment which has no equivalent climatic cases although different groundwater salinities, including glacial meltwater, are addressed by alternative cases.

However, two climate scenarios have been considered in the Evolution Report /Smith et al. 2007a/. One corresponds to the repetition of the last glacial cycle (Weichselian-R climate scenario) and the other reflects a moderate increase in the level of CO_2 emissions (the Emissions-M scenario).

Additional cases for advection/canister corrosion failure

In addition, in SR-Can there are two cases addressing sensitivities in the advection/canister corrosion failure mode, i.e.:

- To fuel dissolution rate.
- To concentration limits (i.e. no credit taken for concentration limits).

They essentially examine the alternatives for the releases to the geosphere in the absence of diffusion control of the near field. In KBS-3H safety assessment, two cases consider increased and reduced fuel dissolution rates by a factor 10 (PD-HIFDR and PD-LOFDR, respectively, see Table 5-2).

Sensitivity to deposition hole acceptance/rejection criteria is also assessed for the advection/canister corrosion case, as this reflects an uncertainty in the reference evolution. Different transmissivities for acceptance criteria and application of the FPI criterion result in variations between zero and 40 canister failures at the Forsmark site.

Transport of volatile nuclides in gas

SR-Can considers an alternative process in a case addressing radionuclide transport in the gas phase; this case is applicable to all failure modes except advection/canister corrosion as, in this case, there is no buffer to retain a gas phase. The KBS-3H safety assessment has a similar case for transport of radionuclides as volatile species by gas. A significant difference between the two treatments of this case is that, in the KBS-3H safety assessment, an initial penetrating defect is assumed, thus the time of gas release is dependent on the gas generation rate since the defect is initially present. Half the gas and C-14 inventory (the only nuclide considered in this case) is lost at the initial breakthrough with the remaining C-14 being released as further gas is generated. The C-14 as methane is transferred directly from the canister to the geosphere where it is assumed to either be transported as gas into the biosphere, where oxidation and dissolution into carbonate species occurs, or alternatively the methane gas is oxidised to dissolved carbonate species in the geosphere with concomitant reduction of sulphate by sulphate-reducing bacteria at the interface between sulphate-rich brackish water and saline water.

In contrast, in SR-Can, the time for gas breakthrough is determined by the failure time of the copper shell as well as the corrosion rate of the canister insert. At the time of gas breakthrough, half of the inventory of C-14 and Rn-222 is assumed to be released immediately to the biosphere. The remaining gaseous inventory (and the Rn-222 that is produced) is then assumed to be released together with the gas that is produced continuously. However, this release is neglected, since it will be insignificant in comparison with the pulse release. If the release occurs in the first 10,000 years (unlikely), the release of C-14 would be ~ 10 GBq, compared to release of Rn-222 which would be about 25 GBq if the release occurred after 100,000 years. Annual mean lifetime risk from the gaseous pulse in the biosphere is calculated for ingestion (C-14) and inhalation outdoors and indoors.

SR-Can cases to illustrate barrier function

Finally, a number of additional cases are made in SR-Can to illustrate barrier function – these might be termed "what if" cases as they address extreme combinations of loss of buffer and canister functions and are not discussed further here as they have no equivalent in the KBS-3H safety assessment.

Further KBS-3H safety assessment cases

The KBS-3H safety assessment, on the other hand, has additional cases to investigate uncertainties with different significance for, or potential impact on, KBS-3H compared to KBS-3V. In particular, the cases addressing processes at the buffer-rock interface, involving spalling or iron-bentonite interaction, which were described in the preceding Section 5.3.2 (iii), and a case addressing potential loss or redistribution of buffer mass. This case is to account for the number of processes that have been identified and which could lead to a loss of bentonite mass from, or redistribution of bentonite mass within, a drift compartment especially in the early phase of repository evolution including the operational phase. Scoping calculations have been performed to estimate the impact of these processes on buffer density (see Appendix B in /Gribi et al. 2007/).

5.3.4 Comparison with other assessments conducted internationally

In Table 5-7, the main cases or scenarios (terminology varies between assessments) analysed in a number of recent safety assessments are summarised.

It was noted in TILA-99 that all safety assessments of spent fuel disposal in copper canisters in crystalline bedrock, including Swedish /SKI 1991, 1996, SKB 1992, 1995/, Canadian /Wikjord et al. 1996, Gierszewski et al. 2004/ and Finnish assessments /Vieno et al. 1992, Vieno and Nordman 1996/, as well as TILA-99 itself, had come to the conclusion, or have implicitly assumed, that initially intact copper canisters preserve their integrity for a very long time subject to the influence of the expected normal evolution of the disposal system. Thus, in order to analyse radiological consequences, it has to be assumed that one or more canisters is initially defective. In SR-Can, although no initial penetrating defects are expected, evolution in case of a growing pinhole failure is described and its consequences evaluated (Section 10.5 of /SKB 2006a/). Following the same approach, in the Radionuclide Transport Report /Smith et al. 2007b/ the processes due to the presence of a hypothetical initial penetrating defect are also considered.

It is notable in Table 5-7, that the two assessments for concepts with copper canisters (OPG Third Safety Case and SR-97) have a base scenario, which has no consequences, but also consider a defective container scenario whereas the other assessments, of steel canisters, do not need a defective container scenario as canisters are assumed to fail, usually at a conservative point in time such as 1,000 years after emplacement, as part of the normal evolution.

The "Altered (characterisation defect) scenario" used by Andra is a case designed to demonstrate the robustness of the EBS even with the failure of the geosphere barrier to perform as expected. Other than this, the cases or scenarios covered are very similar in subject and intention, allowing for the limitations of scope of individual assessments.

5.3.5 Conclusions from comparison of assessment cases

The preceding section examining the assessment cases used in the KBS-3H safety assessment, TILA-99 and SR-Can demonstrate that there are no omissions from the KBS-3H assessment, other than for uncertainties excluded explicitly from consideration due to the assessment objectives or the KBS-3H design. Moreover, the supporting explanations and analyses from the Process and Evolution reports /Gribi et al. 2007 and Smith et al. 2007a/ are used to justify differences in the approach taken to treating some processes for KBS-3H, particularly compared to what was done in SR-Can.

Table 5-7. Cases or scenarios explicitly analysed in recent safety assessments (modified from /Gierszewski et al. 2004/)[1].

Safety assessment name and references	Cases or scenarios analysed
OPG Third safety case	– Base scenario
Canada /Gierszewski et al. 2004/	– Defective container scenario
	– Human intrusion scenario
Dossier 2005 Granite	– Normal evolution scenario
France /Andra 2005a/	– Altered (characterisation defect) scenario
H12	– Base scenario
Japan /JAEA 2000/	– Uplift and erosion scenario
	– Climate and sea-level change scenario
	– EBS construction defect scenario
	– Human intrusion scenario
SR 97	– Base scenario
Sweden /SKB 1999/	– Canister defect scenario
	– Climate scenario
	– Earthquake scenario
	– Intrusion scenario
Project Opalinus Clay	– Reference scenario (groundwater transport)
Switzerland /Nagra 2002a/	– Gas transport scenario
	– Human intrusion scenario
	– "What if?" scenarios
	– Alternative design scenarios
	– Alternative biosphere scenarios

[1] Note that many of these scenarios also include variants or sub-scenarios.

5.4 Models

5.4.1 General approach and computer codes

The general approach to analysing the assessment cases in the KBS-3H safety studies, in common with most safety assessments carried out internationally, is to separate near-field, geosphere and biosphere modelling. Advective transport is assumed to dominate in the geosphere, whereas diffusion is the dominant transport mechanism in the near field (taken here to comprise the buffer and the canister, including the canister interior). Conservatively, it is assumed that radionuclides may diffuse from the near field to the geosphere but not vice versa.

Near-field analyses have been performed with the REPCOM code /Nordman and Vieno 2003/. REPCOM is a compartment model that has been developed by the Technical Research Centre of Finland (VTT) for radionuclide transport analyses in the near field of repositories for low and intermediate level waste or spent fuel. The phenomena that can be modelled using REPCOM are:

- Release from the waste – several waste types, each with different release functions, can be included.
- Advective and/or diffusive transport within a system of engineered barriers.
- Sorption on solid surfaces.
- Solubility limitation of concentrations.
- Radionuclide decay and ingrowth.

Detailed descriptions of how these phenomena are treated, including the governing equations solved by REPCOM, are given in /Nordman and Vieno 1994/.

Geosphere (far-field) analyses have been performed with the FTRANS code /FTRANS 1983, Nordman and Vieno 1994/. FTRANS is a dual-porosity model for flow and transport. In the flow porosity domain (in the KBS-3H safety assessment taken to be a single, representative geosphere fracture), phenomena that can be modelled with FTRANS are:

- Groundwater flow.
- Advective radionuclide transport.
- Longitudinal dispersion (not considered in the KBS-3H safety studies).

In the matrix porosity domain (the wallrock adjacent to the fracture), phenomena that can be modelled are:

- Diffusion.
- Sorption on solid surfaces.

Radioactive decay and ingrowth are represented in both domains and transfer of radionuclides across the boundary between the domains takes place by diffusion. Verification of REPCOM and FTRANS is described in the Radionuclide Transport Report and references therein /Smith et al. 2007b/.

Releases from the near field to the biosphere are converted to dose using dose conversion factors derived for an indicative stylised drinking water well scenario (WELL-2007). This uses assumptions that are the same as in TILA-99, but with updated ingestion dose coefficients. Further discussion of the biosphere is considered to be outside the scope of the present report.

The following sections give more details of the near field and geosphere modelling approaches adopted in the base case for a canister with an initial penetrating defect, and comparisons are made with the TILA-99 and SR-Can approaches. Other KBS-3H cases are analysed using variants on these approaches, as described in detail in the Radionuclide Transport Report /Smith et al. 2007b/.

5.4.2 The approach to near field modelling

The geometry of the domain addressed by the base case near-field model for a canister with an initial penetrating defect is illustrated in Figure 5-2. Geometrical parameter values are given in Table 5-8.

A drift section containing a canister with an initial penetrating defect is assumed to be intersected by a single transmissive fracture. Conservatively, it is assumed that the canister is located at a position that minimises the transport distance across the buffer between the defect and the fracture mouth (i.e. the centre plane of the fracture is assumed to pass through the centre of the defect, as illustrated in Figure 5-2).

The spent fuel dissolution model used in SR-Can has been used in all safety assessment cases. Alternative models for spent fuel dissolution have been proposed but further discussion of these models is beyond the scope of this report. Following SR-Can, it is assumed to take 1,000 years for contact of water with the fuel/metallic parts to take place and for transport pathways to be established; this value is regarded as pessimistic (see, e.g. p. 404 of SR-Can Main Report, /SKB 2006a/). Thereafter, the supply of water to the canister interior is conservatively assumed to be unlimited and fuel dissolution is assumed to take place at a constant fractional rate. The gap inventory is conservatively assumed to be instantaneously released and mixed with water ("instant release fraction"). It is further assumed that the inventory of activation products in metallic parts is released congruently with the corrosion of the metal. Released radionuclides are dissolved in water occupying the void space in the canister interior or are precipitated if relevant solubility limits are exceeded. Any colloids formed when solubility limits are exceeded are assumed to remain in the canister interior.

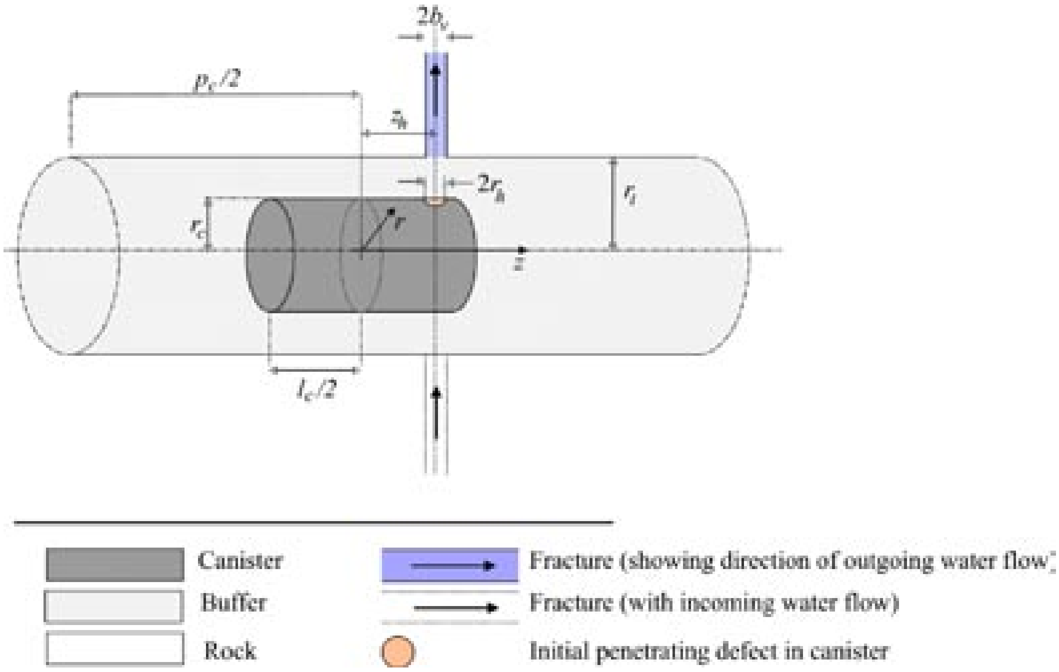

__Figure 5-2.__ Geometrical domain of the near field model in the base case for an initial penetrating defect. Parameter values are given in Table 5-8. Other geometrical parameters and parameter values are defined in the Radionuclide Transport Report /Smith et al. 2007b/.

Table 5-8. Geometrical parameter values for the near field model in the base case for an initial penetrating defect – symbols are those defined in Figure 5-2.

Parameter	Unit	Symbol	Value	Source
Canister outer diameter	m	2rc	1.05	/Raiko 2005/
Canister length	m	lc	4.835	/Raiko 2005/
Canister pitch	m	pc	11.0	/Autio et al. 2007/
Drift diameter	m	2rt	1.850	/Börgesson et al. 2005/
Hole diameter	m	2rh	10^{-3}	See main text
Hole length	m	dh	0.05	Copper shell thickness
Hole position (axial distance of hole centre from canister centre)	m	zh	2.417	Hole taken to coincide with canister end (distance of weld from end assumed negligible)
Fracture aperture	m	2bv	3×10^{-5} (1)	See Radionuclide Transport Report /Smith et al. 2007b/

[1] The fracture aperture is based on the relationship $b_v = \frac{\sqrt{T}}{2c}$. *T* [$m^2/s$] is the fracture transmissivity and *c* is a constant, given in /Lanyon and Marschall 2006/ as 2 $s^{-1/2}$.

Once a radionuclide transport pathway is established, the transport of dissolved radionuclides from the canister interior to the defect is conservatively assumed to be instantaneous. Transport resistances of the inner structural parts of the canister, the fuel and the fuel cladding are disregarded, being subject to poorly quantifiable uncertainties. The penetrating defect provides a resistance to the release of radionuclides to the buffer that will evolve over time due to internal processes within the canister. A model simplification is, however, adopted whereby the transport resistance of the defect remains constant until a specified time, after which it is entirely lost.

Radionuclides are transported through the buffer predominantly by diffusion. Equilibrium, linear sorption[25] is assumed on buffer pore surfaces. Buffer pore surfaces, being negatively charged, repel anions. Anion concentrations in narrow pores and near to pore surfaces in larger pores are therefore less than in the case of non-anions, for given concentrations at the boundaries. This "anion exclusion" effect is treated in transport modelling by assigning the matrix a lower porosity and a lower effective diffusion coefficient when modelling anion transport compared to the values for non-anions. Transport-relevant properties are homogeneous throughout the buffer. In the base case, the presence of the corroded supercontainer is assumed not to perturb these properties. This source of uncertainty is covered by cases addressing processes originating at the buffer-rock interface in the Radionuclide Transport Report /Smith et al. 2007b/. Although sorption on iron corrosion products or on the iron itself will probably take place, because of uncertainties it cannot be relied upon quantitatively in the safety assessment.

If, due to radioactive ingrowth during transport, near-field solubility limits are reached within the buffer, then precipitation of the migrating element will occur, maintaining the dissolved concentration at the solubility limit. For transport modelling purposes, however, due to limitations of the REPCOM code, solubility limits are applied only inside the canister and at the buffer/rock interface.

5.4.3 The approach to geosphere modelling

In reality, geosphere transport takes place in a network of fractures with significant variability in their flow and transport properties. The highly simplified geosphere transport modelling carried out in the present safety assessment, however, considers a single, representative geosphere fracture that intersects a deposition drift near to the location of a failed canister.

Transport is retarded by matrix diffusion and, for many dissolved species, sorption on matrix pore surfaces. The fracture has a width W [m], length L [m] and flow rate Q [m^3/y]. FTRANS input parameters are chosen in such a way as to give the required value for the lumped parameter WL/Q, which, as discussed in Section 11.5 of /Vieno and Nordman 1999/, represents the "transport resistance" of the geosphere.

Using FTRANS, matrix porosity in the wall rock adjacent to the fracture can be subdivided into different sub-domains, each with different transport properties. This is used to differentiate between mineralogically altered wallrock immediately adjacent to the fracture, and more distant, unaltered wallrock.

The modelling approach and parameter values used are based largely on TILA-99, although more recent developments in the understanding of the Olkiluoto site, and, in particular, discrete fracture network modelling carried out in support of the KBS-3H safety studies (Lanyon and Marschall 2006), are used to provide additional support for the parameter values selected (for example, in terms of their conservatism).

5.4.4 Comparison with TILA-99 and SR-Can

The approach to geosphere modelling in the KBS-3H safety studies is identical to that used in TILA-99 and also similar to that used in SR-Can, although different computer codes are used. Both the KBS-3H safety studies and TILA-99 use the FTRANS geosphere transport code.

[25] The amount of any element sorbed on buffer pore surfaces is assumed to adjust rapidly to changes in aqueous concentration and to be proportional to this aqueous concentration at any time, i.e. for transport modelling, equilibrium linear sorption is assumed, quantified by an element-dependent sorption constant (K_d). The assumption of equilibrium linear sorption entails a simplification of relatively complex sorption processes. The assumption of linearity is, however, usually met at the low concentrations that are of interest and the assumption of equilibrium is met if the sorption has a timescale that is much shorter than the timescale for slow diffusive transport across the buffer.

SR-Can uses the code FARF31 but the processes modelled and geometrical simplifications made are largely the same. Both the KBS-3H safety studies and TILA-99 differentiate between more porous altered wallrock adjacent to fractures, and the unaltered wallrock further from the fractures while SR-Can treats the wallrock as a homogeneous matrix. Longitudinal dispersion can be included in geosphere models using both FTRANS and FARF31. This process was, however, omitted in the KBS-3H safety studies and the majority of cases analysed in TILA-99, although it was included in SR-Can. In SR-Can uncertainties in the longitudinal dispersion have not been analysed. Instead, a Peclet number of 10 was used in the safety assessment (p. 154 in the Data Report, /SKB 2006e/). With a higher Peclet number, the dispersion has a lesser impact on the results than that shown in SR-97 /SKB 1999/ where a Peclet number of 2 was used for some of the cases. Calculations carried out in support of TILA-99 show that the impact of longitudinal dispersion on geosphere release is minor.

A major difference was that, in TILA-99 and SR-Can, detailed flow modelling was carried out to support the selection of flow-related input parameters for geosphere transport modelling. In KBS-3H, although discrete fracture network modelling was carried out for the rock immediately around a deposition drift, this was not used to support geosphere transport parameter selection. Rather, WL/Q was set to a reasonable value based on experience from TILA-99. The TILA-99 value of 50,000 years per metre assigned to the geosphere transport resistance parameter WL/Q in the base case is based, in the first instance, on statistical data for hydraulic conditions at the Olkiluoto site /Löfman 1996/. The conservatism of this choice is, however, supported by the more recent discrete fracture network modelling of the Olkiluoto site carried out by /Lanyon and Marschall 2006/.

SR-Can, like the KBS-3H safety studies and TILA-99, used a compartment model for near-field release and transport. TILA-99 used the same code – REPCOM – as used in the KBS-3H safety studies but in the case of TILA-99 only a 1-dimensional model was implemented compared to the 2-dimensional model for the KBS-3H calculations. The code used in SR-Can was COMPULINK, which is based on COMP32 /Vahlund and Hermansson 2006/. There are, however, some differences in the near-field modelling approach adopted in the three assessments, as described below.

Regarding the treatment of the partitioning of initial radionuclide inventory, it is assumed in both the KBS-3H safety studies and in TILA-99 that the inventory of activation products in metallic components, such as the fuel cladding, is released congruently with the corrosion of the metal. A more pessimistic approach is taken in SR-Can, where no credit is taken for the delay due to the limited rate of metal corrosion. In SR-Can, the radionuclide inventories in these components are included in the instant release fraction.

Figure 5-3 compares the TILA-99 model geometry for a KBS-3V-type near field (essentially the same as that in SR-Can) with the model geometry for a KBS-3H near field. The figure is taken from /Nordman and Vieno 2003/, in which a first study was undertaken applying REPCOM to the KBS-3H geometry.

In the TILA-99 safety assessment, two groups of cases were analysed: (i), a disappearing canister and (ii), a penetrating defect (hole) through the canister wall. In the disappearing canister cases, there were three radionuclide escape routes from the near-field model domain into the geosphere:

- From the buffer around the canister into the rock fissures intersecting the deposition hole (Q_F in Figure 5-3).
- From the backfill in the top of the deposition hole into the excavation damaged zone (EDZ) below the tunnel floor (Q_{DZ}).
- From the tunnel into the rock or EDZ (Q_{TDZ}).

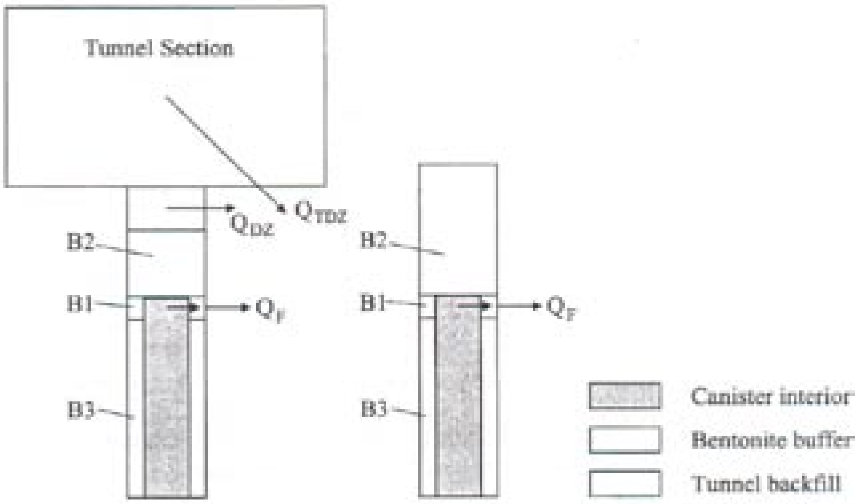

Figure 5-3. *Near-field model geometry for TILA-99 concept (left) and KBS-3H (right). B1, B2 and B3 denote different parts of the buffer. (After Figure 3-1 in /Nordman and Vieno 2003/).*

The releases via the three routes were summed to yield the total release rate from the near field to the geosphere. Only the first of these routes was, however, considered when modelling cases involving a canister with a penetrating defect. The model was thus comparable in terms of geometry and escape route to the KBS-3H safety assessment near-field model. SR-Can also considered three escape routes from the near field into the geosphere, essentially equivalent to Q_F, Q_{DZ} and Q_{TDZ} shown in Figure 5-3. Unlike TILA-99, however, all three routes were considered when dealing with cases involving a canister with a penetrating defect.

In TILA-99, only a hemispherical volume of buffer centred on the canister hole and with a radius of 35 cm (the thickness of the TILA-99 concept buffer between the canister and rock) was taken into consideration when modelling transport through the buffer. The modelled buffer volume therefore comprised less than 1% of the total buffer volume in a deposition hole. This conservative approach was adopted to simplify modelling; only one-dimensional radial diffusion from the defect to the rock was modelled. In the KBS-3H safety studies, two-dimensional matrix diffusion is modelled, including radial diffusion from the hole towards the rock and longitudinal diffusion parallel to the drift axis. Cylindrical geometry is used in the calculations with the conceptual model set up as described in /Nordman and Vieno 2003/.

In the TILA-99 cases involving a canister with a penetrating defect, radionuclide transfer from the outer surface of the hemispherical buffer volume to the rock was modelled by means of the transfer coefficient Q_F. Q_F was evaluated based on the conservative assumption also used in the KBS-3H safety studies that all groundwater flow in the rock around the deposition hole took place in a single fissure adjacent to the hole in the canister. As described in Section 11.6 of /Vieno and Nordman 1999/, the total transfer coefficient form the buffer to the rock was given by:

$$Q_F = \frac{Q_{cf} Q_{bl}}{Q_{cf} + Q_{bl}} \qquad \text{(Eq. 5-1)}$$

where Q_{cf} [m^3/s] is an equivalent flow rate from the surface of the canister into the mouth of the fracture and Q_{bl} [m^3/s] is the ground water flow rate in a thin boundary layer of water in the fracture adjacent to the buffer/rock interface into which mass transfer from the buffer occurs.

In the KBS-3H safety studies, it is similarly assumed that there is only one escape route from the near-field model, represented with the transfer coefficient Q_F. However, since Q_{cf}, is modelled explicitly by REPCOM, Q_F is set equal to Q_{bl} to avoid "double counting" of the mass transport resistance provided by the buffer between the canister and the fracture wall.

In SR-Can, Q_F (termed Q_{eq1}, the equivalent flow rate at a deposition hole, in SR-Can Data Report, Section 6.6 SKB 2006e) was evaluated using two alternative groundwater flow models – a continuum model and a discrete fracture network model. In addition to the several other differences between the two designs, the formulae used for Q_F differed according to the type of groundwater flow model. In the case of the discrete fracture network model, there could in principle be multiple fractures intersecting a single deposition hole in some model realisations. A conservative simplification was, however, made whereby the flow rates of all fractures intersecting the deposition hole were assigned to a single fracture. As in the KBS-3H safety studies and in TILA-99, this fracture was placed on the opposite side of the buffer to the canister defect, hence minimising the transport distance and diffusional transport resistance. The resulting formula for Q_F used in SR-Can is equivalent to that used the KBS-3H safety studies.

5.5 Data in the KBS-3H safety assessment

5.5.1 General approach to data selection

The general approach in the safety assessment for the KBS-3H safety assessment has been to use, where appropriate, the near-field databases from SR-Can and the Olkiluoto geosphere data from TILA-99. Data in SR-Can are mostly specified in terms of distributions and, in general, data for the KBS-3H safety assessment is taken from either the centre or the pessimistic end of these distributions. This being the case, an exhaustive comparison of databases between the three assessments is not very illuminating. However, there are some changes in the data used in the KBS-3H safety assessment compared to the original databases and it is of interest to discuss the reasons for and implications of those changes.

At the outset, it must be recognised that the TILA-99 data were based on different types of waters, pH and redox ranges, as compared to the reference groundwater in equilibrium with bentonite used in the KBS-3H assessment, and that this may affect all the solubility data to some extent but, especially, the elements which are pH and redox dependent, e.g. many of the actinides. The pH range for solubility data in TILA-99 was 7–10 but the conditions varied from very reducing to very oxidising (see Table 2-4 in /Vuorinen et al. 1998/). Also, the thermodynamic data used in TILA-99 represented the state-of-the-art thermodynamic database almost 10 years ago and, for many of the elements, there have been several improvements in the data in that time, such as for Pu and Th, as discussed below. For example, data concerning Pu(III), which are relevant in the reducing conditions under study, have recently been improved in the NEA thermodynamic database.

As part of the future development of the Posiva Safety Case Portfolio for Olkiluoto, it is intended to document the derivation of the data used throughout the quantitative safety assessment. However, a project decision was made not to prepare a separate data report at the present time (in contrast to SR-Can; /SKB 2006e/) and all data used in the KBS-3H safety studies are reported in Appendix A of the KBS-3H Process Report /Gribi et al. 2007/, except for the solubility and transport data (sorption and diffusion parameters) which are presented in the Radionuclide Transport Report (/Smith et al. 2007b/; solubility data in Appendix E). The bases for data selection and assumptions used have been reported as much as possible in those appendices. A more complete data report for both KBS-3H and KBS-3V designs in Olkiluoto will be published at a later date.

5.5.2 Near-field data

In the KBS-3H safety assessment, the time taken for water to contact the fuel/metal parts and for transport pathways to be established, and the dissolution rate of the fuel thereafter, are taken directly from SR-Can (in the case of the fuel dissolution rate, the peak of a triangular distribution recommended for use in SR-Can by /Werme et al. 2004/ is used). Fractional corrosion rates for the zircaloy and other metal parts, which, as noted above, are not used in SR-Can, are taken from TILA-99 /p. 101 of Vieno and Nordman 1999/. The spent fuel characteristics and inventory data are taken from /Anttila 2005/ and the activity inventory assumes a period of 30 years cooling ex reactor.

i) Solubility limits

Solubility limits for the KBS-3H near field have been estimated by /Grivé et al. 2007/ for a range of groundwater types. Values are given in Table 5-9 for the base case assumption of dilute brackish groundwater in equilibrium with bentonite.

The solubility limits that vary markedly between TILA-99 and the KBS-3H safety assessment are identified in Table 5-8 and the consequences of the more significant of these changes are discussed below. It should be noted that the solubility data for TILA-99 represents conservative values for non-saline reducing conditions (Table 11-2 in /Vieno and Nordman 1999/) so some differences would be expected given the more realistic emphasis in the KBS-3H assessment.

Ni

The significantly higher solubility limit applied to Ni requires some comment as it is some 40 times higher than the TILA-99 conservative value and higher even than the "very conservative value" (1 x 10^{-3} M).

Table 5-9. KBS-3H base case solubility limits. Blue shading indicates a solubility limit >10 × higher than in TILA-99 (conservative solubility values for non-saline reducing conditions, Table 11-2 in /Vieno and Nordman 1999/), yellow shading indicates a solubility limit > 10 × lower than TILA-99.

Element	Solubility (mol / dm)	Solubility limiting phase
Am	4.0×10^{-7}	$Am(CO_3)_2Na \cdot 5H_2O$
C	High [1]	–
Cl	High	–
Cm	4.0×10^{-7}	Based on analogy with Am
Cs	High	–
I	High	–
Nb	3.8×10^{-5}	Nb_2O_5
Mo [2]	2.6×10^{-8}	MoO_2
Ni	4.3×10^{-3}	$Ni(OH)_2$
Np	1.1×10^{-9}	$NpO_2 \cdot 2H_2O(am)$
Pa	3.0×10^{-7}	Pa_2O_5
Pd	2.5×10^{-6}	$Pd(OH)_2$
Pu	1.1×10^{-6}	$Pu(OH)_4(am)$
Ra	2.2×10^{-8}	$RaSO_4$
Se	3.1×10^{-10}	$FeSe_2$
Sm [3]	7.5×10^{-8}	$SmOHCO_3$
Sn	1.2×10^{-7}	SnO_2 (am)
Sr [3]	9.1×10^{-5}	Celestite $SrSO_4$
Tc	4.2×10^{-9}	$TcO_2 \cdot 1{,}6H_2O$
Th	6.3×10^{-9}	$ThO_2 \cdot 2H_2O$
U	9.5×10^{-10}	$UO_2 \cdot 2H_2O$
Zr	1.7×10^{-8}	$Zr(OH)_4$ (aged)

[1] "High" indicates that no solubility limit is applied in radionuclide release and transport calculations

[2] Not included in CC and RS cases due to short half-life

[3] Not included in release calculations due to short half-lives and no in-growth

In TILA-99, the solubility-limiting solid was not specified and the estimated solubility was largely based on measured concentrations at the investigation sites in Finland and a limited literature review. For Opalinus clay, $NiCO_3$(cr) was the solubility-limiting phase resulting in a solubility of about 10^{-5} M /Nagra 2002a/ but it was noted that the solubility-limiting phase would change[26] to $Ni(OH)_2$ under conditions of low partial pressure of CO_2. /Grivé et al. 2007/ basically used the same Nagra-PSI TDB /Hummel et al. 2002/ but updated it in accordance with the latest thermodynamic data published within the NEA-TDB project on Se, Ni and Zr (Ni data from /Gamsjäger et al. 2005/). On this basis, /Grivé et al. 2007/ selected the nickel carbonate, hellyerite, which contains 6 H_2O, as a possible solid for Ni. As a consequence, however, the solubility limiting solid phase under the Olkiluoto conditions is not the carbonate but the hydroxide. In addition, changes to the stability constant for the species $NiCO_3$(aq), which was the dominant Ni species in SR-97 /SKB 1999/, meant that in the Olkiluoto groundwaters the main aqueous Ni species is not $NiCO_3$(aq) but Ni^{2+}. As a result, the main solubility limiting reaction becomes:

$$Ni(OH)_{2(s)} + 2\ H^+ = Ni^{2+} + 2\ H_2O \qquad \text{(Eq. 5-2)}$$

which has $\log K_{sp} = 11.03$. At a pH= 7 to 8 as in Olkiluoto conditions, this implies a Ni solubility of 10^{-3} to 10^{-4} M justifying a higher solubility limit for use in the base case calculations.

However, the results obtained with this higher solubility limit are actually very little changed from the TILA-99 results (cf. Figures 5-4 and 5-6), presumably as the reduced conservatism of the near-field model in the KBS-3H safety studies offsets the solubility increase, as other transport parameters (K_d and D_a) are very similar.

Nb

The solubility limit for Nb has been reduced from the conservative and rather pessimistic 1 x 10^{-3} M in TILA-99 to a value much closer to other recent safety assessments such as Project Opalinus /Nagra 2002ab/. The Nb concentration and speciation is strongly dependent on pH, thus the speciation is also affected by the uncertainties in the pH range for the groundwaters considered. The solubility is much higher at the high pH (pH 10), which was considered as the upper range of pH in TILA-99 for the non-saline groundwater /Vuorinen et al. 1998/. This is one major factor resulting in a higher solubility in TILA-99 than for the base case in the present study (with dilute brackish groundwater in equilibrium with bentonite, pH 7.4). It is also acknowledged that the thermodynamic database for niobium is restricted. With this reduction to the Nb solubility limit, the Nb-94 releases from the geosphere in the PD-BC are very similar to the TILA-99 SH-sal50 case (cf. Figures 5-4 and 5-6).

Pd

The solubility limit for Pd has been increased by a factor of 250 from the TILA-99 conservative value of 1 x 10^{-8} M which has the effect of increasing the importance of Pd-107 releases from the geosphere in the KBS-3H safety assessment results (Figure 5-4) compared to TILA-99 (Figure 5-6). The higher solubility in KBS-3H safety assessment is at least partly due to the different solubility-limiting solid, amorphous $Pd(OH)_2$ as compared to the oxide PdO used in TILA-99, and also the different thermodynamic data used in the two assessments.

Th and U

The solubility limits of both U and Th have been revised downwards by about 2 orders of magnitude compared to the TILA-99 conservative values (3 x 10^{-7} M and 5 x 10^{-7} M, respectively)

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[26] Specifically, if $\log(pCO_2)$ was less than –3.5.

which makes Th, in particular, significantly less soluble than in other recent assessments including SR-Can, where the range is 1 x 10^{-7} to 4 x 10^{-5} M, and Project Opalinus, which used 7 x 10^{-7} M. The change to the thorium solubility limit reflects the work by /Grivé et al. 2007/ in updating the solubility database for the KBS-3H: a more consistent and coherent selection of Th data was made than that of /Duro et al. 2006/ for SR-Can. Also, recently published data for aqueous hydroxides and solid has been taken into account, including data for Th aqueous hydroxides from /Altmaier et al. 2005/, where previous values from /Neck and Kim 2001/ have been checked and slightly improved.

For U, the calculated solubility limit is much closer to other recent safety assessments such as Project Opalinus /Nagra 2002ab/. As for Th, there are still uncertainties associated with the speciation of uranium which may affect the results, e.g. hydroxides and the stabilisation of $UO_2(CO_3)_3^{4-}$, in the negative redox regime especially at higher pH. Uranium solubility is very sensitive to redox conditions, which also might be reflected in the different results for the groundwaters in TILA-99 and in this study.

Cm

In the present study, curium solubility is based on the chemical analogy with americium. For most TDBs and compilations, Am and Cm are considered equivalent and data for the two elements may be used interchangeably /Grivé et al. 2007/. Curium solubility has been revised upwards by one order of magnitude compared with the TILA-99 conservative solubility values for non-saline reducing conditions (Table 11-2 in /Vieno and Nordman 1999/). This may be due to the different thermodynamic data, the solubility-limiting solid ($CmOHCO_3$ in TILA-99) and lack of carbonate complexes for curium in the case of TILA-99. However, the solubility of curium is higher at the high pH (pH 10), which was considered as the upper range of pH in TILA-99 for the non-saline groundwater /Vuorinen et al. 1998/, and this is also a major factor in the differences in solubilities between TILA-99 and the present safety assessment.

Sm

Samarium solubility has been revised downwards more than two orders of magnitude compared with the TILA-99 conservative solubility values for non-saline reducing conditions (Table 11-2 in /Vieno and Nordman 1999/). This may be due to the different thermodynamic data and the solubility-limiting solid ($Sm_2(CO_3)_3$ in TILA-99). Additional factors are the more realistic approach taken in KBS-3H safety studies, as well as the different groundwater conditions. In the fresh, reducing groundwater conditions in TILA-99, phosphate complexes are important at high pH (pH 10), increasing the solubility /Vuorinen et al. 1998/.

Se

The solubility of selenium has been revised downwards by 4 orders of magnitude compared with the TILA-99 conservative solubility values for non-saline reducing conditions (Table 11-2 in /Vieno and Nordman 1999/). In addition to the more realistic approach taken in the KBS-3H safety studies, the change in the solubility limit reflects the changes in the thermodynamic data as well as the different groundwater conditions. There are uncertainties related to co-precipitation effects and the definition of the redox state of the system. Selenium solubility depends very strongly on Eh which, in TILA-99 groundwaters, ranged from about –250 to –410 mV (vs. SHE), where the lowest Eh value gives the highest solubility. The effect of lower Eh is also seen in KBS-3H solubility calculations as the solubility for Se is highest for the lowest Eh around –500 mV (vs. SHE), corresponding to a pH_2 of 100 atm, which takes into account the anaerobic oxidation of iron into magnetite.

ii) The selection of radionuclides present as anions and as non-anions

In TILA-99, C, Cl, Se, Pd, Sn and I were assumed to appear as anions in all cases. In the KBS-3H assessment, I, Cl, Se and Mo are treated as anionic when assigning porosities and effective diffusion coefficients in the buffer, with remaining elements being treated as neutral and cationic complexes. Mo was not considered as one of the safety-relevant elements in TILA-99. Pd is assumed to be dominated by neutral complexes based on speciation calculations /Grivé et al. 2007/. Other nuclides are present at least partly in anionic form, depending on the groundwater composition, pH and redox conditions. Speciation also changes depending on the oxidation state, which can also vary according to groundwater composition. For example, U(VI) is dominant in carbonate-rich waters even with reducing conditions, due to the stability of the carbonate complexes formed, whereas in low-carbonate groundwaters, the U(IV) will become dominant under similar redox conditions.

Speciation calculations indicate that Th and Nb will also be present as anionic complexes, that Pu will be present in anionic and non-anionic forms in roughly equal amounts and that U will be present, at least in part, in anionic form as U(VI). Furthermore, some C may be present in anionic form as carbonate complexes, as well as in the form of neutrally charged methane and organic acids. For consistency with SR-Can, however, Nb, Th, Pu, U and C are treated as being entirely neutral or cationic in the near field in the base case. The assumption of neutral or cationic form is clearly conservative in cases where there is either no difference in sorption or sorption of the anionic form is lower than that of the neutral or cationic forms. This is the case for C, since all C species are non-sorbing in the buffer according to SR-Can (Table A-13 of /SKB 2006b/). For other elements assumed to be present in neutral or cationic form, the conservatism or otherwise of this assumption is not immediately apparent from the data.

For Th, the speciation is still uncertain but there are limited data for the anionic hydroxy-carbonate Th complexes formed, which dominate in the waters studied. However, the anionic speciation is in conflict with the well-known sorption properties of Th on various materials, including both clays and fracture minerals in the rock. Thus, due to the limitations and uncertainties with the thorium speciation, Th is treated as non-anionic species in the present study. Th is also treated as neutral species in SR-Can and in TILA-99.

For Nb, in addition to the lack of thermodynamic data, there is a lack of Nb sorption data for bentonite. However, based on available data, a K_d value for sorption of niobium on bentonite has been defined /Ochs and Talerico 2004/, which indicates that niobium is either neutral or cationic. Due to the limitations and uncertainties, discussed above, with respect to the solubility limits and the fact that Nb sorbs on bentonite, Nb is treated as neutral species in the base case. The recognised possibility of anionic speciation of Nb in the near field and the far field is considered in a variant case (PD-BCN).

For Sn, the aqueous speciation is dominated by the hydrolysis complexes of Sn(IV), $Sn(OH)_4$, with some contribution of the anionic species $Sn(OH)_5^-$ in some of the groundwaters.

In the PD-BC, carbon is assumed to be predominantly in methanic form, which is assumed not to sorb. It is possible that microbial oxidation of methane could take place along with sulphate reduction if brackish, sulphate-rich water reaches repository depth during the future evolution of the Olkiluoto site, leading to the formation of carbonate, which would be expected to sorb weakly (with a K_d of 0.0001, according to TILA-99). This possibility is considered in the variant cases PD-BCC, PD-VVERC and PD-EPRC (see Table 5-2). Assuming that all carbon is present as neutral species, such as methane or organic acids (for which K_d is set to zero in the calculations), is believed to be a conservative approach as a $K_d = 0$ implies a rapid transport of the organic species through the buffer and geosphere into the biosphere. Thus, in the KBS-3H safety assessment, Cl, Se, Mo and I are also assumed to be present as anions in the near field in the base case.

iii) Buffer transport properties

Buffer porosities, effective and apparent diffusion coefficients used in the KBS-3H assessment are based mainly on the recommended values given in Table A-11 of the SR-Can Data Report for all groundwater types /SKB 2006e/. Thus buffer porosities and effective diffusion coefficients for the base case are taken to be:

- 0.43 and 1.2×10^{-10} m^2/s for neutral and cationic species.
- 0.17 and 1.0×10^{-11} m^2/s for anions.
- 0.43 and 3.0×10^{-10} m^2/s for the particular case of Cs.

The apparent diffusion coefficient, which appears in the diffusion equation (Eq. 5-3), is related to the effective diffusion coefficient D_e [m^2/s], porosity ε and the sorption constant K_d [m^3/kg] using:

$$D_a = \frac{D_e}{\varepsilon + (1-\varepsilon)\rho_s K_d} \qquad \text{(Eq. 5-3)}$$

where ρ_s is the mineral density of the bentonite buffer, taken to be 2,700 kg/m^3.

Buffer sorption coefficients (K_d values) are given in Table 5-10, based conservatively on the lower limit values given in Table A-12 of the SR-Can Data Report for saline and non-saline groundwaters /SKB 2006e/.

Sorption can vary significantly with speciation/oxidation state. The oxidation states assumed in the KBS-3H safety assessment are based on speciation calculations by /Grivé et al. 2007/. The calculations show that U, Pu and Np, in particular, may be present in the buffer in more than one oxidation state. Np(IV) dominates in the case of Np and is the assumed oxidation state. In the case of Pu, Pu(III) dominates in all waters at neutral pH and reducing conditions determined by the large amounts of hydrogen and iron present, except for glacial meltwater (ice melting water) with a higher pH, where Pu(IV) dominates.

Table 5-10. Buffer sorption coefficients (K_d values).

Element	K_d [m^3 / kg]	Element	K_d [m^3 / kg]
Am	10	Pd	0.3
C	0	Pu	4
Cl	0	Ra	0.001
Cm	10	Th	6
Cs	0.018	Se	0
I	0	Sm	0.8
Mo	0	Sn	2.3
Ni	0.03	Sr	0.0009
Nb	0.2	Tc	2.3
Np	4	U	0.5
Pa	0.2	Zr	0.1

According to Table A-12 of the SR-Can Data Report /SKB 2006e/, Pu(IV), with a lower limit K_d of 4, is less sorbing than Pu(III), which has a lower limit K_d of 10. So, conservatively Pu(IV) is the assumed Pu oxidation state for the transport calculations. For U, U(IV) and U(VI) species can both be present in significant proportions under the expected redox conditions in the near field of the KBS 3H repository at Olkiluoto. However, the lower end of the range of K_d values for U(VI) in SR-Can is smaller by a factor of about 7 than that for the U(IV) species so, conservatively, it is the value of 0.5 m^3/kg for the former which is assumed to apply in the KBS-3H safety assessment.

There are still some issues related to the K_d values of radionuclides varying with changing conditions that may need to be examined with further modelling studies. For example, a radionuclide with a high K_d can accumulate in the geosphere and be released when conditions (e.g. redox conditions) change. This is an issue for further work for both KBS-3H and -3V.

5.5.3 The near field/geosphere interface

As described in Section 5.4.2, a drift section containing a canister with an initial penetrating defect is assumed to be intersected by a single transmissive fracture. This fracture is assigned a transmissivity of 3×10^{-9} m^2/s, which is the highest transmissivity in a drift section that, in the current design, would be acceptable for the emplacement of supercontainers and distance blocks. This criterion is discussed in Appendix B of /Smith et al. 2007a/. It is derived primarily from considerations of the role of the geosphere as a transport barrier. Based on previous safety assessments, it ensures that the host rock provides an effective barrier to the transport of radionuclides released in the event of canister failure. It also protects the buffer against piping and erosion. In the current reference design, piping and erosion can be excluded if the maximum initial inflow rate of groundwater into a drift section containing a supercontainer and distance block is about 0.1 litres per minute or less (Appendix L of /Autio et al. 2007/). This corresponds to a single fracture with a maximum transmissivity of about 3×10^{-9} m^2/s, assuming the applicability of Darcy's law in a radial configuration (Thiem's equation). The actual correspondence of a fracture transmissivity of 3×10^{-9} m^2/s and an inflow of 0.1 litres per minute, however, needs to be further investigated, since repository excavation may have a perturbing influence on the hydrostatic pressure around the drift (hence on inflow). It is assumed that water flow in the fracture is driven by a regional hydraulic gradient of 0.01 /Löfman 1999/.

The selection of criteria for acceptable transmissivity values, which are criteria for the design, is still ongoing. The flow rate affects the rate at which radionuclides are transferred from the buffer to the flowing groundwater. The transfer rate is quantified in terms of a transfer coefficient (or equivalent flow rate), which, in the base case for an initial penetrating defect in the KBS-3H safety assessment, is about 1.4 litres per year /Smith et al. 2007b/. As noted in Section 5.4.2, the transmissivity of the intersection, and hence transfer coefficient, are considered to be pessimistic values, at the high end of the expected range. Considering the preliminary information about the Olkiluoto site, it is estimated that over 80% of the 10 m sections (corresponding roughly to a drift section containing one supercontainer and one distance block) considered have a drift inflow below 0.1 litres per minute (Figure 15 in /Hellä et al. 2006/).

As described in Section 5.4.2, radionuclides are transferred by diffusion from the buffer to the nearest fracture intersecting the drift (Figure 5-2). Advective transport at the buffer/drift boundary occurs only at the fracture/buffer intersection. Here, radionuclides enter a thin diffusion "boundary layer" from where they are advected downstream.

Provided the elemental concentrations at the outer boundary of the buffer are less than the corresponding solubility at the buffer/rock interface, then concentrations across the boundary are assumed to be continuous. Radionuclides are released to the geosphere at a rate $C.Q_{bl}$, where C is the radionuclide concentration at the boundary and Q_{bl} is a transfer coefficient, as defined above (Section 5.4.2). If, on the other hand, the solubility of a given element is exceeded, the combined rate of release of all isotopes of a given element is limited by $C_s.Q_{bl}$, where C_s is the elemental solubility.

The solubilities applied at the buffer/rock interface are the near-field solubilities given in Table 5-9, which are derived for buffer porewater. In reality, the water at the interface is likely to have an uncertain transitional composition between that of buffer porewater and the groundwater. However, use of geosphere solubilities, which could be lower for some elements due to the different composition of groundwater, would give rise to the issue of apparent precipitation at the boundary.

As noted above, it is a limitation of the REPCOM code that solubilities are applied inside the canister and at the buffer/rock interface but not internally within the buffer. If, in reality, solubilities were to be exceeded at the buffer/rock interface then, in principle, colloids could form by precipitation and be transported by advection in the geosphere by water flowing through the fracture. Any apparent precipitation at the interface due to near-field solubilities being exceeded is, however, likely to be an artefact of the REPCOM approach. There are a few cases in which radioactive ingrowth during transport across the buffer can lead to solubilities above the limits at the buffer/rock interface – the buffer is essentially saturated with U at the solubility limit, and so anything that decays to a U isotope during transport across the buffer will lead to some precipitation and hence immobilisation of U at the buffer/rock interface if solubility limits are applied there.

In reality, colloids are more likely to form inside the buffer but would be immobile because of the fine buffer pore structure. Colloids could form at some location in the geosphere near to the buffer/rock interface in those cases where far-field solubilities are lower than near-field solubilities. If transported, however, such colloids would be likely to redissolve, as concentrations in solution fall as a result of dilution, and would not significantly affect the overall radionuclide transport times in the geosphere. Thus, colloid formation at the interface and transport in the geosphere is not included in the calculations.

5.5.4 Geosphere data

Parameter values for matrix diffusion and matrix pores are shown in Table 5-11, for transport resistance of the geosphere in Table 5-12 and for sorption in Table 5-13. These values are taken for the most part from TILA-99, using data for reducing and, where relevant, non-saline conditions representing fresh groundwater conditions in the rock (TDS < 1g/L). The TILA-99 conservative values, rather than realistic values, are used since the conservative values were used to evaluate the TILA-99 reference scenarios (Chapter 11 in /Vieno and Nordman 1999/).

Table 5-11. Geosphere matrix porosity and effective diffusion coefficients (from Table 11-10 in /Vieno and Nordman 1999/, assuming non-saline groundwater). In the present study Cl, Se, Mo, and I are assumed to be present as anions (see main text).

Parameter	Distance from fracture	Species	Value
Porosity (%)	0–1 cm	Anions	0.1
		Non-anions	0.5
	1–10 cm	Anions	0.02
		Non-anions	0.1
Effective diffusion coefficient (m^2/s)	0–1 cm	Anions	10^{-14}
		Non-anions	10^{-13}
	1–10 cm	Anions	10^{-15}
		Non-anions	10^{-14}

Table 5-12. Geosphere parameter values that apply to all migrating species in the base case for an initial penetrating defect.

Parameter	Unit	Value	Source
Transport resistance of geosphere (*WL*/*Q*[1])	y/m	50,000	Median value for both saline and non-saline conditions at Olkiluoto; Table 11-19 in TILA-99 /Vieno and Nordman 1999/. The mean transport resistance is likely around 10^6 to 10^7 but with a few pathways down to 50,000 y/m /Lanyon and Marschall 2006/
Rock matrix grain density	kg/m^3	2,700	/p. 119 in Vieno and Nordman 1999/
Maximum rock matrix penetration depth	m	0.1	/p. 119 in Vieno and Nordman 1999/
Fracture aperture	m	3×10^{-5}	See Table 5-8

[1] *W* [m] is the width of the flow channel, *L* [m] is the transport distance and *Q* [m^3/y] is the flow rate in the channel.

Table 5-13. Geosphere sorption coefficients (Kd values) (from Table 11-9 in /Vieno and Nordman 1999/, conservative values, assuming reducing conditions and non-saline groundwater).

Element	K_d [m^3 /kg]	Element	K_d [m^3 /kg]
Am	0.04	Pd	0.001
C	0	Pu	0.5
Cl	0	Ra	0.2
Cm	0.04	Th	0.2
Cs	0.05	Se	0.0005
I	0	Sm	0.02
Mo	0.0005	Sn	0.001
Ni	0.1	Sr	0.005
Nb	0.02	Tc	0.05
Np	0.2	U	0.1
Pa	0.05	Zr	0.2

There are, however, some exceptions to the use of TILA-99 data. These are the K_d values for C and Mo. As noted above, C is assumed to be predominantly in methanic form and is assumed not to sorb, although variant cases PD-BCC, PD-VVERC and PD-EPRC (Table 5-2) test the influence of C in the form of carbonate (weakly sorbing with a K_d of 0.0001, according to TILA-99).

There are no data available in the literature on which to base a K_d value for Mo in crystalline rocks. The value given in Table 5-11 has been chosen by expert judgement, based on a study of sorption on illite by /Motta and Miranda 1989/, a comparison of the cation exchange capacities (CECs) of illite and rock, and the known pH-dependency of K_d values for sorption on kaolinite.

The transport resistance of the geosphere in the base case for an initial penetrating defect (and for most of the assessment cases analysed in the KBS-3H safety assessment) is taken from TILA-99 (Table 5-12). In particular, the value WL/Q = 5×10^4 y/m is the TILA-99 median value, based on statistical data for hydraulic conditions for both saline and non-saline groundwater at Olkiluoto (Table 11-19 in /Vieno and Nordman 1999/). This value is likely to be much higher in light of the more recent site description information.

The rock matrix immediately adjacent to the fractures may be mineralogically altered, as reflected in Table 5-11 by the higher porosities and higher effective diffusion coefficients in the first centimetre adjacent to the fracture wall. In the present study, as in TILA-99, lower higher porosities and lower effective diffusion coefficients were applied from one to ten centimetres from the wall. It was assumed that the rock matrix further than ten centimetres from the fracture wall is inaccessible to migrating radionuclides.

5.5.5 Biosphere

The KBS-3H Radionuclide Transport Report /Smith et al. 2007b/ uses the WELL scenario (representing an indicative stylised drinking water well) to derive a safety indicator: the "WELL-2007 dose" arising for an individual of the most exposed group living at the Olkiluoto site. The WELL-2007 dose is the result of a conversion of release rates from the geosphere to the biosphere using the updated WELL-2007 dose conversion factors (DCFs); these DCFs are the same as the WELL-97 DCFs used in TILA-99, except for Rn-222 for which a different dose coefficient has been used.

The WELL scenario is, in principle, independent of biosphere properties, since the water is drawn directly from the geosphere-biosphere interface zone.

The biosphere analysis in the KBS-3H project is based on landscape modelling (a coupled eco-system model, similar to the approach in SR-Can). The radiological impact on the biosphere (i.e. doses to humans and effects to other biota) is calculated from the radionuclide transport in the biosphere from a set of scenarios, selected after consideration of biosphere processes and evolution. Biosphere analysis results are reported in /Broed et al. 2007/, and supported by a landscape configuration report /Broed 2007/.

In these reports, Landscape Dose Factors (LDFs) have been calculated in order to compare to the SR-Can LDFs. However, the difference in the regulatory requirements leads to rather different approaches to derive the LDFs, as discussed in /Broed et al. 2007/. In addition to the landscape modelling, a new agricultural well scenario is introduced (AgriWELL-2007), also used as basis for deriving a safety indicator. AgriWELL-2007 is based on the same well properties as WELL-2007 but includes utilising the water for watering cattle and irrigating crops in addition to drinking water, which is the single uptake pathway for WELL-2007. New exposure pathways are consumption of contaminated meat, crops and animal products (milk, meat and eggs).

5.6 Results of assessment cases

A limited subset of the results from the cases considered in KBS-3H is given here, where it is of interest to compare these results with similar cases from SR-Can and TILA-99. The results on the radionuclide release calculations are given in full, with related discussion, in the Radionuclide Transport Report /Smith et al. 2007b/.

i) Initial penetrating defect failure mode base case (PD-BC) results

Figures 5-4 and 5-5 show the results for the initial penetrating defect base case (PD-BC) in the KBS-3H assessment in terms of geosphere release rates and annual individual dose based on the WELL-2007 DCFs, respectively. For TILA-99, the near-field and geosphere release rates for activation and fission products are shown in Figure 5-6 for the SH-sal50 case, i.e. small initial hole and saline groundwater (actinides are not shown as there are no releases > 1Bq/y from the geosphere over the 1 million year period). The corresponding dose rate for SH-sal50 is shown in Figure 5-7. Firstly, the obvious difference from the KBS-3H case is that there is no delay in the SH-sal50 case before radionuclides are released from the near field, and only 20 years before Cl-36 and I-129 are released from the geosphere, as this case did not take any benefit for the time before water pathways were established in the buffer to the canister defect. Secondly, the development of the initial defect in PD-BC, so that all transport resistance from the canister is assumed to be lost at 9,000 years, causes a peak in releases which has no equivalent in the SH-sal50 case, where the canister retains some transport resistance over time.

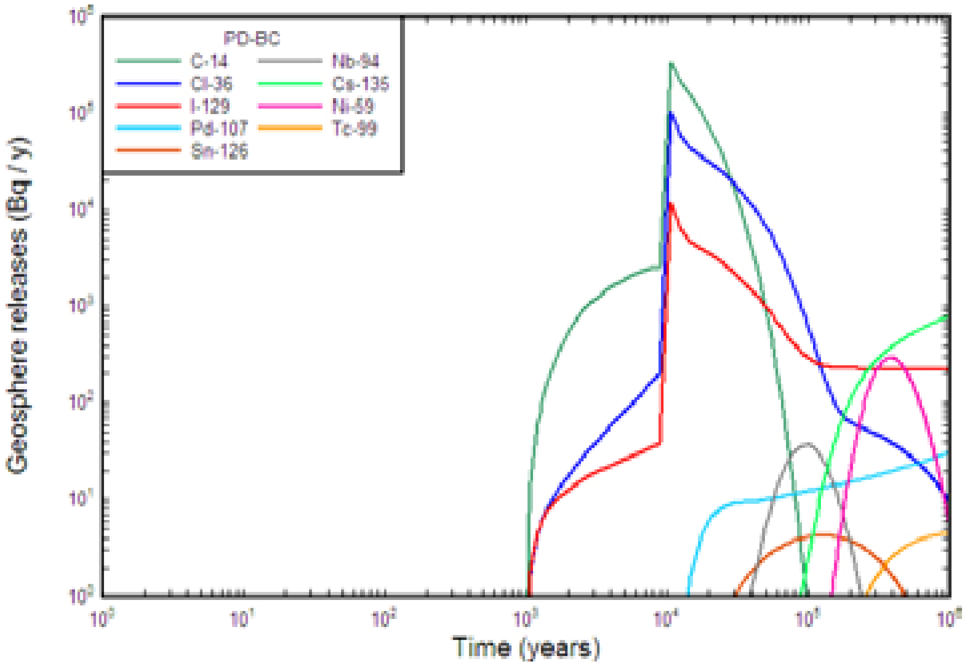

Figure 5-4. *Geosphere release rates for fission and activation products in the KBS-3H/ Olkiluoto initial penetrating defect base case (PD-BC). There are no actinide releases (>1 Bq/y).*

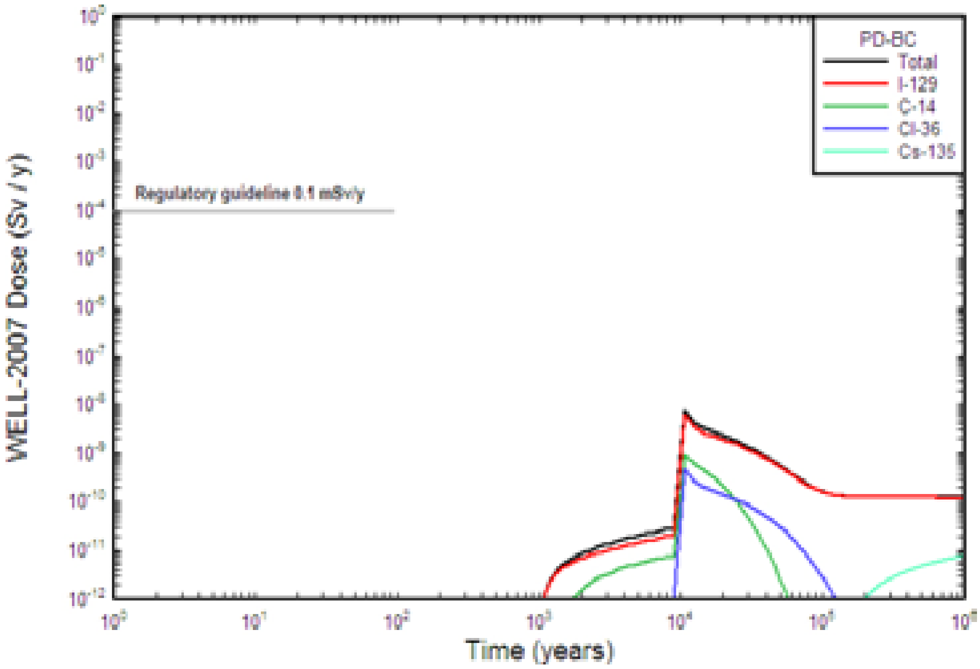

Figure 5-5. *Annual individual dose (using WELL-2007 DCFs) over time for the KBS-3H initial penetrating defect base case (PD-BC). The regulatory guideline of 0.1 mSv/y is shown for reference but is strictly applicable only for the period of predictable environmental conditions, which is taken as the first 10,000 years after repository closure (see the Radionuclide Transport Report for further discussion; /Smith et al. 2007b/).*

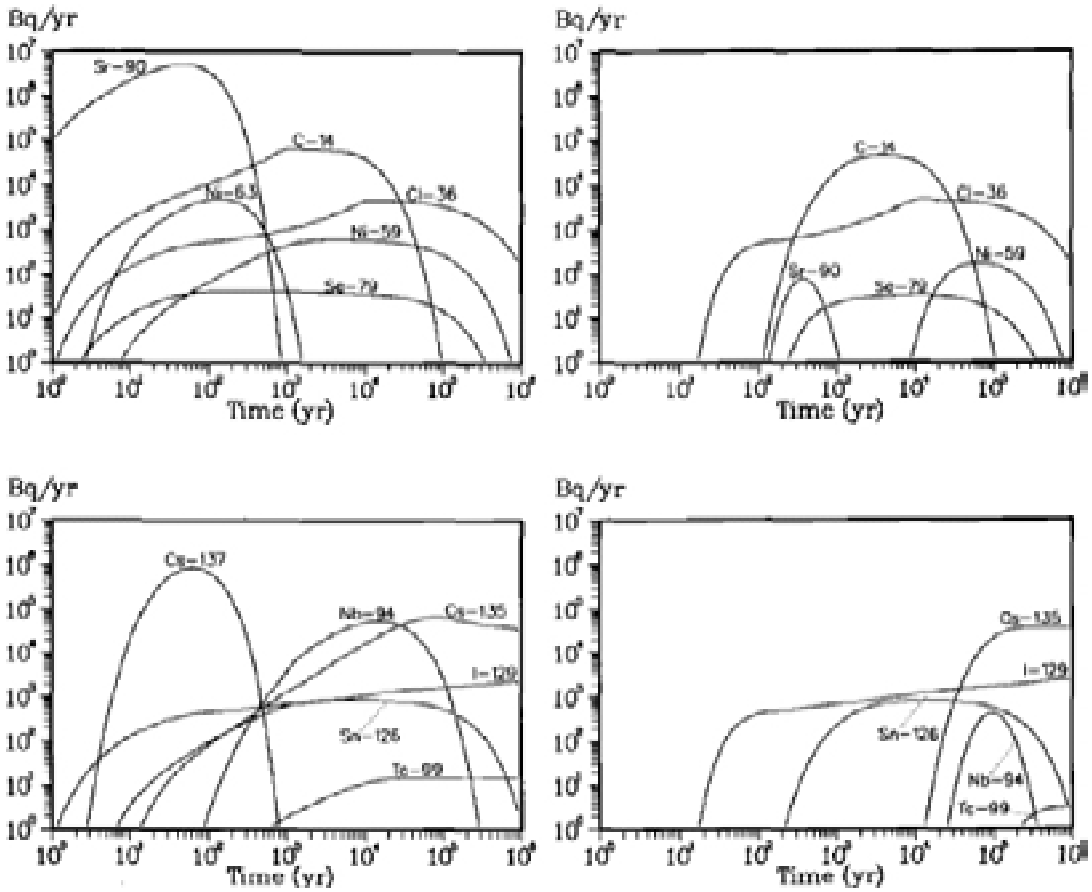

Figure 5-6. Releases rates for activation and fission products from the near field into the geosphere (left) and from the geosphere to the biosphere (right) for the TILA-99 SH-sal50 case (i.e. small hole, saline groundwater). There are no actinide releases (>1 Bq/y) from the geosphere within 1 My (actinide near-field releases are not shown).

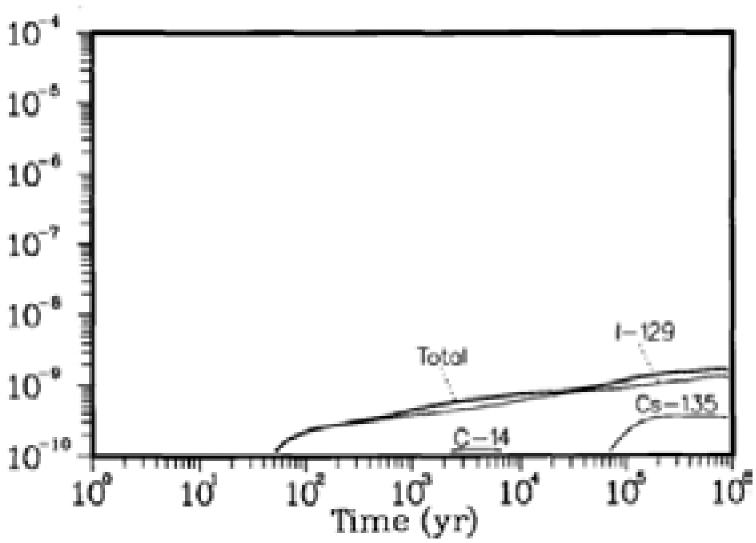

Figure 5-7. Dose rate (Sv/y) in the TILA-99 SH-sal50 case. /Vieno and Nordman 1999/. Note the truncated dose scale, compared to Figure 5-5.

The geosphere releases for selected nuclides and the annual effective dose for the SR-Can growing pinhole failure (PH) base case are shown in Figures 5-8 and 5-9, respectively. Note that Figure 5-8 shows the results of a deterministic calculation in which the peak (mode) values were used for all parameters with triangular or log-triangular distributions. The peak releases for the nuclides except Ni-59 are more than an order of magnitude lower than in the corresponding KBS-3H penetrating defect cases results. The release of Ra-226 is not shown in the suite of nuclides in Figure 5-8 because its releases are much higher than the scale used for the other radionuclides. However, Ra-226 which, in the base case, has high solubility and low sorption in the buffer, is the main contributor to the increased dose rate seen after 100,000 years in Figure 5-9, when I-129 releases are near constant.

This is shown in Figure 5-10 where the dose is decomposed to show the contributions from the dominant nuclides – I-129 and Ra-226. The dominance of Ra-226 at longer times in SR-Can is a result of the stochastic modelling of the flow paths in SR-Can compared to the KBS-3H and TILA-99 assessments. These releases of Ra-226 are dominated by contributions from the relatively small portion of deposition holes for which the geosphere transport properties lead to highly transmissive features, hence limited decay in the geosphere. Therefore, in SR-Can, the Ra-226 releases from the geosphere to the biosphere are dominated by the Ra-226 present in the canister and the decay of Th-230 in the buffer and geosphere plays a negligible role.

The KBS-3H and TILA-99 assessments use a single representative geosphere pathway and the equivalent contribution from Ra-226 is absent due to greater decay of the Ra-226 during longer transport times in the geosphere. Test calculations performed for the KBS-3H assessment, showed that the Ra-226 release to the biosphere is dominated by Th-230 in the geosphere. Only if the geosphere transport resistance is substantially reduced (WL/Q=5,000 instead than 50,000 as in the base case and especially in saline water because of the lower sorption coefficient of Ra-226), does Ra-226 reach the biosphere. Thus it is transport not the solubility of Th-230 that controls the Ra-226 releases in the KBS-3H Radionuclide Transport calculations /Smith et al. 2007b/.

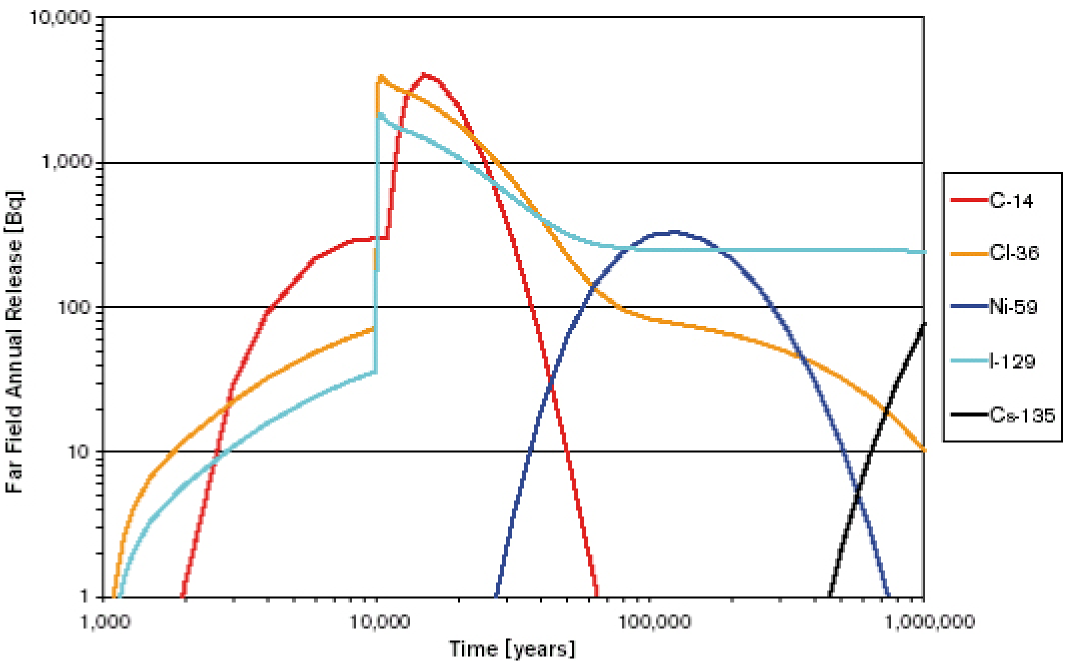

Figure 5-8. *SR-Can deterministic calculations of far-field releases resulting from near-field releases through Q1 (equivalent flow to Q1 from deposition hole 5x10*$^{-6}$ *m*^{3}*/y, transport resistance 4x10*$^{-6}$ *y/m), using the analytic models, for the pinhole failure mode (Data were taken as peak (mode) values for all parameters with triangular and log-triangular distributions). The release of Ra-226 is much higher than the scale shown but Ra-226 is taken into account in the overall dose calculation, see Figure 5-9 below. (Figure 10-15 of SR-Can Main Report, /SKB 2006a/).*

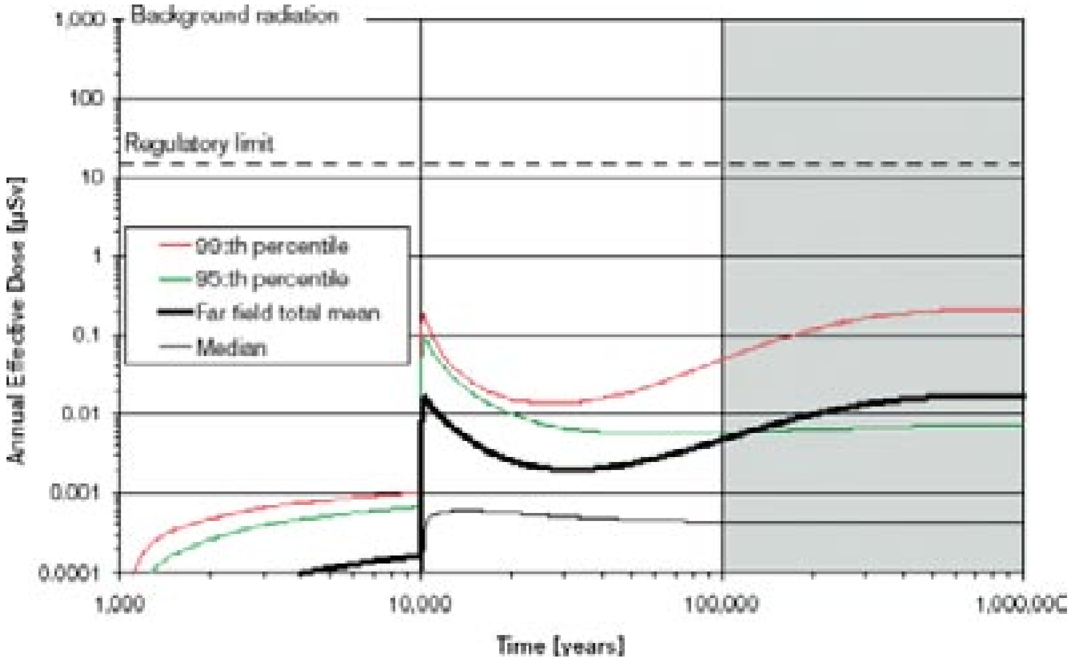

Figure 5-9. *Result of the probabilistic base case calculation of the SR-Can pinhole failure mode for Forsmark (fully correlated DFN model). The 1st and 5th percentiles are both zero since a fraction of the deposition holes are not connected to geosphere transport paths that reach the surface. (Figure 10-1, SR-Can Main Report, /SKB 2006a/).*

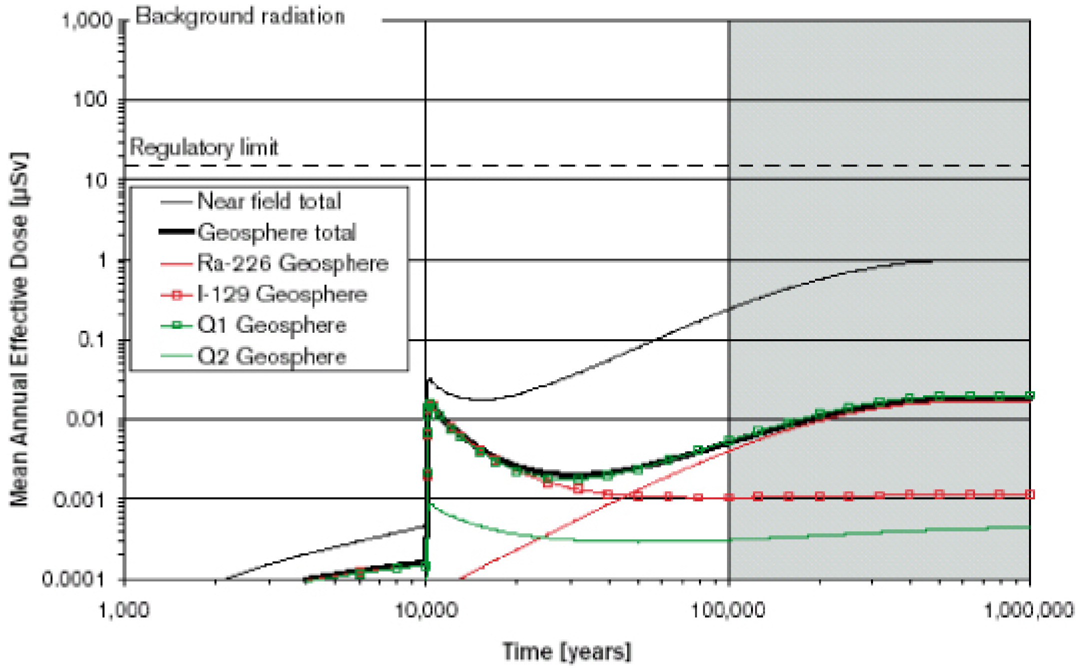

Figure 5-10. *The Forsmark pinhole failure base case decomposed with respect to dominant nuclides (Ra-226 and I-129) and release paths (Q1 and Q2). The effect of discarding geosphere retention is also shown (near field total, i.e. LDF applied to releases from the near field model). 10,000 realisations analytic model. (Figure 10-18, SR-Can Main Report, /SKB 2006a/).*

However, there are several uncertainties concerning the solubility of Ra-226 as the solubility-limiting solid is very much dependent on both the sulphate and carbonate content in the water and also on the possibility of co-precipitation processes with other elements (e.g. Ba or Ca) leading to reduced concentration of Ra /Grivé et al. 2007 and Vuorinen et al. 1998/.

There are conceptual uncertainties in the system due to the omission of the potential reduction of sulphate to sulphide /Grivé et al. 2007/. If this process occurred at the high hydrogen pressures indicated in the definition of the redox states (especially for the case of pH_2 100 atm), reduced sulphate would lead to much higher solubility of Ra as $RaCO_3$ would then be the solubility-limiting solid instead of $RaSO_4$, which is used in all reference waters for the radionuclide transport calculations /Smith et al. 2007b/.

On the other hand, co-precipitation with major elements in the water has not been taken into account; only pure solid phases have been considered in the calculations. Thus co-precipitation of Ra with e.g. gypsum or calcite is not considered, although this would lead to lower Ra concentrations than predicted from pure solid phases. In summary, the uncertainties in the speciation and model simplifications may lead to very different results between the KBS-3H radionuclide transport and release results /Smith et al. 2007b/ and other assessments (e.g. TILA-99 and SR-Can). This emphasises the care is required when comparing calculational case results from different safety assessment – even notionally similar cases can differ markedly in details which significantly affect the outcome.

The effect of changes to the transit times for radionuclides in the geosphere is clearly illustrated in the KBS-3H cases that use alternative values for the geosphere transport resistance. Figure 5-11 shows geosphere releases for the KBS-3H assessment PD case with a low geosphere transport resistance of 5,000 (PD-LOGEOR), compared to the PD base case (PD-BC) value of 50,000. In contrast, Figure 5-12 shows the PD case with a high geosphere transport resistance of 500,000 (PD-HIGEOR). No other parameters are changed from the base case.

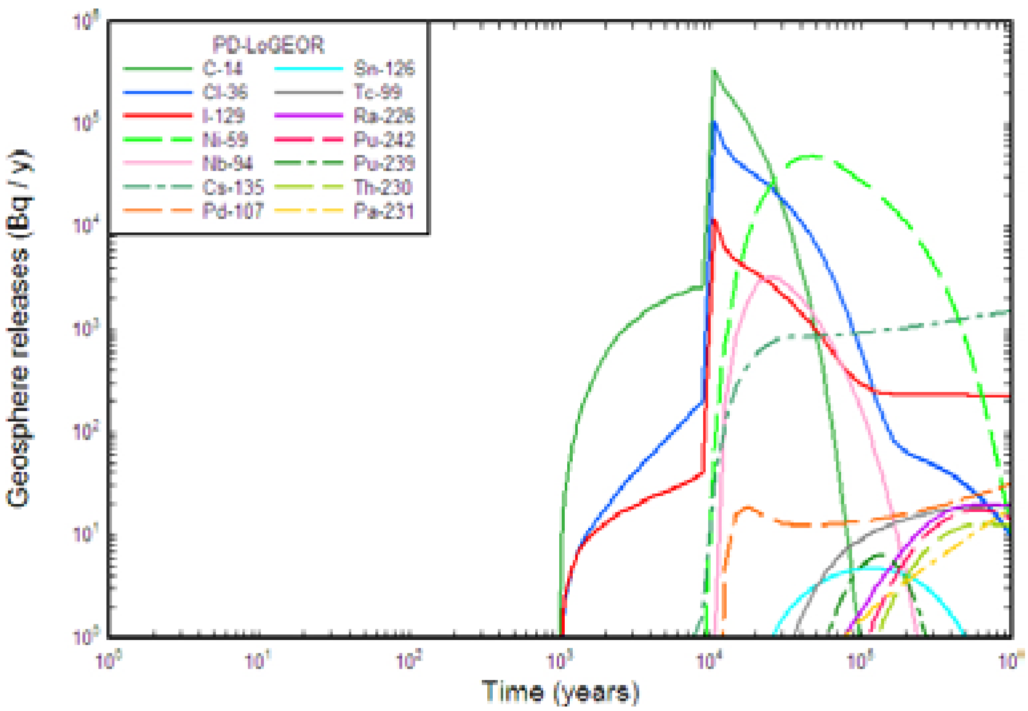

Figure 5-11. *Radionuclide releases from the geosphere in the KBS-3H assessment PD case assuming* ***low*** *geosphere transport resistance (PD-LOGEOR).*

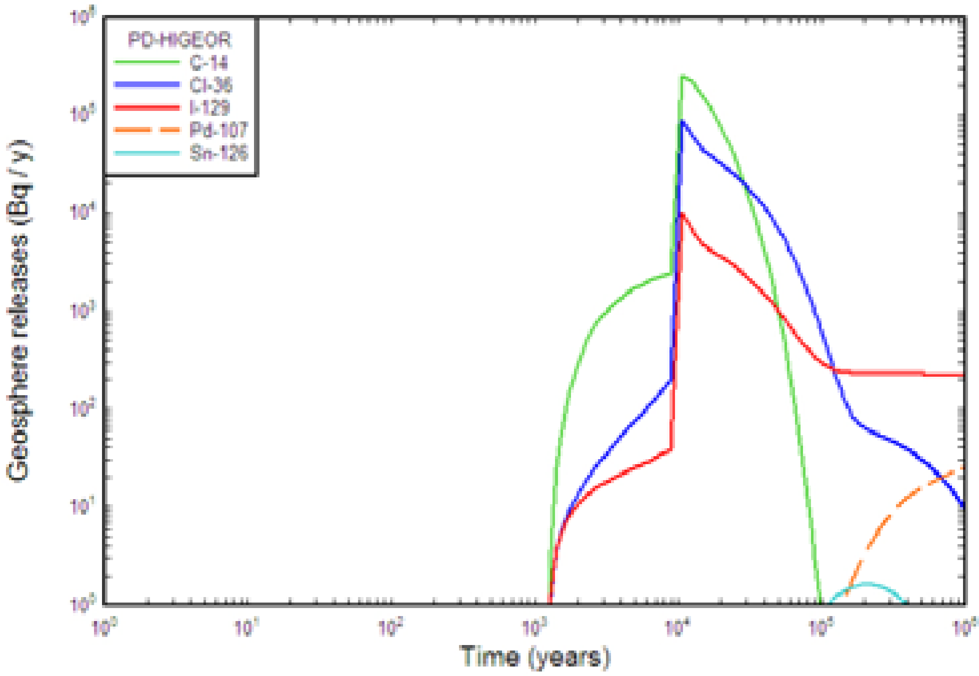

Figure 5-12. *Radionuclide releases from the geosphere in the KBS-3H assessment PD case assuming* ***high*** *geosphere transport resistance (PD-HIGEOR). Note the much diminished contribution from the actinides, including Ra-226.*

A comparison of these figures (and also Figure 5-5) clearly illustrates the effect of changing the geosphere transport resistance and thus slowing down the pulse of Th-230 and daughter Ra-226 leaving the near field. In the base case (Figure 5-5) and the high geosphere transport resistance case (Figure 5-12), the releases of Ra-226 are below the scale illustrated.

ii) Copper corrosion failure mode base case (CC-BC) results

The results of the copper corrosion base case (CC-BC) calculations for the KBS-3H safety assessment and SR-Can are shown in Figures 5-13 and 5-14, respectively. Again, a clear difference is the timing of releases in the SR-Can case, where the first of several, probabilistically determined, canister failures is close to 500,000 years compared to a single failure at 100,000 years in the KBS-3H case. Also, in SR-Can, the releases from the near field and the geosphere are totally dominated by Ra-226 as most of the Ra-226 released from the near field is transmitted through the geosphere because the failed canisters are located in deposition holes intersected by large, highly transmissive fractures with low retention.

This is a more pessimistic case than in the KBS-3H safety assessment where the PD-BC geosphere is used. However in this case, in order to calculate the amount of dissolved uranium and daughters released to the geosphere, the transfer coefficient from the canister interior to the geosphere, Q [m^3/y] is set equal to the groundwater flow rate through the eroded buffer. If the eroded buffer is assumed to have a hydraulic conductivity far greater than the rock and to "capture" the flow from a portion of the fracture that is twice the tunnel diameter, Q takes a value of 3.10×10^{-3} m^3/y compared to around 10^{-6} m^3/y at the hole in the canister in the PD-BC (See Table 4-8 in /Smith et al. 2007b/). This gives rise to the increasing dose at longer times in Figure 5-13 which is absent from the PD-BC results.

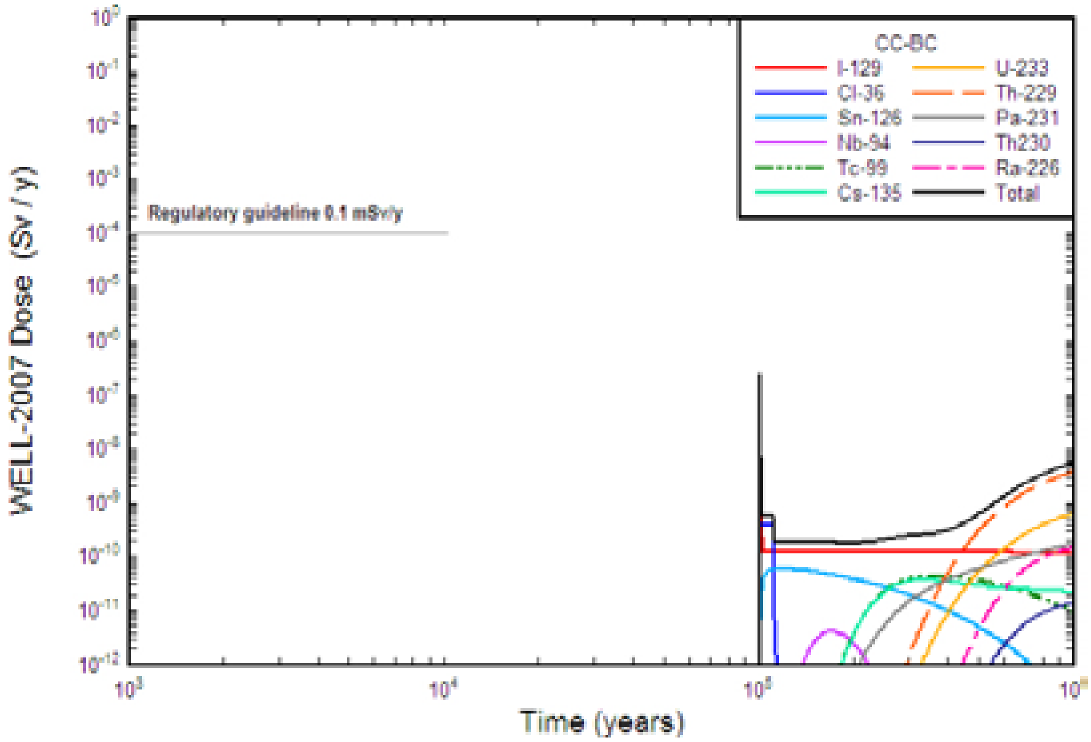

Figure 5-13. *KBS-3H assessment copper corrosion base case (CC-BC) dose (using WELL-2007 DCFs) over time. A single canister fails at 100,000 years.*

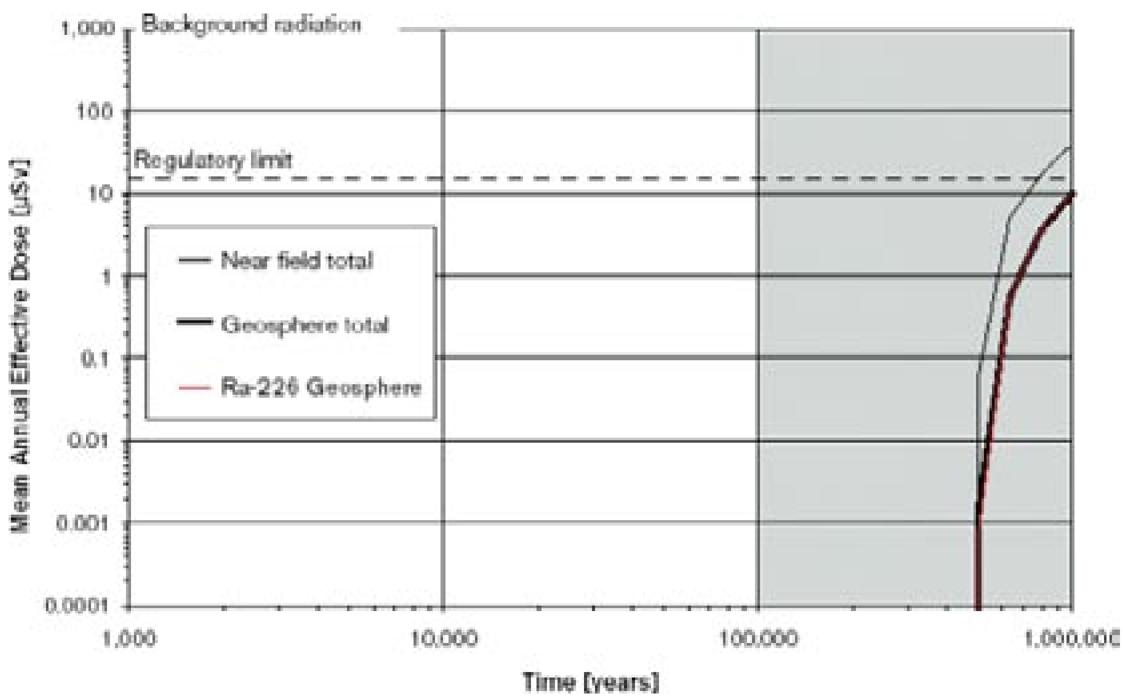

Figure 5-14. *SR-Can probabilistic calculation of near-field and geosphere annual effective doses for CC base case for the Forsmark semi-correlated hydrogeological DFN model. 10 canisters fail during the one million year assessment period; positions selected in accordance with the FPI criterion. (Figure 10-42, SR-Can Main Report, /SKB 2006a/).*

iii) Rock shear failure mode base case (RS-BC) results

The results of calculation of the failure due to rock shear base case (RS-BC) for KBS-3H and SR-Can are shown in Figures 5-15 and 5-16, respectively. In both cases a single canister is postulated to fail and both resultant dose curves are dominated by Ra-226 release which, in the case of the KBS-3H safety assessment (Figure 5-15), is great enough to exceed the dose peak due to the IRF at 70,000 years. Compared to PD-BC (Figure 5-5), the maximum dose is around one order of magnitude greater but arises at rather later times and persists for a much greater period of time. As with the SR-Can advection/canister corrosion case, the reduced transport resistance of the geosphere due to the assumption that the shearing event creates a much more conductive pathway to the biosphere leads to short transit times so that Ra-226 dominates the dose despite its short half-life.

In SR-Can, the peak dose is around 3 orders of magnitude higher than the pinhole (PH) base case. It is noteworthy that the consequences are very similar to those calculated deterministically for the advection/corrosion failure (which assumes just a single canister fails at 100,000 years). This means that if failure due to a shear movement is followed by buffer erosion, the consequences will not increase compared with the case where the buffer remains intact after the shearing.

iv) Results for other selected cases

The strong possibility of future changes to the groundwater composition in coastal sites, which may also be affected by glaciation, has been an issue for both Posiva and SKB so it is of interest to look briefly at the results of cases that address changing groundwater compositions (KBS-3H safety assessment, TILA-99) or also changes to the hydrology that could occur with glaciation (SR-Can).

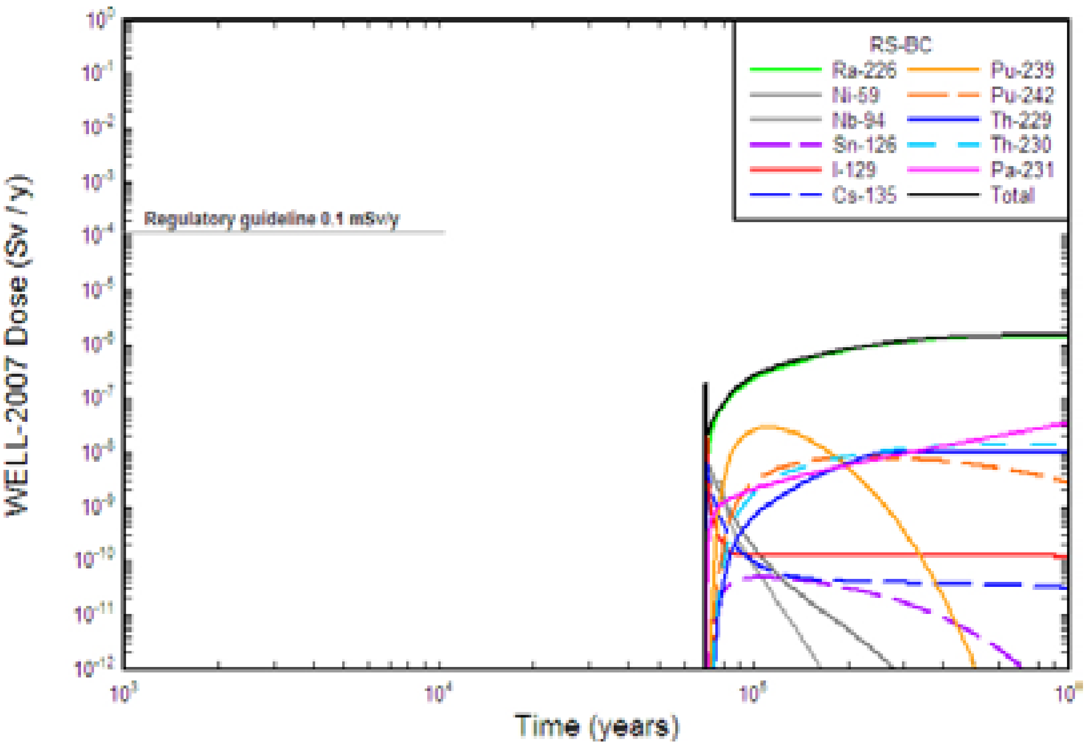

Figure 5-15. *KBS-3H rock shear base case (RS-BC) dose (using WELL-2007 DCFs) with time. The initial peak is due mainly to C-36 and I-129.*

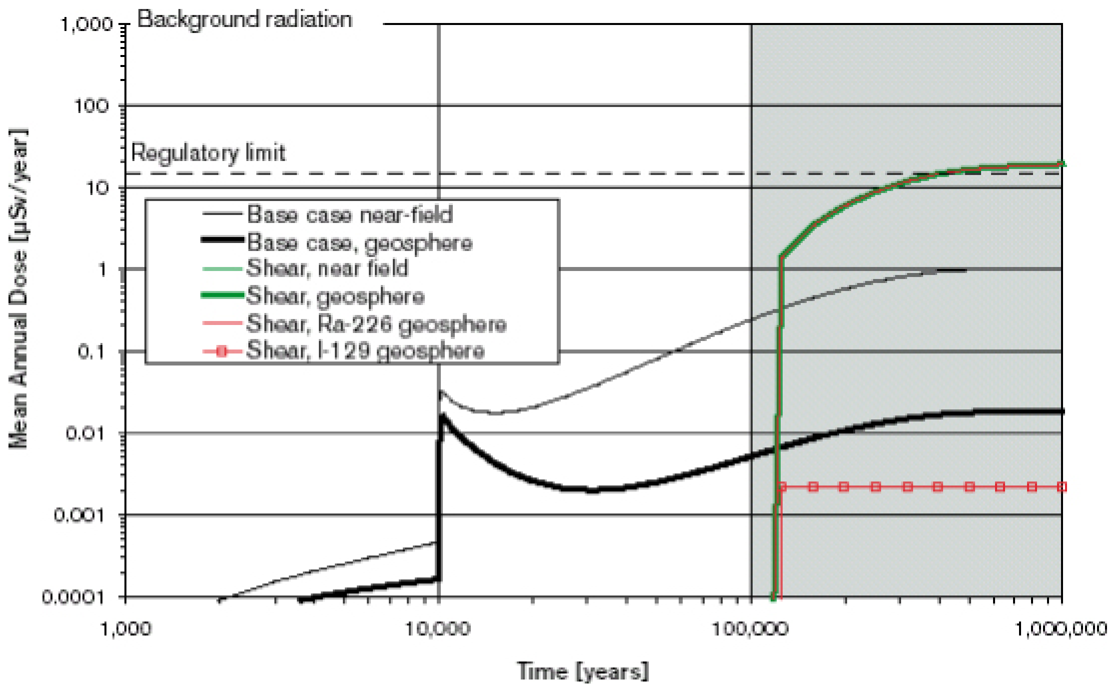

Figure 5-16. *Probabilistic results of a calculation postulating failure of one canister due to rock shear at 100,000 years in SR-Can. 10,000 realisations, analytic model. Note the curves for the PH (growing pinhole failure) base case are plotted for comparison ("Base case" curves). (Figure 10-50, SR-Can Main Report, /SKB 2006a/).*

Figure 5-17 shows the dose as a function of time for the KBS-3H PD-BC and the PD case with saline and highly saline groundwaters (PD-SAL and PD-HISAL). Despite the use of different elemental solubilities, sorption and diffusion coefficients, the resultant doses are almost unaffected. Only at longer times when nuclides which are solubility-limited or more strongly sorbing are finally released, do the curves diverge slightly. Figure 5-18 shows the results for the TILA-99 SH (small hole) and DC (disappearing canister) cases with both saline and non-saline conditions. The DC cases are somewhat similar in their results to the KBS-3H PD-BC in which the defect size increases at 10,000 years after which the canister is assumed to have no further transport resistance. In the SH case, there are small differences in dose between the saline and non-saline conditions. The dose for all 4 cases is dominated by I-129 to which salinity makes very little difference (except for apparent diffusivity) so the differences arise from sub-dominant nuclides like C-14 and Cl-36, and Cs-135 at longer times. But in both the KBS-3H safety assessment and TILA-99, it is clear that changing groundwater composition between low and high salinities is likely to have little significant impact on radionuclide transport overall.

The results for the KBS-3H case with glacial groundwater are shown in Figure 5-19 and the results for the SR-Can glacial climate cases in Figure 5-20. In the SR-Can cases, the altered climate affects the dose conversion factors (LDFs) because of the changed vegetation and land use as well as the groundwater composition. Thus the increased permafrost LDF for Ra-226 results in higher doses for this case, whereas the low LDFs for the ice margin case reduce doses significantly, even with increased geosphere flow.

In the KBS-3H assessment, for the PD case with glacial groundwater compositions (PD-GMW), I-129 is still the dominant nuclide but, due to the long period before any releases are considered, the contribution of C-14 has disappeared compared to PD-BC. The delay reflects the Weichselian-R scenario in which the next glacial retreat, thus the next possibility for penetration of glacial meltwater to repository depth and the most likely time for significant post-glacial earthquakes that could facilitate this, occurs in around 70,000 years time. For the assessment case, the delay before releases commence is taken to be 100,000 years (see Chapter 7 in /Smith et al. 2007a/).

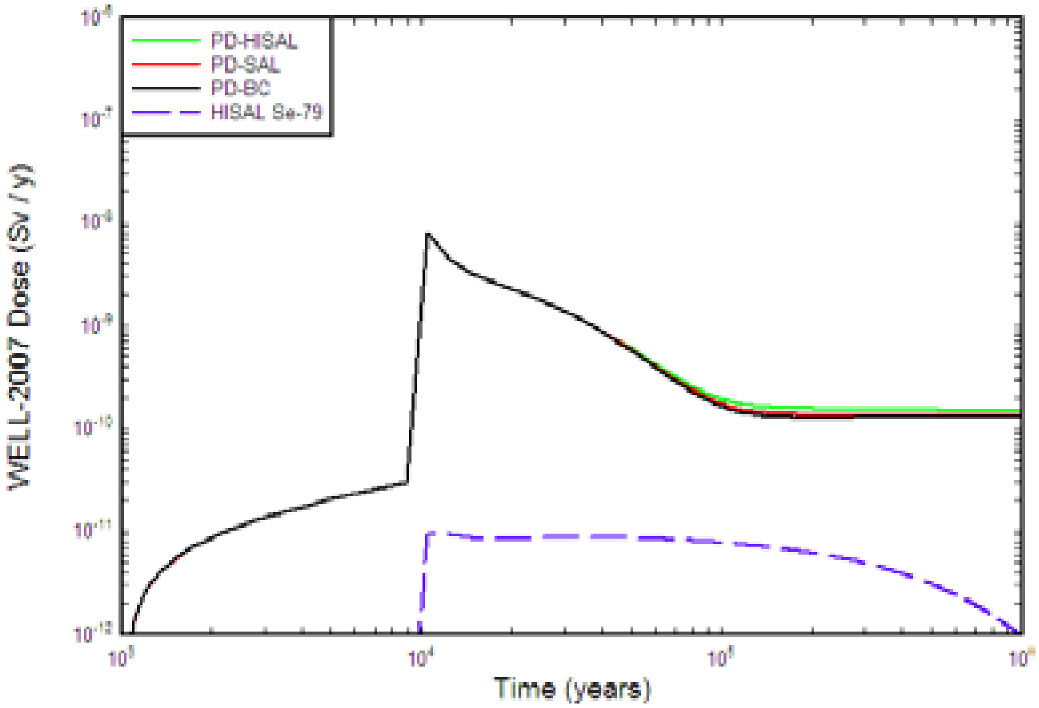

Figure 5-17. *Comparison of annual individual doses arising for the KBS-3H PD base case (PD-BC) and the PD cases with saline (PD-SAL) and highly saline groundwaters (PD-HISAL). Se is the element most affected by the increased salinity but still does not noticeably affect the total dose. Se is below the cut-off for the PD and PD-SAL cases. See the Radionuclide Transport Report /Smith et al. 2007b/ for further detail. (Note the expanded dose scale for this figure).*

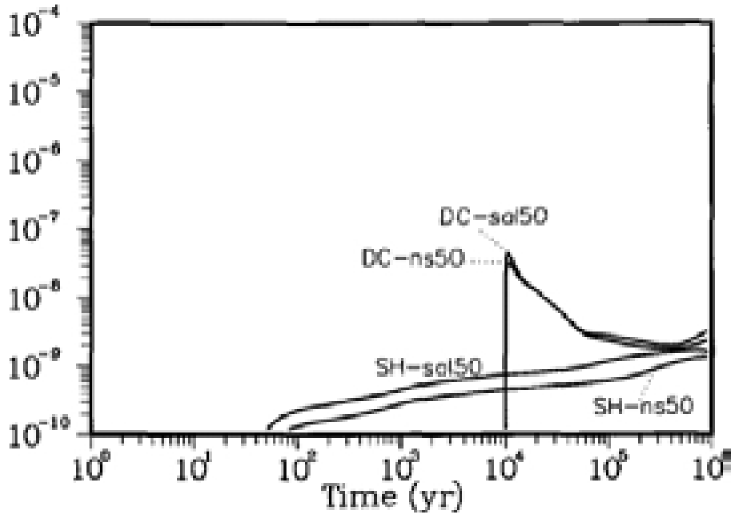

Figure 5-18. *Dose rate (Sv/y) for the TILA-99 small hole (SH) and disappearing canister (DC) cases for saline (sal50) and non-saline (ns50) conditions.*

The maximum dose and the contributions from the dominant nuclides – I-129, Cl-36 and Nb-94, with Cs-135 at longer times – are very similar between the cases suggesting that, as with the saline water variants, the chemistry of the groundwater has less effect than changes to the flow properties of the buffer and geosphere barriers.

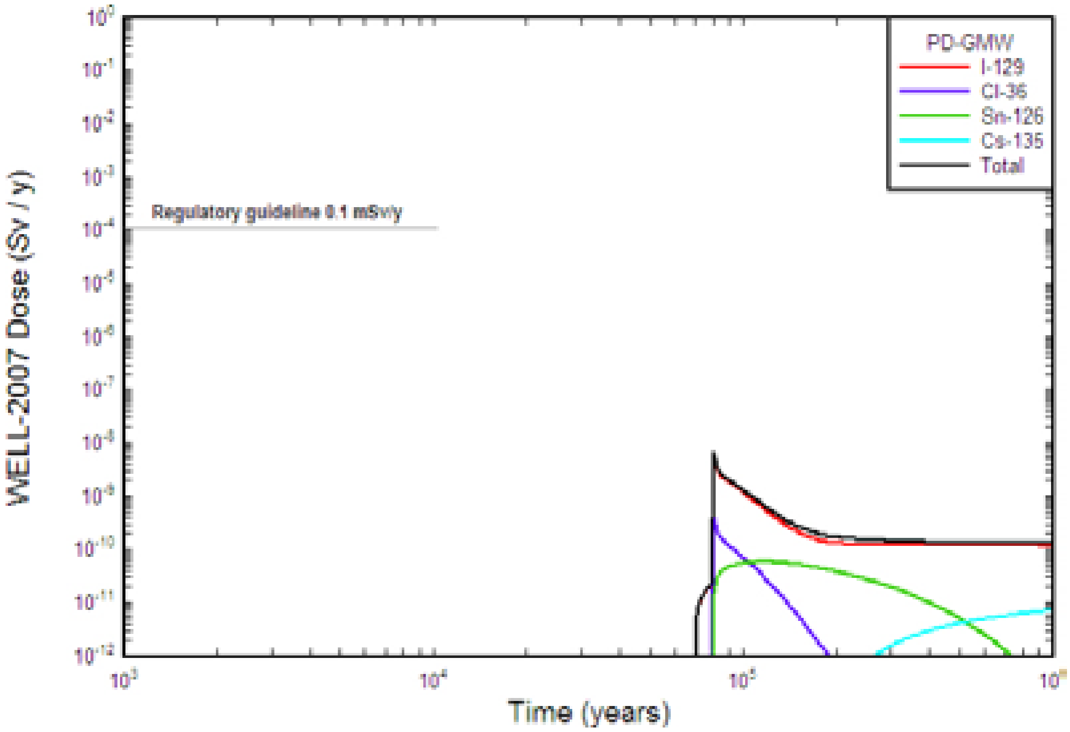

Figure 5-19. *Annual individual dose (using WELL-2007 DCFs) with time for the KBS-3H assessment PD case with glacial groundwater conditions (PD-GMW).*

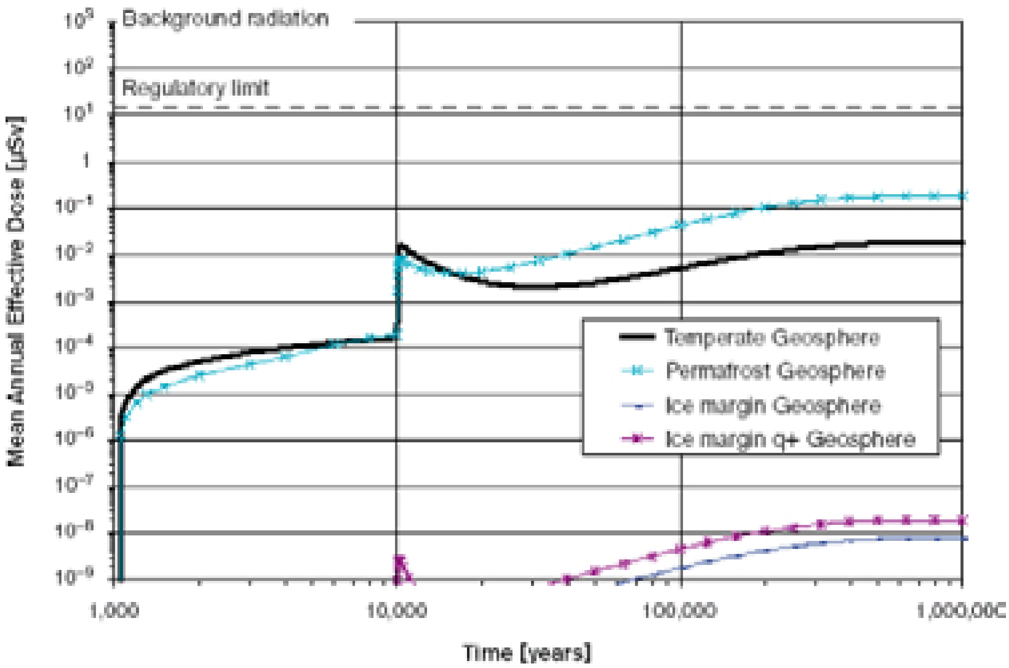

Figure 5-20. *Annual effective doses for the PH failure mode case and altered climate conditions at Forsmark in SR-Can. The glacial conditions affect LDFs, geochemistry of the groundwater and, in the "ice margin q+ geosphere" case, the flow through the geosphere is also increased (10,000 realisations, analytic model).*

5.7 The strategy to treat uncertainties

The safety assessment is built on the analysis of how a system with an initial state evolves as a result of actions on the system by a number of internal processes and external influences/events. From this description, various types of uncertainties can be identified relating to, for example, how well the initial state is known, how completely the internal and external processes are understood and how accurately they can be modelled because of shortcomings in data. In defining a structure for a rigorous approach to these issues, it is customary to categorise uncertainty into /NEA 1997/:

- System (or scenario) uncertainty which relates to difficulties in identifying a complete set of processes and interactions between them to fully describe the repository system and its evolution over long periods of time. Thus system uncertainty relates to completeness of the assessment.
- Conceptual (or model) uncertainty which relates to understanding of processes and the way in which models are used to represent a set of features, events and processes (FEPs) and interactions. Inevitably models must simplify the complexity of real systems, for example by use of 1 or 2-dimensional representations of 3-dimensional processes such as transport, thus introducing conceptual uncertainty.
- Parameter uncertainty which relates to the data used for parameters in the models. There are a number of aspects to take into account in the management of parameter uncertainty. These include correlations between data, the distinction between uncertainty due to lack of knowledge and that due to natural variability and situations where conceptual uncertainty is treated through a widened range of input data. The input data required by a particular model is in part a consequence of the conceptualisation of the modelled process, meaning that conceptual uncertainty and data uncertainty are to some extent intertwined. Also, there are several conceivable strategies for deriving input data. One possibility is to strive for pessimistic data in order to obtain an upper bound on consequences in compliance calculations; another option is the full implementation of a probabilistic assessment requiring input data in the form of probability distributions.

The Data Report produced for SR-Can /SKB 2006e/ describes in a systematic way how uncertainties are treated for each system component – spent fuel, canister, buffer and backfill – and geosphere data. For each component, these three types of uncertainty, as well as correlations among uncertainties, are discussed, with a final quantification of uncertainty based on expert judgment. As a decision was made not to produce a similar Data Report in the KBS-3H safety studies, no such discussion of uncertainties and quantification has been presented in a single document (see Section 5.5.1).

With respect to system uncertainties, philosophically, it is not possible to demonstrate that all relevant FEPs and interactions have been included in the assessment. However, the repository system, its initial state, and the processes and factors, both internal and external that could affect its evolution have been set out in Process /Gribi et al. 2007/ and Evolution /Smith et al. 2007a/ reports. These reports explicitly document the FEPs and their handling in the assessment. From these, it has been possible to make a comparison with the FEP databases developed in previous assessments and in other similar systems and assessments and also with more generic FEP databases published by international organisations such as the NEA /NEA 2000/. The role of formal FEP management is thus essentially to provide a checklist of all FEPs against which the treatment, including exclusion, can be logged to provide a traceable record. Quality assurance (QA) systems are then implemented to review for completeness and also ensure that FEPs are treated appropriately and adequately in the assessment.

The handling of conceptual uncertainty for processes is described in the Process Report /Gribi et al. 2007/. For each process relevant to KBS-3H, the knowledge base including remaining uncertainties is described and, based on that information, a method of handling the process in the safety assessment is established. Through the use of a defined format for all process descriptions, it is ensured that the processes and their associated conceptual uncertainties are described in a consistent manner. Conceptual uncertainty for external influences is handled in a more stylised manner, essentially through the definition of a sufficient set of cases. However, the restricted scope of the KBS-3H safety assessment means that consideration of external influences such as climate (other than changes to groundwater composition) or human activities is neglected.

Parameter uncertainties are treated by the definition of cases for sensitivity analyses such as, for example, fuel dissolution rates or redox conditions (that affect nuclide solubilities) or by bounding calculations to determine the best- and worst-case scenario to explore the robustness of the safety concept. The cases are described in the Radionuclide Transport Report /Smith et al. 2007b/ along with the parameters used.

The management of uncertainties continues to be discussed internationally with respect to deep geological disposal /IAEA 1997, NEA 1999b, 2002b, 2004ab/. Posiva and SKB are engaged in discussions within the international community, in particular through NEA activities, on the systematic treatment of uncertainties /NEA 2004b/.

Posiva is also currently participating to the PAMINA[27] project, under the auspices of the European Commission, to review international approaches to safety assessment methods and uncertainty management strategies.

In practice, the various common approaches to reducing uncertainties used by several national programmes (USA, Switzerland, France, Sweden, Japan) are outlined below /Chapman and McCombie 2003/:

1. Apply good science and continue well-chosen R&D activities throughout the repository development programme.
2. Use robust designs and analyses.
3. Aim for simplicity.
4. Use a structured approach including iterative assessments.
5. Use multiple lines of reasoning, a range of models and natural analogues.
6. Document the elicitation of expert judgment.
7. Perform quality assured analyses and have these peer reviewed.
8. Encourage international cooperation and evaluation.

The strategies of Posiva and SKB to manage uncertainties are consistent with the above principles, as described in /Vieno and Ikonen 2005/ and in SR-Can /SKB 2006a/. Posiva's strategy for the management of uncertainties has been recently updated in the Safety Case Plan 2008 /Posiva 2008/.

5.7.1 Process of expert elicitation

One of the strategies to handle the various types of uncertainty (scenarios, conceptual and parameters), is that of expert elicitation /NEA 2004b, 1999b/. Expert elicitation is needed in particular for:

- Conflicting data sources.
- Data collected using laboratory-scale experiments where the uncertainty is on a field scale.
- Unverified models or measuring procedures.
- Analogue chemicals and trace elements.
- Limited evidence.
- Data insufficiency to estimate internal uncertainty.

[27] PAMINA (Performance Assessment Methodologies in Application to Guide the Development of the Safety Case) started in October 2006 and will last three years.

The international community called for a formal process for using and documenting expert judgments in uncertainty management /NEA 2004a, 2004b/. Radioactive waste management programmes in France, Switzerland, UK and Sweden formally described their process for expert elicitation in their safety assessments /e.g. Andra 2005b, Nagra 2002b, NIREX 1998, SKB 2006e/. Posiva has implicitly elicited expert judgement for safety-case related documents in the past but without using a formal process. For example, the main reports and activity planning documents are reviewed by Posiva's safety committee, by the International Advisory Group for Olkiluoto Investigations (INAGO) and finally by its regulator, STUK. Furthermore, national and international technical peers routinely review reports (or report sections) from Posiva and SKB before publication. Formal reviews of documents by the regulator are carried out by national and international technical experts engaged by STUK. SKB follows similar practices and has a similar site investigation review team, the Site Investigations Expert Review Group (SIERG). Reviews of safety assessment documents by SKB's regulator SKI are also supported by external experts. SKB and Posiva also have a long-standing tradition of mutually reviewing drafts of reports prior to publication whenever possible. Posiva is currently examining ways to formalise the process of expert elicitation and to make it more transparent and traceable /Posiva 2008/.

5.7.2 Use of deterministic and probabilistic assessments

Internationally, two broad approaches are taken to the analysis of assessment cases in presence of significant uncertainties: deterministic and probabilistic. A deterministic approach is used in the KBS-3H safety assessment in the calculation of the consequences of repository evolution and radionuclide release. This means that assessment cases are defined and analysed with the models and parameter values for each case individually specified. The alternative is the probabilistic approach in which parameter values are sampled randomly from a pre-defined range of possible values distributed according to a probability distribution function (PDF). The two approaches are not mutually exclusive, however, and a safety assessment may use both approaches to treat uncertainties in different areas or use deterministic calculations within a mainly probabilistic approach to elucidate influences between parameters, as in SR-Can /SKB 2006a/. In a similar manner, probabilistic calculations were used in Project Opalinus /Nagra 2002a/ for sensitivity analyses in an otherwise deterministic assessment.

The deterministic approach is chosen because it can give a clear illustration of the impact of specific parameters and hence the impact of uncertainties in these parameters. A disadvantage of this approach, though, is that the response of models to parameter variations is not always smooth and readily understood. Moreover, the consequences of combinations of variation (both parameter uncertainty and natural variation) in several parameters simultaneously are not easily explored.

The main difficulty in the probabilistic approach is defining PDFs that quantify in single distributions widely different types of uncertainty (e.g. "aleatory" uncertainties related to variability or randomness and "epistemic" uncertainties arising, for example, where there is a range of plausible alternative models consistent with current scientific knowledge). Furthermore, the treatment of some uncertainties involves model assumptions that are hypothetical and highly conservative (e.g. the treatment of a perturbed buffer/rock interface as a highly conductive "mixing tank"), and it is unclear whether or not it is meaningful to assign a probability attached to such assumptions. A large number of calculations are run for a given calculational case so that the full extent of parameter space can be explored for that case. The outcome is then a profile of consequences and their associated frequencies – hazard and probability. Thus this approach is a natural tool for evaluating compliance with regulatory risk limits.

Though the deterministic approach separates the elements of risk by calculating consequences alone, the other element – probability or frequency – must still be evaluated in order to make decisions about, for example, the appropriate treatment of individual FEPs or combinations of FEPs in scenarios, those which can be excluded on the basis of low likelihood or frequency and those which must be included in deterministically evaluated assessment cases.

In the KBS-3H safety assessment, overall system understanding is supported by probability calculations of specific parameters. For example, stochastic modelling is employed in analyses of flow and transport of groundwater and solutes /Lanyon and Marschall 2006/, as well as in the biosphere assessment – at least, concerning the main uncertainties in the biosphere transport modelling /Ikonen 2006/. However, throughout most of the radionuclide transport modelling (release from spent fuel assemblies, near field transport, and geosphere transport), the uncertainties are more related to limited knowledge than to spatial or temporal variability so a deterministic approach is considered more suitable than a stochastic one.

In the KBS-3H safety assessment, the deterministic analyses consider three base cases, one for each of three broad canister failure modes that are judged to be plausible: defective canister, corrosion, and rock shear. Parameters in the base cases are, in most instances, selected to be either realistic or moderately conservative in the sense that they are expected to lead to an overestimate of radiological consequences. The selection of models and parameter values is made according to "expert judgement" (see above) based on previous assessments, additional data gathering, laboratory studies and so on. This has been documented insofar as possible in Appendix A of the KBS-3H Process report. However, it is rare that parameter values are precisely known and there may also be other models for processes. Therefore, a larger number of variant cases exploring the impact of specific uncertainties affecting radionuclide release and transport is defined for each base case. The variant cases for the most part take a more pessimistic view of uncertainties than the base cases.

In general, no attempt is made to quantify the likelihood or probability of a particular case. An exception is the case of canister failure due to rock shear, where an estimate of probability of this event occurring over a one hundred thousand year time frame has been made, see Section 5.3 (for further details, see Chapter 6, /Smith et al. 2007b/).

5.8 Reserve FEPs

A reserve FEP is an event or process that is considered likely to occur and to be beneficial to safety and which is deliberately excluded from assessment cases, or at least from their analysis, when the level of scientific understanding is insufficient to support quantitative modelling, or when suitable models, codes or databases are unavailable. Such FEPs are termed reserve FEPs since they may be mobilised at a later stage of repository planning if the level of scientific understanding is sufficiently enhanced, and the necessary models, codes and databases are developed. The existence of reserve FEPs constitutes an additional, qualitative argument for reserves of safety beyond those indicated by the quantitative analysis /Nagra 2002a/.

The concept of reserve FEPs was introduced by Nagra during the Kristallin-I project and was later embraced by the NEA in its post-closure safety case guidelines for geological repositories /NEA 2004a/. The main reserve FEPs identified in the KBS-3H safety assessment are listed in Table 5-14. Similar reserve FEPs have been identified in other safety assessments, such as the Opalinus Clay project /Nagra 2002a/, SR-Can /SKB 2006a/ and TILA-99 /Vieno and Nordman 1999/.

5.9 Quality assurance

In a safety case it must be shown that the system considered in the safety assessment is one that can be realised in practice and therefore the safety case should include any quality management procedures required to ensure that the specifications of the engineered features are met /NEA 2004b/. An effective quality assurance (and control) system is also one of the tools to manage uncertainties, as pointed out in /Vieno and Ikonen 2005/. Quality assurance and control meas-

Table 5-14. FEPs identified in the KBS-3H safety studies as Reserve FEPs due to the presence of uncertainties. Reserve FEPs may be treated more realistically in the future when additional information becomes available. (Modified from the list in Section 8.8.8.3 in /Nagra 2002b/).

Reserve FEPs	Comment
The co-precipitation of radionuclides with secondary minerals derived from spent fuel, canister corrosion and supercontainer shell corrosion.	For example, co-precipitation of Ra-226 inside the canisters with Ba (from Cs-137) is neglected.
Sorption of radionuclides on canister or supercontainer shell corrosion products.	This FEP is categorised as reserve because there is some doubt about its influence, because the initial incorporation of radionuclides into magnetite and/or Fe-containing clays can create a secondary source releasing the scavenged radionuclides at a later time as the geochemical conditions change and/or the corrosion products age and recrystallise.
Natural concentrations of isotopes in solution in bentonite porewater, which could further reduce the effective solubilities of some radionuclides.	Only the concentrations of isotopes originating from the spent fuel are taken into account in evaluating whether solubility limits are exceeded in the reservoirs; the background concentrations of isotopes originating elsewhere are conservatively ignored.
Irreversible sorption of radionuclides in the near field or in the geosphere (surface mineralisation).	Matrix diffusion plus sorption on matrix pore surface are the only phenomena assumed to cause retardation and dispersion in the far-field transport analysis. Sorption on fracture fillings and diffusion into stagnant water pools in the fractures are neglected.
Long-term retardation and immobilisation processes (precipitation/co-precipitation) in the geosphere.	Natural analogue information provides evidence of such processes in the geosphere, see Section 3.3.
	Surface diffusion of cations in the rock matrix, which would improve retardation and dispersion, is neglected.
The delayed release of radionuclides due to the slow corrosion rate of the copper canister.	Radionuclides in the IRF are released immediately following the breaching of the canister.
	For example, physical hindrance or delay time provided by the cast iron insert once the copper canister is penetrated is neglected (i.e. the insert is not initially tight in case of a defective canister).

ures are routinely applied in the industrial world to qualify physical objects. Quality assurance in the field of geological disposal of radioactive waste and, specifically, the making of a safety case, presents the additional challenges of qualifying a largely abstract entity such as a body of knowledge, including data, processes, conceptual models and their application in computer codes. Below is a brief description of quality assurance procedures that are already in place or under development that apply to system components and to the safety assessment, including data, models and computer codes. Further Posiva plans for quality assurance are presented in the Safety Case Plan 2008 /Posiva 2008/.

5.9.1 Quality assurance applied to system components

Quality assurance procedures are in place at all levels of research and development and implementation work, for example:

- Canister manufacturing, sealing and inspection (under development).
- Canister storage, transportation and handling of canisters (under development).
- Buffer and backfill manufacture, characterisation, installation and inspection (under development).

- Construction and operations of Onkalo, especially quality controls on activities that have a potential impact on long-term safety, such as:
  - EDZ development,
  - groundwater inflow methods,
  - foreign materials introduced,
  - any activity involving drilling.
- Site characterisation methods, such as quality controls on:
  - hydraulic testing and flow logging procedures and results,
  - meteorological data,
  - water sampling procedures and results,
  - overcoring stress measurements for rock stress measurement data.

These quality assurance procedures are described in /Posiva 2006/ and references therein. The Olkiluoto Site Description 2006 /Andersson et al. 2007 and references therein/ also describes site characterisation activities and quality assurance procedures. This report also addresses how diverse sources of information (and methods of acquisition) are being brought together to form a consistent picture of the characteristics and history of the site, including possible alternative conceptual models.

To improve the quality of inspections, preliminary procedures will be developed for each component and technique. The role of an independent inspection organisation will be studied for qualification and manufacturing needs. The goal for all inspections is to conform to international standards.

To improve the quality of site monitoring results, automatic transfer of monitoring data from data loggers to all users (as preliminary data) in visual graphs and data availability through web-browsers will be considered to improve the control of possible effects caused by Onkalo, as planned in the monitoring programme.

All data collected at the Olkiluoto site are currently being centralised in a database called POTTI, which replaces the TUTKA database. SKB has an analogous site information central database called SICADA as well as a geographical database called the SKB Geographic Information System. Quality control and data evaluation procedures are being set up to analyse all data with respect to bias and representativeness before they are used /Posiva 2006/. The environmental data are now stored in the Forest Research Institute's database and have gone through its effective quality assurance system. Some of the Olkiluoto site data are from TVO's regulatory monitoring programme, which has its own quality assessment procedures. All data carried through the biosphere assessment and contributing to the doses will be quality assured at the latest in the Biosphere Assessment database, before the final simulations are carried out.

5.9.2 Quality assurance applied to the safety assessment

A particularly challenging application of quality assurance is its application to the safety assessment, including input data, models and computer codes. Input data used in the KBS-3H safety studies has been collected in Appendix A of the KBS-3H Process Report /Gribi et al. 2007/ along with main sources of data and assumptions. Appendix C of the Radionuclide Transport Report contains solubility data for relevant radionuclides and main assumptions concerning their speciation and oxidation state.

SKB has its own quality assurance plan for a long-term safety assessment of a spent fuel repository, as described in SR-Can /SKB 2006a, e and references therein/. SKB also applies a management system that has been certified according to the requirements of ISO 9001:2000. A quality assurance plan for the SR-Can project has been developed and partially implemented. The objectives of the quality assurance plan for SR-Can are to demonstrate the following:

- That adequate project management procedures, procedures for documentation, etc have been followed in the project.

- That all factors relevant for long-term safety occurring in earlier versions of SKB databases, and in the international NEA FEP database, have been considered in the assessment.
- That the exclusion of any of these factors is well justified by identifiable experts.
- That the approaches adopted to handling of all factors included are well justified by identifiable experts.
- How quantitative aspects of the assessment are handled by mathematical models and how the models (computer codes) have been quality assured.
- How appropriate data for quantitative aspects of the assessment have been derived and used in the assessment in a quality-assured manner.
- How the safety assessment reports have been properly reviewed and approved for correct and complete content.

A quality assurance plan for the SR-Site project is being developed on the basis of the experience gathered with SR-Can /SKB 2006ae/.

As part of the data quality assurance procedures, a document management system for the preservation of information has been defined and covers both electronic documentation and paper archives. It also defines the periods for which different documents should be preserved. For some documents there is no end point defined for preservation. Methods to achieve long-term (over periods exceeding a few hundreds of years) preservation of information are still to be defined. Generic studies have been carried out on this topic in various organisations (e.g. Andra and JAEA in Japan) but the preservation of data (in particular, electronic data) remains a challenge. Preservation of information is especially important in view of the period of several decades between the licensing of the repository and its final closure.

5.9.3 Quality assurance applied to models and codes

The long-term safety of a spent fuel repository is assessed predominantly with the aid of models and the application of a conceptual model is usually via a computer code. This requires:

- A scientific evaluation of the understanding of the processes involved in the modelling.
- The formulation of mathematical models that simulate the process or system of coupled processes, based on the understanding of the phenomena.
- The translation of the mathematical model into a computer code.
- Derivation of input data.
- Execution of the code and derivation of output.

All these aspects need to be documented and quality assured /SKB 2006ae/. Models used in the KBS-3H safety assessment are discussed in Section 5.4. All computer codes used in Posiva's safety assessments are developed according to a quality assurance procedure and verified by comparison with analytical solutions, alternative codes and experimental data.

Confidence in the modelling results is increased by means of the simulation of experiments and of natural analogue data. Posiva participates or has participated in international model validation studies such as the INTRAVAL (International Project to Study Validation of Geosphere Transport Models) project, which ended about 10 years ago, when the need for site-specific validation studies emerged. Currently, benchmarking studies are used to compare different codes to the same system. For instance, the VTT-developed code REPCOM has been verified against PORFLOW in /Nordman and Vieno 2003/. In the framework of the KBS-3H studies, a comparison of KBS-3H near-field radionuclide release and transport results has been made with results obtained using the SPENT code used by Nagra for the near field in recent safety assessments in Switzerland (see Appendix B of /Smith et al. 2007b/). Posiva is also indirectly participating in international integration groups, such as DECOVALEX (DEvelopment of COupled models and their VALidation against EXperiments in nuclear waste isolation) and NFPRO (Near-Field PROcesses) aimed at using different tools for modelling coupled thermo-mechanical-hydrological and chemical processes in deep geological disposal systems. Since 1992, Posiva and SKB have been actively involved in the Äspö Task Force on flow and transport where different codes and conceptual models are tested on the same experimental set-up. One task force within this project was dedicated specifically to models used in the safety studies.

SKB has established a model database called SIMONE storing quality assured, discipline-specific site models /SKB 2006a/.

Posiva is also participating to a task force comparing models for engineered barrier systems. This task force was originally managed by SKB but in future it may be transferred to the European Union auspices, under the name THERESA. Posiva and SKB also participate in international fora on biosphere models, such as BIOPROTA (www.bioprota.com), and in the end-users group of the European Commission's project ERICA (www.erica-project.org) on the assessment of exposures to non-human biota. Radionuclide transport codes are also being compared and validated by both Posiva and SKB.

6 Assessment results and complementary safety indicators

In this chapter, some results of the KBS-3H safety assessment are considered. However, is not the intention of this chapter to review the results of the assessment cases – these are given in full in the Radionuclide Transport Report /Smith et al. 2007b/ and a selection have been discussed in Chapter 5 (Section 5.6) – but to assess the overall implications of the results. The aim is to examine the results from a wider perspective than the comparison of the WELL-2007 safety indicator doses or the geosphere/biosphere flux maxima with the regulatory guidelines for a repository to be considered safe in the long term.

To this end, the results of the base case dose and release calculations are considered in terms of their significance compared to, for example, radiation exposure of humans arising from naturally-occurring radionuclides or risks, avoidable and unavoidable, of other types encountered in normal life. Results from other assessments are also compared with those from the KBS-3H safety studies, although the differences in waste type, disposal concept and site are taken into account.

Complementary performance and safety indicators[28] are measures other than dose of the hazard associated with the wastes and potential releases from the repository to the biosphere. Complementary indicators, such as radiotoxicity (RT), radiotoxicity index (RTI) and radiotoxicity flux (RTF), are calculated for the KBS-3H safety assessment results and compared with those arising from naturally occurring radionuclides in the environment.

6.1 Assessment results in perspective

6.1.1 Significance of the calculated doses compared to natural radiation exposures

The results of the KBS-3H base case calculations for geosphere releases into the biosphere arising from the initial penetrating defect, copper corrosion and rock shear failure modes are shown in Figure 6-1, in terms of annual dose to the exposed individual. The figure also shows the regulatory dose criterion (0.1 mSv/y), which is applicable only for the first 10,000 years /STUK 2001/ but here shown across the time range to one million years, and the range of natural background radiation in Finland for comparison.

From Figure 6-1, it is clear that not only are the calculated doses well below the regulatory criterion for even the worst of these failure modes, they are also many orders of magnitude lower than the typical natural external radiation exposure in Finland, which is around 2.5 mSv/y (0.04 to 0.3 μSv/h; /STUK 2007/), and the average Finnish exposure to all ionising radiation of 3.7 mSv/y (this also includes anthropogenic sources of radiation, such as medical x-rays and the Chernobyl fall-out). The maximum doses arising are 7.7×10^{-6}, 2.5×10^{-4} and 1.5×10^{-3} mSv/y, respectively, for the penetrated defect, canister corrosion and rock shear cases.

Natural background radiation in Finland is dominated by indoor inhalation of radon. The average annual concentration of radon gas in dwellings is about 120 Bq/m^3 (www.stuk.fi), which probably is the highest in the world, and gives rise to an average annual effective dose of about 2mSv. The present Finnish regulatory limit for indoor air is 400 Bq/m3 and it is 200 Bq/m^3 for new buildings. Based on these limits, the exposure to radon from inhalation has been estimated to be 3-5 mSv/y and 10 mSv/y, respectively, for a person staying in the house for the whole year.

[28] These complementary measures may be termed safety indicators if they can be compared to some standard which indicates the relative level of safety. If no such standard is available, then performance indicator is the normally accepted term.

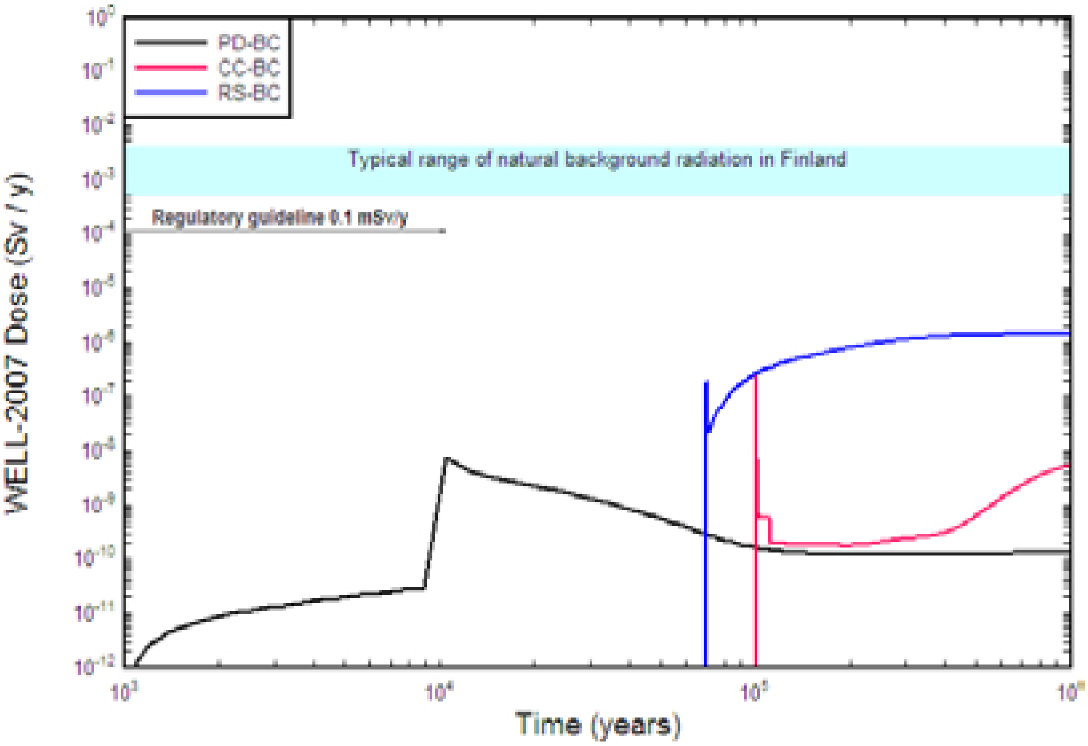

Figure 6-1. *Base cases annual individual dose as a function of time for the penetrated defect (PD-BC), canister corrosion (CC-BC) and rock shear (RS-BC) calculations for KBS-3H, compared to the typical range of natural background radiation in Finland.*

It is interesting to note that some (non-Finnish) populations are also exposed to significantly higher levels of radon, partly due to geology and partly to living arrangements; annual individual dose arising from radon inhalation in the UK can reach 100 mSv /HPA 2005/, 150 mSv in Switzerland /BAG 1992/ and exceed 200 mSv in certain areas of Brazil and India, although it is also notable that there are small but detectable health risks at these exposure levels /UNSCEAR 1993/.

Of course, there are some problems in making such a comparison: the natural background radiation can be measured whereas calculating doses for releases from the repository for the very distant future when the maximum releases arise is inherently a less certain process. Indeed, the array of assessment cases seeks to explore the range of possible future outcomes, some of which are much more likely than others, in order to increase confidence that the calculated results represent a comprehensive coverage of the risks.

However, part of the uncertainty about the calculated doses is related to the conversion of repository releases (in moles per year of a radionuclide, or Bq/y) into a dose to the exposed individual. Some scenario must be developed for how the radionuclides affect the exposed individual.

The exposure to the radioactivity may arise from:

- External radiation, for example, from nuclides in soils.
- Inhalation, when the nuclides are breathed in either as gas, e.g. radon (Rn-222) or as fine dust containing contamination.
- Ingestion, where the nuclides are consumed either as surface contamination on food stuff like fruit, as nuclides incorporated within the food stuff, e.g. in milk or cheese from cows which have fed on contaminated pasture, or dissolved in water.

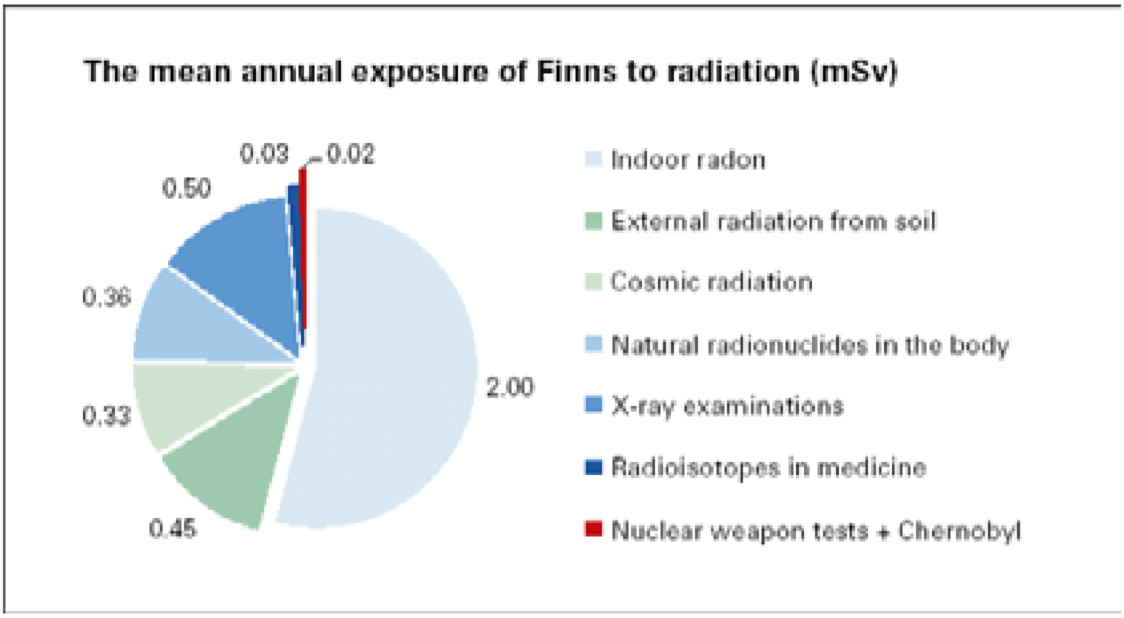

Figure 6-2. *Mean annual exposure (in mSv) of the Finnish population to ionising radiation, including anthropogenic sources /from Mustonen 2006/.*

In some assessments, a sophisticated biosphere model is used in which nuclides are partitioned between water, soil and plants and the individuals in the critical population are assumed to be, for example, subsistence farmers who eat plants and animals raised on contaminated land as well as drinking contaminated water /e.g. Nagra 1994, AECL 1994, SKB 2006a/. The IAEA BIOMASS project has created a number of reference biosphere models that also provide a potentially useful stylised method for dealing with the exposure of a population to the repository releases /IAEA 2003/. As reported in Section 5.5, Posiva are also undertaking a biosphere modelling programme as part of the KBS-3H safety studies which is reported in /Broed 2007 and Broed et al. 2007/.

The obvious problem with such scenarios is that assumptions must be made about the lifestyle, activities and land use for the far distant future when the releases from the repository could occur.

For the initial results of the KBS-3H assessment, a simpler procedure is used: the radionuclide releases are converted to doses by assuming that the exposure is only by drinking contaminated well water. It is assumed that a well is drilled in the vicinity of the repository or the discharge area for contaminated water, that the total annual releases from the repository are dissolved in 100,000 m^3 of water and, from this volume, an individual drinks 500 litres each year. This means that the individual ingests annually a fraction of 5×10^{-6} of the radionuclides released from the repository to the biosphere each year /Vieno and Nordman 1999, Smith et al. 2007b/.

It is of interest to compare drinking 500 litres per year of "repository well water" with municipal and well water in Eurajoki, where Olkiluoto is located and in Finland in general. Radioactivity in municipal water and various shallow (i.e. dug wells and springs) and deep well (i.e. drilled wells) waters is used to calculate the dose from drinking 500 litres per year for comparison with a similar amount of repository well water; the results are shown in Table 6-1.

Table 6-1. The content of radioactivity in some drinking waters from Finland and the dose arising from consumption of 500 litres (L) per year, compared to drinking the same amount of "repository well water".

Water supply	Radon (Bq/L)	Ra-226 (Bq/L)	Uranium (Bq/L)	Dose[1] (mSv)
Municipal water (Finland average)[2]	78	0.0079	0.063	0.14[3]
Municipal water (Eurajoki)[2]	41	–	0.048	0.07[3]
Dug wells and springs (Eurajoki)[2]	50	0.026	0.025	0.09
Drilled wells (Eurajoki)[2]	250	0.035	0.73	0.46
Repository well water[4]	–	– ($1.3 \cdot 10^{-3}$)	–	$7.7 \cdot 10^{-6}$ [5] ($1.5 \cdot 10^{-3}$)

[1] Annual dose based on ingestion, from drinking 500 L of water per year. Note that the radon is the dominant source of radioactivity and dose in all non-repository waters.

[2] Source of data: /Voutilainen 1998/, as translated and compiled by K.-H. Hellmuth in /Pitkänen et al. 2003/.

[3] This does not include a small contribution from anthropogenic nuclides H-3, Sr-90 and Cs-137 (from Chernobyl) which would add approximately a further 3×10^{-4} mSv per year /Mustonen 2006/.

[4] Dose dominated by C-14 in the penetrating defect base case (PD-BC) and Ra-226 in the rock shear base case.

[5] Maximum dose from dilution of annual release (all nuclides) in 10^5 m^3 of well water in the PD-BC (rock shear base case values in brackets).

Even the very small amounts of natural radioactivity (for example, the uranium concentration in the drilled wells corresponding to 0.73 Bq/L is 23 ppb) produce significantly higher doses than the repository well water. It should also be noted that the peak dose from the repository well water in the PD-BC arises at 10,440 years and is considerably lower after this time. In the RS base case, the peak dose occurs at around 1 million years although, in this case, doses over the whole period between releases first occurring at around 70,000 years and the maximum are greater than the maximum releases from the PD-BC.

6.1.2 Risks associated with radiation exposure

The term "risk" is here used to denote the scientific concept where a risk is composed of both a hazard and the probability of that hazard occurring, so:

Risk = Hazard x Probability of that hazard.

In everyday life, of course, the two components of risk become separated so that every day activities, which might have extremely serious consequences, become acceptable as the perception of the likelihood allows people to ignore the hazard. In this way, driving a car or riding a bicycle become acceptable as millions of people voluntarily accept the risk involved even though both activities have a small but significant risk of death.

With respect to radioactivity, it is of interest here to consider what risks are involved in the small doses which may occur some time in the far future due to the repository releases. For radiological protection purposes, the ICRP recommends the following risk coefficients[29], as a mean for all ages and low doses /ICRP 1991/:

- Fatal cancer in exposed individual: 0.05 per Sv.
- Serious hereditary defect in all generations of offspring: 0.01 per Sv.
- Allowance for loss of life expectancy and non-fatal cancer: 0.01 per Sv.

[29] It may be noted that the ICRP risk coefficients apply only to populations and are not intended for use in estimating risks in individuals or sub-groups, and that the related health effects are disputed by some organisations but provide a useful comparative tool here.

Based on these figures, the consequences for a population of 50,000 exposed to a dose of 4 mSv would be 10 deaths due to radiation-induced cancer. Statistically, 1 of these deaths would be due to leukaemia within 20 years and 9 to other cancers over somewhat longer periods of 10 to 40 years /Baertschi and Sumerling 1994/.

For the case of continuous low-level exposure, the assumption is made that the lifelong risk of death due to a radiation-induced cancer is also an average of 0.05 /Sv where the dose is integrated over the whole lifetime.

Using these values and assumptions, the possible induced fatal cancers due to the repository can be estimated: the releases from the KBS-3H repository (assuming the penetrating defect base case PD-BC) give a lifetime (i.e. 70 years) dose of about 5.4 x 10^{-4} mSv to the individuals of the exposed population at the time of greatest releases, which result in an additional individual lifetime risk of about 3 x 10^{-8}. It must be stressed that this is an overestimation of the actual risk since the probability of the exposure is not taken into account. In an exposed population of 50,000, there would be about 2 x 10^{-5} deaths per year, or roughly one death every 50,000 years, attributable to repository-derived radiation-induced cancer. However, it is important to note that the radionuclide releases from the repository will occur in a rather limited area, thus the population most exposed will be very small compared to this and may number only 10's or 100's of people, depending on how the land is used in the distant future.

In comparison, the average exposure of the population in Finland to combined natural background, medical and other anthropogenic sources of radiation is 3.7 mSv which results in around 9 deaths per year in an exposed population of 50,000 people. Thus the additional dose due to the hypothetical release from the repository will, statistically, increase the deaths due to radiation-induced cancers from 9 to 9.00002.

/Baertschi and Sumerling 1994/ give some examples of activities estimated to carry a risk of fatality of one in a million, usually considered to be an acceptable level of risk, but even these activities (Table 6-2) are nearly 100 times more likely than dying from cancer induced by repository-released radiation (PD-BC, over 70 year lifetime); even being struck by lightening is about 10 times more likely (5 x 10^{-7} per year).

Table 6-2. Activities which are estimated to carry a risk of fatality of one in a million (modified from /Baertschi and Sumerling 1994/, Table 5.14).

Activity	Potential hazard(s) [1]
Smoking 2 cigarettes	Cancer and circulatory disease
Living with a cigarette smoker for 2 months	Cancer and circulatory disease
Driving 300 km	Accident
Cycling 50 km	Accident
Flying 4,000 km (commercial airline)	Accident
4 hour flight at an altitude of 10 km	Cancer from cosmic radiation
1 chest X-ray with modern equipment	Cancer from X-rays
Annual consumption of:	
– 150 l Eurajoki municipal water	Cancer from radionuclides (ionising radiation)
– 200 g fresh mushrooms	
– 2 kg meat from a charcoal grill	Cancer from hydrazine derivatives
– 400 g peanuts [2]	Cancer from pyrolitic products
	Liver cancer from aflatoxin B
80 years living at the place and time of maximum release of radionuclides from the repository (with the canister corrosion base case release rates)	Cancer from radionuclides (ionising radiation)

[1] Potential hazard leading to a risk of one in a million; some activities also entail other hazards.

[2] Assuming 1 µg aflatoxin B1 per kg (concentration is generally much lower).

6.2 Complementary indicators

The WELL-97 doses that are calculated in safety assessment modelling of the releases from the near field and the far field are viewed not as predictions or compliance evaluations tools but rather as measures of the capability of the repository/host rock system to provide isolation of the waste and containment of radionuclides over the required time period. Such calculations of dose are termed safety indicators and have been discussed in sections 5.6 and 6.1.

As discussed in Section 6.1, however, there are some difficulties with converting calculated repository releases into doses to the biosphere, not least of which are the assumptions about human activities in the distant future. The regulatory dose criterion of 0.1 mSv/y only applies over the first 10,000 years after repository closure /STUK 2001/ because of increasing uncertainly about the environment; this is acknowledged in the presentation of the results of the KBS-3H assessment cases given in the Radionuclide Transport Report /Smith et al. 2007b/.

As a result, complementary safety indicators, which are not dose-based and which rely on comparisons with natural materials and processes, are increasingly being used to evaluate the performance of the repository in protecting future generations (see, for example, /Nagra 2002a/, Section 8.2.8.2).

Two complementary safety indicators are used in the KBS-3H assessment:

- Radiotoxicity of the spent fuel evaluated as a function of time and compared to that of natural materials such as uranium ores of different grades, or the repository host rock itself. This safety indicator is discussed in detail in Chapter 2.
- Radiotoxicity fluxes due to radionuclides released from the repository over time are compared with natural fluxes due to the movement of groundwater or surface erosion.

Another safety indicator, which could be used in future, is the distribution of radiotoxicity in different components of the repository system, evaluated as a function of time, illustrating the fate of radionuclides and the extent to which they decay before reaching the biosphere. Currently, assessment case results are not calculated in a form from which this information can be extracted.

6.2.1 Radiotoxicity of the spent fuel

The radiotoxicity indices (RTI) (see Chapter 2, Eq. 2-1) of 1 tonne and 5,500 tonnes of Finnish spent nuclear fuel are shown in Figure 2-1 and compared with the RTI of the volume of various grades of uranium ore required to fill the deposition drifts and also that of the Olkiluoto host rock removed to construct the KBS-3H repository. The RTI of 5,500 tonnes, the expected inventory of the repository, is around 5×10^{15} at 40–50 years after removal from the reactor (this is the minimum storage time for cooling; the spent fuel will actually be removed from the reactors over a period of several tens of years, thus some may be significantly older at the time of emplacement) when it will be emplaced in the repository. This total inventory RTI is thus somewhat more than an order of magnitude greater than that of the richest Cigar Lake uranium ore if it were used to fill the deposition drifts. However, while the RTI of the Cigar Lake ore will not significantly change over the next 1 million years, due to the long half lives of the naturally-occurring U and Th isotopes, that of the spent fuel will decrease by more than three orders of magnitude over this time. The RTI of the total spent fuel inventory is also about equivalent to that of the volume of host rock that would be removed during excavation of the drifts (about one million cubic metres, assuming it is all constituted by tonalite-granodiorite, containing naturally-occurring radionuclides) after a little over one million years.

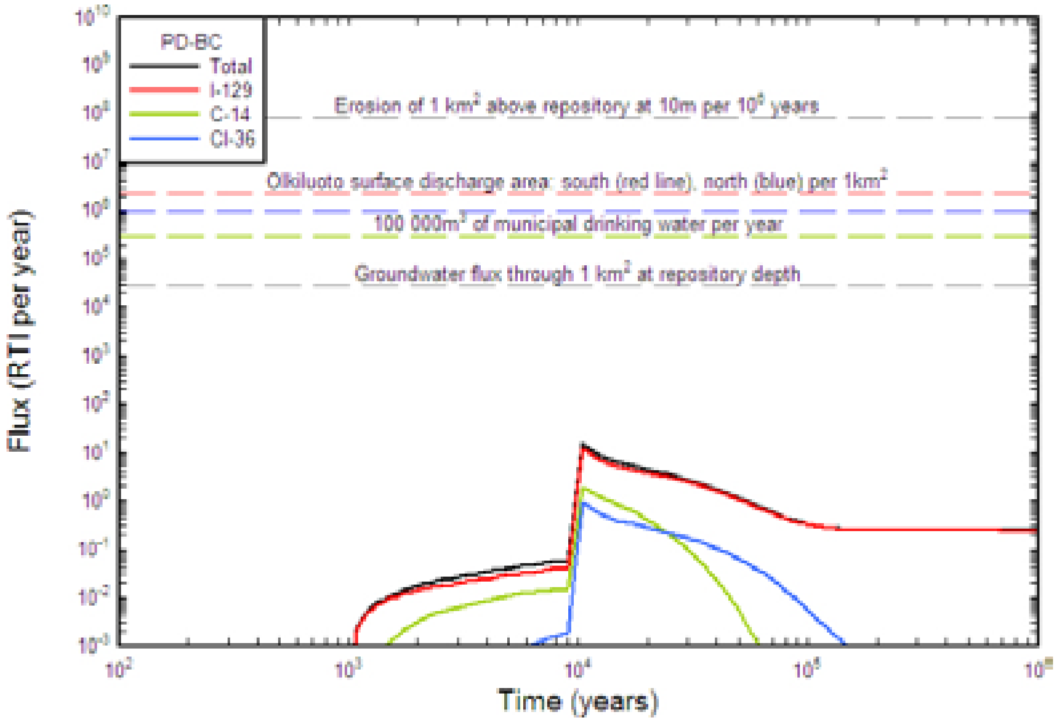

Figure 6-3. *Radiotoxicity flux from the KBS-3H repository into the biosphere for the penetrating defect base case (PD-BC) compared to a range of radiotoxicity fluxes due to naturally occurring radionuclides (see text and Appendix B for further explanation).*

6.2.2 Radiotoxicity fluxes

In Figure 6-3, the radiotoxicity flux of nuclides from the repository into the biosphere for PD-BC is compared with that of radionuclides in groundwater discharges. Three discharge planes are considered, based on the investigations of natural elemental fluxes reported in /Pitkänen et al. 2003/, namely:

- Deep groundwaters discharged across a reference plane of 1 km^2 at a depth of 375 m at the Olkiluoto repository site (upper-deep reference plane of /Pitkänen et al. 2003/). This is noted on Figure 6-3 as “groundwater flux through 1 km^2 at repository depth”.
- Shallow groundwaters discharged at the Northern discharge area (1 km^2) at Olkiluoto site.
- Shallow groundwaters discharged at the Southern discharge area (1 km^2) at Olkiluoto site.

In addition, to widen the comparison, the radiotoxicity flux of the surface rock eroded from the 1 km^2 above the repository (at an erosion rate of 10 m per 1 million years, see Section 4.4.1) is also shown. Further, the radiotoxicity flux of the annual consumption of 100,000 m^3 of municipal drinking water is shown as this is the dilution volume for the annual repository releases used in calculating the WELL-2007 dose conversion factors discussed previously. By analogy with the WELL-2007 DCFs, where individual annual consumption is assumed to be 500 litres, the 100,000 m^3 shown in Figure 6-3 is thus the annual consumption of drinking water by a population of around 200,000 people.

In view of the stability of the geological environment of Olkiluoto, these natural fluxes might be considered to be relevant to temperate interglacial periods in the far distance future and thus provide an appropriate comparison for the releases from the repository at these long times. It is clear that the repository radionuclide fluxes would be significantly dominated by the orders of magnitude larger natural fluxes, especially that relating to erosion of the surface rocks. The data used to calculate the RTFs shown, as well as the RTIs discussed in Section 6.1, are given in Appendix B.

6.3 Bounding analyses

6.3.1 Consequences of multiple canister failure within the regulatory compliance period

The results for all the assessment cases shown in Figures 6-1 and 6-3 involve just a single canister from a total of about 3,000 so the doses could be significantly higher if more canisters were to fail. Considering failure of all the canisters within the million-year assessment period constitutes a bounding case – it is not in any way an expected future outcome, as all canisters are expected to retain their integrity over this period, but is calculated only to demonstrate the hypothetical worst outcome. In this case, the maximum doses would be about 3,000 times higher in the case that all the canisters failed, although this would depend somewhat on the distribution of failures over time. If all the canisters developed defects over time which released radionuclides in a manner equivalent to the penetrating defect case, the maximum hypothetical dose would be around 0.02 mSv/y – still below the regulatory criterion.

In the most extreme case, however, assuming all the canisters failed by rock shear so that the geosphere also fails to provide a significant barrier to the radionuclides, the total dose would be about 4.3 mSv/y predominantly due to Ra-226. This dose is slightly higher than the top end of the range of natural background radiation and similar to the Finnish average annual exposure to all ionising radiation sources of 3.7 mSv/y. Although this exceeds both regulatory criterion and natural background radiation doses in Finland, it is still within what might be termed "reasonable bounds" since other populations experience significantly higher natural background radiation (as noted in Section 6.1.1) which is considered acceptable: "*In areas of high natural background radiation, an increased frequency of chromosome aberrations has been noted… No increase in frequency of cancer has been documented in populations residing in areas of high natural background radiation*" /NRC 1990/. Even so, it has to be emphasised that this is an extremely unrealistic case, not just because of the nature of the failure of the canisters, but because it is assumed that the radionuclides released throughout the whole repository are all captured by the single well that supplies drinking water to the exposed population – highly unlikely when the size of the repository footprint (ca. 1.6 km^2) is considered.

6.3.2 Consequences of ultimate failure of the multi-barrier system in the farthest future

Over a sufficiently long time frame, all canisters will eventually fail. For the majority of canisters, the most likely eventual failure mechanism is the slow corrosion of the copper shell, leading to failure after several hundred thousand years or more. In the hundreds to thousands of millions of years before the repository horizon is exposed at the surface by erosion and uplift, the evolution of the repository materials is uncertain and any comments are necessarily speculative. It is possible, but unlikely, that the fuel, radionuclides and repository construction materials will eventually become widely dispersed in the geological environment. It is more likely that at least some of the materials, including the spent fuel, will remain largely in situ. For example, the copper may be partly replaced under reducing conditions by a suite of copper sulphides, which are insoluble and not likely to become dispersed until erosion brings the repository horizon close to the surface, and the fuel matrix may experience only limited dissolution over time due to its relative geochemical stability. Thus, in some respects, after very long times the repository materials may tend to resemble a heterogeneous uranium ore body, perhaps analogous to granite- or sediment-hosted Cu-U deposits. The consequences of possible exhumation of the repository are difficult to assess, given the extreme length of time before this could occur. However, the processes involved are likely to be similar to exhumation of small uranium deposits where the local climatic and topographic conditions primarily determine the rate at which the ore body is dispersed.

7 Summary and conclusions

The aim of this report is to provide information on the long-term safety of a KBS-3H repository at the Olkiluoto site by bringing together the evidence and arguments which are complementary to the quantitative safety assessment. This chapter summarises the main points of the preceding chapters that assist in fulfilling this aim.

7.1 Support for the concept of geological disposal

Geological disposal has been chosen as the preferred long-term management option for spent fuel in Finland as, according to current international understanding, deep geological disposal is the only spent fuel management option that offers the long-term passive safety required for the safe disposal of spent nuclear fuel.

The general concept of geological disposal is supported by many years of investigations internationally into a variety of repository design concepts. In Finland and elsewhere, repository design concepts have been developed and demonstrated to provide the required stability and longevity in different geological environments and for a range of geochemical conditions.

The requirements for a robust disposal system have led to a conscious decision to rely on a few, well known and understood materials such as copper, steel and clay for deployment in engineered structures which act in concert with the isolation and protection provided by the geological environment. Disposal concepts based on the principle of multiple, redundant barriers have been demonstrated to provide high levels of long-term safety even where uncertainties remain in the behaviour of some individual components.

The development of the repository system is complemented by the staging of the implementation process over approximately 100 years whereby the repository design is flexible enough to be optimised to make use of improving site data, scientific knowledge and technical experience throughout the stages of site investigation and characterisation, repository construction and operation and even in the closure and decommissioning phase.

Integrated safety assessments have been conducted by numerous disposal organisations worldwide over more than 30 years for a wide range of sites, host rocks and repository concepts. The overriding message from their accumulated results and conclusions is support for the possibility of safe geological disposal and confidence in both the robustness of the long-term safety and the practical feasibility of the concepts assessed.

7.1.1 Support from the KBS-3V programme

It is acknowledged that the KBS-3H disposal system is different in key aspects, particularly relating to emplacement of the spent fuel canisters and buffer, from the well-known and assessed KBS-3V design. However, it is also clear that the safety functions of the main safety-relevant components (i.e. canister, buffer, tunnel seals and host rock) are essentially the same as in the original KBS-3V, thus much of the long-term evolution of the repository system and the processes which influence long-term safety are common between the two designs. As a result, much of the 30 years' scientific R&D and technical development from the KBS-3V programme is directly applicable to the KBS-3H project, allowing future work on the latter design to focus on KBS-3H specific issues.

7.1.2 Support from natural systems

Observations of natural systems provide indirect evidence for safe geological disposal. Evidence to support repository system performance as well as understanding and confirmation of important, safety-relevant processes have been gained from studies of natural and archaeological analogues. These have included uranium ore deposits, in many different geological environments, which demonstrate the barrier effect of natural clay materials and the longevity of uranium minerals, analogous to spent fuel, under conditions similar to those expected around a deep geological repository. Natural uranium deposits also provide insights on radionuclide retention mechanisms in the geosphere through processes of precipitation, co-precipitation and sorption on the minerals filling the fractures.

Natural systems can be very useful in giving confidence that data used in the quantitative assessment are reasonable. For example, when setting solubility limits for elements, observations from natural systems under more realistic conditions than are possible in laboratory experiments can support the choices of solubility-limiting phases and speciation in groundwater on which these limits are based. Moreover, data from natural systems can be used to test chemical models and databases and to identify influences, such as microbial activity, which are not easily amenable to quantitative treatment with current models.

The long-term durability of native copper in relevant conditions is illustrative evidence for the long-term stability of copper canisters. Several examples of natural copper ores and one archaeological analogue have corroborated the evidence that the sub-surface conditions in Finland will also preserve copper.

Natural systems also have an important role to play with respect to the long timescales involved in radioactive waste disposal. The Finnish regulatory dose criteria /STUK 2001/ acknowledge that, due to increasing uncertainties about the surface environment in the future, the calculation of dose to a human population becomes less meaningful in the long term, especially after the onset of a future glaciation. Other calculated performance and safety indicators can, however, be used to complement calculated doses. Such indicators can show, for example, how the potential toxicity of spent fuel due to ionising radiation compares with that of naturally occurring radioactive materials, such as uranium ore bodies, and how calculated radionuclide releases from the repository compare with naturally occurring radionuclide fluxes. Natural analogues allow us to examine the results of processes that have been occurring in natural systems for periods of time comparable to, or longer than, those used to assess the safety of the repository. In some cases, it is not possible to extract all the detailed information we would like to have but the messages these analogues can convey should not be underestimated: that the Cigar Lake uranium deposit has no surface geochemical anomaly to indicate its presence after 1.3 billion years of existence is evidence of the most direct kind that a system with a suitable hydrogeological and geochemical environment can provide safety over the timescales required.

7.2 Support from the properties of the Olkiluoto site

Before any other more subtle considerations of hydrogeological or rock mechanical conditions, a suitable site for a geological repository for radioactive waste must have long-term stability and absence of exploitable natural resources, such as mineral deposits or geothermal energy potential.

Geological stability is defined in this respect as the absence of significant uplift over the next one to ten million years and of major tectonic activity which could lead to fault activity in the repository area.

Understanding of the geological environment and evolution of the Olkiluoto site means that long-term geological stability can be confidently expected based on the history of the regional over the last several hundred million years.

Investigations of the site and surrounding areas have led to detailed knowledge of the site and, on the basis of this knowledge combined with more general understanding of the genesis of ore deposits, it can also be demonstrated that there is no significant potential for exploitable mineral deposits.

Olkiluoto site characterisation activities have been ongoing for over 20 years and the current state of knowledge about the site can be summarised by noting that surface conditions, geology, rock mechanical properties and the status of in situ stress are well understood although some uncertainties remain. It has been found that the rate of groundwater flow at the planned repository depth is low and geochemical conditions are favourable to the engineered barrier system with reducing conditions, low levels of sulphide and moderate salinity of about 10–20 g/L. Furthermore, the number of major, fast water transport pathways via the fracture zones is low in the repository area and their characteristics are known, although there are some uncertainties about such features outside the Well Characterised Area.

7.2.1 Support from Onkalo activities

The construction of the Onkalo underground characterisation facility will perturb the near field of the planned repository but this has been anticipated by implementation of a monitoring programme that established baseline conditions before the construction began and that will continue throughout the Onkalo operations and also the construction and operation of the eventual repository.

The most extensive use of the information collected from the monitoring system is for the further characterisation and understanding of the Olkiluoto site. New information can lead to changes in existing geological, hydrogeological, geochemical or rock-mechanical models of the site and, should these be important, changes in design or construction methods may also be considered.

Given the current schedule for Onkalo construction, the first data from observation niches built in Onkalo at repository depth will be available at the earliest in 2009–2010. According to Posiva's license application schedule, these data cannot be incorporated in the interim report on licensing preparedness due in 2009 /Posiva 2006/. However, as a wealth of information from Onkalo will be available by the end of 2010 or early 2011, these data will be presented in the final Complementary Evaluations Report for the KBS-3V design alternative due in 2011 in support of the license application due in 2012.

Onkalo monitoring will not stop when the repository goes ahead (if the construction license is granted). It is likely that there will still be open questions and possibly significant uncertainties with respect to some aspects of the site that only further data from repository depth can resolve. These late data will feed into the next generation of models, along with information provided during repository construction, in support of future safety assessments required at key programme milestones. These data will also be relevant to the KBS-3H safety assessment as they concern mainly site properties and not design features.

7.3 Support from comparison with TILA-99 and SR-Can

KBS-3H is based around the same copper canister as TILA-99 and SR-Can and, as in those assessments, the normal evolution of the repository expects that the canisters will be very long-lived, with no failures within the million-year assessment period. However, if there are no releases of radionuclides, it is difficult to assess the performance of the repository and the disposal concept. Thus KBS-3H follows TILA-99 and SR-Can in assuming a number of failure modes for the canister which can be used to explore the processes resulting in radionuclide releases and the impacts of uncertainties in the evolution of the disposal system. The primary failure mode is a hypothetical initial penetrating defect. In this failure mode, the other safety barriers of buffer and geosphere are undisturbed and should function as expected. This failure mode is therefore useful to investigate uncertainties related to the evolution and performance of these barriers and a large number of calculational cases are defined to meet this end. The other two failure modes, by copper corrosion after erosion of the buffer and by rock shear, assume that performance of the geosphere is also degraded. In the rock shear case, the geosphere is effectively circumvented by the assumption that the fracture causing the canister damage becomes a fast transport pathway to the biosphere. A smaller number of calculational cases are based on these failure modes, mainly to examine uncertainties specific to these failure modes.

A detailed comparison of the calculational cases with those of SR-Can and TILA-99 has confirmed that there are no omissions or gaps in the KBS-3H assessment apart from where limitations related to the scope of the assessment mean that the treatment of some uncertainties is put aside at this stage. The emphasis of the assessment is on uncertainties relating to the evolution of the near-field conditions due to the 3H-specific components, such as effect of the supercontainer on the transport barrier provided by the buffer. Uncertainties that are not considered relevant in discriminating between the performance of KBS-3V and KBS-3H repositories are either not addressed or are analysed in less detail than others. These include uncertainties in the transport barrier provided by the geosphere, in the biosphere and related to future human actions.

The conceptual models used in the KBS-3H have been developed from those used in TILA-99, partly as a result of increased understanding of the processes involved, and availability of more data in some cases, and partly in order to be able to assess the processes with different significance to, or potential impact on, KBS-3H compared with KBS-3V assessed in TILA-99. Many of the KBS-3H assessment conceptual models draw heavily on developments in the recent SR-Can study for processes common to both the KBS-3H and -3V designs.

Comparison of the results of the KBS-3H safety assessment calculations with those of TILA-99 and SR-Can confirms that they are consistently in the same order of magnitude. Moreover, where there are differences between results, it is possible to identify the reasons. For example, SR-Can results have noticeable releases of Ra-226 to the biosphere which give rise to an increase in dose after about 100,000 years in several assessment cases. There is no similar contribution from Ra-226 seen in KBS-3H assessment results in most cases because these releases in SR-Can reflect the influence of a small number of fast geosphere pathways that arise from the stochastic geosphere model. A similar contribution from Ra-226 only occurs in KBS-3H in the cases where the single representative geosphere flow path is assigned parameter values which make its properties more similar to the SR-Can fast pathways as, for example, in the rock shear case.

The site investigations as well as the safety assessment activities are fully supported by quality management systems which will also be extended to manufacturing and emplacement of components as the repository programme moves into an implementation stage. Knowledge management systems are also being developed to ensure the rigorous control of the information used in the whole programme over the long time frame up to, and even after, repository closure.

7.4 Support from complementary analyses

Consideration of radiotoxicity of the spent fuel, compared to naturally-occurring radioactive materials, such as U ore bodies, allows some interesting points to be demonstrated. In particular, the steep decline in the radiotoxicity of the spent fuel over the first thousands of years due to decay of activation products contrasts strongly with naturally-occurring U and Th. These natural elements have primordial isotopic compositions (i.e. unaffected by man-made processes) and very long half-lives, thus, over the million year period of the assessment, show no apparently change in radiotoxicity. So about 200,000 years after repository closure, the spent fuel has decayed to about the same radiotoxicity as the larger quantity of uranium which was used to produce the original fuel. This comparison is not made in order to suggest that the spent fuel is “safe” at this time but that the magnitude of the hazard has reduced to the level comparable with a naturally occurring hazard of a similar but not identical form.

Releases of radionuclides from the repository in various cases can be converted to doses to a hypothetical population of individuals who drink from a contaminated well. This allows the doses to which they are exposed from the repository to be compared with doses arising from natural background radiation and consumption of ordinary drinking water. On this basis, the calculated doses are insignificant – not only orders of magnitude below regulatory dose criteria, but more than three orders of magnitude smaller than the dose arising from drinking the same amount (500 litres) of ordinary tap water in Eurajoki, where Olkiluoto is located. Even in very pessimistic cases such as the rock shear failure case in which the geosphere barrier is largely circumvented, the repository dose is about 50 times smaller than drinking an assumed yearly volume of 500 litres of Eurajoki tapwater.

Comparisons of the risks associated with these very low doses with other risks emphasise their insignificance. However, it is acknowledged that the calculation of dose requires assumptions about the behaviour of populations in the far future that are difficult to justify – even if they are considered “stylised cases” and not predictions of future behaviour.

In order to address such shortcomings, further evaluations of the repository releases were made on the basis of radiotoxicity flux in place of the dose comparisons. These evaluations also confirm the insignificance of the calculated repository releases when they are compared to radiotoxicity fluxes associated with groundwater discharge in the Olkiluoto area, or erosion of the not-particularly U-rich rocks in the area.

Finally, the issue of the “worst case scenario” was addressed by considering a bounding case where all spent fuel canisters failed within the assessment period. This is contrary to all expected and conservative assumptions in the calculated cases where only a single canister, or a small number of canisters, is calculated to fail. However, even in this extreme, hypothetical case, the maximum annual effective dose arising from all 3,000 canisters is around 4 mSv per year at one million years. This is about the same as the annual exposure of the Finnish population to natural and anthropogenic radiation sources (including all exposures) and less than the natural background in some countries.

7.5 In conclusion

The preceding chapters have gathered together evidence of many types to support the long-term safety of a KBS-3H repository at Olkiluoto and to complement the quantitative assessment of the performance. This evidence covers a range of scales including:

- General supporting arguments for the concept of geological disposal.
- Support for the robustness of the KBS-3H design by comparison with similar concepts.
- Support for the rigour and completeness of the quantitative assessment by comparison with those made elsewhere on similar concepts and/or similar host rocks.
- Support from natural analogues for the choices of data used in the assessment.
- Support from complementary evaluations of the calculated releases which indicate the relative insignificance of the calculated doses and the low level of hazard implied.
- Support from natural analogues that the repository system is founded on sound understanding of the behaviour of the components over the long timescales required.

This evidence will be supplemented in the coming years with results from the Onkalo activities, improved scientific understanding of processes and data gathering from natural analogue studies to update complementary lines of evidence within the framework of the next Complementary Considerations Report for the KBS-3V repository design, which Posiva plans to publish in 2011.

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Appendix A

Summary of the differences between KBS-3V and main issues requiring further work

In this appendix, the main differences between the KBS-3V and KBS-3H alternative repository designs for spent fuel disposal in crystalline rock are summarised. The analysis is based on the Olkiluoto site, although the main outcomes may also be applicable to other repository sites for spent fuel in crystalline rock. The descriptions of processes and system evolution for a KBS-3H repository, and the difference analysis with KBS-3V, indicates that most safety issues are common to the two designs, but that there are also differences. Often, these are differences in the significance to, or potential impact of, an issue to each of the designs. Key safety issues that are judged to have different significance to, or potential impact on, KBS-3H compared with KBS-3V, concern mainly the early, transient evolution of the repository.

Table A-1 is structured according to the subsystems fuel/cavity in canister, canister, buffer/distance block, supercontainer and other steel structural materials, drift end plugs and backfill, and geosphere. The main uncertainties associated with processes occurring in each of these systems for KBS-3H are also briefly summarised. Several other uncertainties that are common to the KBS-3H and -3V designs (e.g. uncertainties related to the evolution of the site and climatic conditions) are not discussed here.

Table A-1. Major differences between KBS-3V and 3H and issues for further work (from Table 3-1 of the Safety Assessment Summary Report, /Smith et al. 2007c/).

System components/ (groups of) processes	KBS-3V	KBS-3H
Copper canister, cast iron insert, fuel/cavity in canister		
The canister, insert and fuel are the same in both alternatives		
Buffer		
Piping/erosion by water and gas, chemical erosion	Within deposition hole at buffer/rock interface in the case of high initial inflow rates (however, the holes can be selected individually and those with larger inflows will be rejected). Also, in the longer term, chemical erosion is possible in the event of an influx of glacial meltwater. Loss of buffer around one canister due to piping/erosion or chemical erosion by glacial meltwater will not affect the buffer around neighbouring canisters.	Piping/erosion may affect buffer density at bentonite/rock interface in canister sections with high initial inflow rates and in canister sections adjacent to these; mitigating the effects of piping/erosion is considered to be a major challenge in the design of KBS-3H and has led to the consideration of two candidate designs and various design alternatives. Deposition drift sections with inflows larger than a specified limit are not used for deposition – but sealed tightly. This will affect the utilisation degree of deposition drifts. Design is still under development /Autio et al. 2007/. Chemical erosion is possible in the event of an influx of glacial meltwater. Loss of buffer around one canister due to piping/erosion or chemical erosion by glacial meltwater may affect the buffer around neighbouring canisters, since the buffer density along the drift will tend to homogenise over time.
Displacement of buffer/distance block (leading to a reduction in bentonite density)	Swelling of buffer from deposition hole into drift above the hole may lead to lowering of bentonite density; rock stress distribution leads to risk of rock slabs at mouth of deposition hole.	Axial displacement of distance block by hydraulic pressure build-up may lead to lowering of bentonite density and must be counteracted by a rapid emplacement rate and by the use of steel plugs and steel rings bolted to rock, as described in the current reference design /Autio et al. 2007, Börgesson et al. 2005/. Axial displacement due to heterogeneous swelling is limited by friction and by drift end plug.

System components/ (groups of) processes	KBS-3V	KBS-3H
Iron/bentonite interaction	Relevant only for failed canisters.	In addition to the processes relevant to KBS-3V, significant geochemical interactions between supercontainer and buffer will take place (iron/smectite interaction, iron-silicate formation, cation exchange, etc); these processes may affect the buffer density, swelling pressure, hydraulic conductivity and other properties; these effects are locally limited at early times, but may develop with time and affect larger parts of the buffer /Johnson et al. 2005, Carlson et al. 2006, Wersin et al. 2007/.
Gas transport and possibly gas-induced porewater displacement	Relevant only for failed canisters.	In addition to the processes relevant to KBS-3V, significant gas effects are expected /Johnson et al. 2005/ due to anaerobic corrosion of supercontainer and other steel components (retarded resaturation, air trapping, gas dissolution/diffusion/advection, gas pressure build-up, gas leakage, gas pathways along drifts, etc); during this early phase, no radionuclide transport is expected.
Effects of engineering and stray materials	Effects of concrete bottom plate, stray materials, bentonite pellets.	Effects of steel rings, rock bolts, steel feet, water/gas evacuation pipes, grouting, spray and drip shields, cement.
Supercontainer and other structural components within the deposition drifts		
Materials, geometry, properties	N/A	
Steel corrosion and formation of corrosion products	N/A	For the expected steel corrosion rate, complete conversion to oxidised species occurs within a few thousand years.
Gas generation by anaerobic corrosion of steel	N/A	Gas generation rates are significant although the overall amount of gas produced is moderate; for the effects of gas, see buffer.
Effects of volume expansion (magnetite formation)	N/A	Volume expansion of corrosion products may increase buffer density and swelling pressure.
Ion release to bentonite porewater	N/A	Leads to iron/bentonite interaction.
Effect of supercontainer on water flow paths along the periphery of the drift	N/A	The physical properties of the corroded supercontainer have not been evaluated. Although the porosity and hydraulic conductivity of the corrosion products may be low, the possibility that fracturing could lead to the formation of pathways for water flow and advective transport cannot currently be excluded. Selected radionuclide transport calculation cases cover the case of a disturbed buffer/rock interface due to the presence of iron corrosion products in contact with bentonite.
Displacement of supercontainer/buffer by swelling of distance blocks	N/A	See buffer.
Breaching of supercontainer shells by bentonite swelling	N/A	The supercontainer shell may be breached by the different forces due to bentonite swelling acting inside and outside the supercontainer shell (secondary effect, because the supercontainer has no safety function).

Deposition drift, central tunnel, access tunnel, shafts, boreholes

A major difference is in the geometry and backfilling of the KBS-3H deposition drifts compared with the KBS-3V deposition tunnels. In KBS-3H, supercontainers are emplaced along relatively narrow deposition drifts, separated by compacted bentonite distance blocks. In KBS-3V, deposition holes are bored from relatively large diameter deposition tunnels, backfilled with swelling clay or clay/crushed rock mixture.

For other underground openings (access tunnel, shafts, boreholes) no major differences have been identified.

Geosphere

System components/ (groups of) processes	KBS-3V	KBS-3H
Gas transport, gas-induced porewater displacement	Relevant only for failed canisters.	Limited storage volume and transport capacity within deposition drift, combined with increased gas generation (rates and total amount). Gas dissolution/diffusion/advection in groundwater, gas pressure build-up, gas-induced porewater displacement, capillary leakage. For tight canister sections: gas transport along drift (EDZ) to the next transmissive fracture, possibly involving reactivation of fractures in near-field rock, when minimal principal stress is exceeded.
Transmissive fractures and flow conditions	The selection of deposition hole locations is more flexible than in KBS-3H because rock sections with larger inflows can be rejected.	Local variations in groundwater flow conditions along the drift may lead to variable saturation time for the buffer along the drift.
Mechanical stability of the drift/tunnel	High stresses at the mouth of deposition holes and at the top of backfill tunnel.	Lower rock stresses than in KBS-3V because the deposition drifts can be better adapted to the stress field.
Orientation of fractures	KBS-3V is more sensitive to sub-horizontal than to sub-vertical fractures with respect to potential damage to the engineered barrier system by rock shear.	KBS-3H is more sensitive to sub-vertical fractures with respect to potential damage to the engineered barrier system by rock shear.
Biosphere, human activities		
No major differences identified		

Appendix B

Data and calculations for RTI and RTF

The information and data used to generate the radiotoxicity indices and radiotoxicity index fluxes of spent fuel, repository releases and natural materials shown in figures in Chapter 2 and 6 is presented in this appendix.

The radiotoxicity index or RTI(t) /Nagra 2002a, Hedin 1997/ is here defined as the hypothetical dose at time t resulting from ingestion of the activity A_j (t) [Bq] of radionuclide j, divided by 10^{-4} Sv (derived from the Finnish regulatory dose limit for the first several thousand years):

$$RTI(t) = \frac{\sum_j A_j(t) D_j}{10^{-4}\, Sv} \qquad \text{(Eq. B-1)}$$

where D_j [Sv/Bq] is the dose coefficient for ingestion.

B.1 Radiotoxicity index of spent fuel

The description of the radionuclide composition and activity of 1 tonne of Finnish spent BWR fuel (40 MWd/kgU, 4.2% enrichment) is given in /Anttila 2005/ at eight different times up to 1 million years after unloading from the reactors. The nuclide activity data are reproduced in Table B-1.

The large number of nuclides with half-lives short compared to the time over which the RTI is calculated means that there will be a significant decrease in RTI over time as shown in Figure 2-1 as these nuclides decay to insignificance. The dose coefficients for the radionuclides in the spent fuel are listed in Table B-2 based on /ICRP 1996/ and /Avila and Bergström 2006/.

The RTI for the spent fuel at the eight times calculated by /Anttila 2005/ are given in Table B-3. The RTI values for the full inventory on Figure 2-1 were obtained by multiplying the RTI for 1 tonne of spent fuel by 5,500 to account for the total amount of spent fuel planned for disposal in the repository /Pastina and Hellä 2006/ – the differences in RTI of different spent fuel types/burn-ups is not taken into account.

B.2 Radiotoxicity index of natural materials

The natural materials that are used in the radiotoxicity comparisons in Chapter 2 are natural uranium, uranium ores and the Olkiluoto tonalite-granodiorite, which is a component of the TGG (tonalitic-granodioritic-granitic gneisses) and is taken as a “representative host rock type” for its radionuclide content. The radionuclides of interest in these natural materials are U-235, U-238, Th-232 and K-40. The dose coefficients used for these radionuclides in natural materials is given in Table B-4. Note that the dose coefficients for Th-232, U-235 and U-238 include radioactive decay products (the daughters) as the parent nuclides are assumed to be in equilibrium with the daughters for solid materials.

Table B-1. Radionuclide activity (Gbq/tU) of 1 tonne spent nuclear fuel (Finnish BWR-type 40 MWd/kgU, 4.2% enrichment) at eight times (in years) after unloading from the reactor (after /Anttila 2005/).

Time (y)	0	5	30	100	1,000	10,000	100,000	1,000,000
Light elements								
H-3	1.04E+01	7.83E+00	1.92E+00	3.75E-02	3.97E-24	0.00E+00	0.00E+00	0.00E+00
C-14	2.79E+01	2.79E+01	2.78E+01	2.75E+01	2.47E+01	8.31E+00	1.55E-04	0.00E+00
Cl-36	1.14E+00	1.14E+00	1.14E+00	1.14E+00	1.13E+00	1.11E+00	9.03E-01	1.14E-01
Co-60	9.19E+04	4.76E+04	1.78E+03	1.78E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ni-63	1.90E+04	1.83E+04	1.54E+04	9.49E+03	1.87E+01	1.60E-26	0.00E+00	0.00E+00
Zr-93	7.49E+00	7.49E+00	7.49E+00	7.49E+00	7.49E+00	7.46E+00	7.16E+00	4.76E+00
Nb-93m	9.88E+02	8.00E+02	2.83E+02	2.82E+01	1.37E+01	8.50E+00	7.16E+00	4.76E+00
Nb-94	2.60E+01	2.60E+01	2.60E+01	2.59E+01	2.51E+01	1.85E+01	8.55E-01	3.85E-14
Mo-93	9.20E+00	9.19E+00	9.14E+00	9.02E+00	7.54E+00	1.27E+00	2.28E-08	0.00E+00
Total	2.14E+07	2.91E+05	1.81E+04	9.76E+03	2.42E+02	1.77E+02	7.40E+01	9.70E+00
Actinides								
Pb209	0.00E+00	0.00E+00	0.00E+00	3.28E-05	3.76E-03	4.80E-01	1.13E+01	2.80E+01
Pb210	0.00E+00	0.00E+00	0.00E+00	6.55E-04	1.30E-01	5.50E+00	4.34E+01	1.77E+01
Pb214	0.00E+00	0.00E+00	0.00E+00	1.18E-03	1.30E-01	5.50E+00	4.35E+01	1.77E+01
Bi210	0.00E+00	0.00E+00	0.00E+00	6.55E-04	1.30E-01	5.50E+00	4.34E+01	1.77E+01
Bi214	0.00E+00	0.00E+00	0.00E+00	1.18E-03	1.30E-01	5.50E+00	4.35E+01	1.77E+01
Po210	0.00E+00	0.00E+00	0.00E+00	6.55E-04	1.30E-01	5.50E+00	4.34E+01	1.77E+01
Po214	0.00E+00	0.00E+00	0.00E+00	1.18E-03	1.30E-01	5.50E+00	4.34E+01	1.77E+01
Po218	0.00E+00	0.00E+00	0.00E+00	1.18E-03	1.30E-01	5.50E+00	4.35E+01	1.77E+01
Ra226	0.00E+00	0.00E+00	0.00E+00	1.18E-03	1.30E-01	5.50E+00	4.35E+01	1.77E+01
Th229	0.00E+00	0.00E+00	0.00E+00	3.28E-05	3.76E-03	4.80E-01	1.13E+01	2.80E+01
Th230	0.00E+00	4.06E-03	1.63E-02	5.61E-02	7.12E-01	7.07E+00	4.30E+01	1.77E+01
Th234	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.17E+01
Pa233	1.09E+01	1.10E+01	1.15E+01	1.40E+01	3.09E+01	3.60E+01	3.50E+01	2.61E+01
Pa234m	1.17E+01	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.17E+01
U233	0.00E+00	8.91E-04	2.11E-03	5.99E-03	1.01E-01	1.47E+00	1.25E+01	2.77E+01
U234	4.90E+01	5.02E+01	5.59E+01	6.69E+01	8.16E+01	7.99E+01	6.46E+01	1.58E+01
U236	1.30E+01	1.30E+01	1.30E+01	1.30E+01	1.35E+01	1.64E+01	1.82E+01	1.77E+01
U238	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.16E+01	1.17E+01
Np237	1.08E+01	1.10E+01	1.15E+01	1.40E+01	3.09E+01	3.60E+01	3.50E+01	2.61E+01
Np239	5.24E+08	6.56E+02	6.55E+02	6.51E+02	5.98E+02	2.56E+02	5.41E-02	1.81E-06
Pu238	8.39E+04	8.82E+04	7.24E+04	4.17E+04	3.74E+01	2.24E-19	0.00E+00	0.00E+00
Pu239	8.79E+03	8.93E+03	8.93E+03	8.91E+03	8.70E+03	6.81E+03	5.20E+02	1.82E-06
Pu240	1.87E+04	1.87E+04	1.88E+04	1.87E+04	1.70E+04	6.56E+03	4.88E-01	6.10E-06
Pu241	3.62E+06	2.84E+06	8.50E+05	2.89E+04	3.37E+00	1.62E+00	1.05E-03	0.00E+00
Pu242	7.57E+01	7.57E+01	7.57E+01	7.57E+01	7.56E+01	7.43E+01	6.29E+01	1.18E+01
Am241	4.46E+03	3.01E+04	9.35E+04	1.09E+05	2.59E+04	1.71E+00	1.05E-03	0.00E+00
Am242m	2.28E+02	2.23E+02	1.97E+02	1.40E+02	1.67E+00	1.02E-19	0.00E+00	0.00E+00
Am242	2.35E+06	2.22E+02	1.96E+02	1.39E+02	1.67E+00	1.02E-19	0.00E+00	0.00E+00
Am243	6.56E+02	6.56E+02	6.55E+02	6.51E+02	5.98E+02	2.56E+02	5.41E-02	1.81E-06
Cm242	1.39E+06	7.74E+02	1.62E+02	1.15E+02	1.38E+00	8.43E-20	0.00E+00	0.00E+00
Cm243	4.85E+02	4.29E+02	2.34E+02	4.26E+01	1.33E-08	0.00E+00	0.00E+00	0.00E+00
Cm244	6.42E+04	5.32E+04	2.04E+04	1.40E+03	1.50E-12	0.00E+00	0.00E+00	0.00E+00
Cm245	3.65E+00	3.65E+00	3.65E+00	3.62E+00	3.37E+00	1.62E+00	1.05E-03	0.00E+00
Total	1.10E+09	3.05E+06	1.07E+06	2.10E+05	5.31E+04	1.42E+04	1.32E+03	5.72E+02

Time (y)	0	5	30	100	1,000	10,000	100,000	1,000,000
Fission products								
H3	2.51E+04	1.90E+04	4.66E+03	9.09E+01	9.63E-21	0.00E+00	0.00E+00	0.00E+00
Se79	3.33E+00	3.33E+00	3.33E+00	3.33E+00	3.32E+00	3.25E+00	2.63E+00	3.17E-01
Kr85	4.59E+05	3.32E+05	6.60E+04	7.14E+02	3.78E-23	0.00E+00	0.00E+00	0.00E+00
Sr90	3.64E+06	3.22E+06	1.74E+06	3.10E+05	7.35E-05	0.00E+00	0.00E+00	0.00E+00
Y90	3.80E+06	3.22E+06	1.74E+06	3.10E+05	7.35E-05	0.00E+00	0.00E+00	0.00E+00
Zr93	8.38E+01	8.39E+01	8.39E+01	8.39E+01	8.38E+01	8.35E+01	8.01E+01	5.33E+01
Nb93m	6.93E+00	2.18E+01	6.27E+01	8.28E+01	8.38E+01	8.35E+01	8.01E+01	5.33E+01
Tc99	6.17E+02	6.19E+02	6.19E+02	6.19E+02	6.17E+02	5.99E+02	4.46E+02	2.32E+01
Pd107	4.09E+00	4.09E+00	4.09E+00	4.09E+00	4.09E+00	4.08E+00	4.04E+00	3.67E+00
Cd113m	1.24E+01	1.00E+01	2.93E+00	9.36E-02	5.69E-21	0.00E+00	0.00E+00	0.00E+00
Sn126	2.14E+01	2.14E+01	2.14E+01	2.14E+01	2.13E+01	2.00E+01	1.07E+01	2.09E-02
Sb126	1.25E+04	3.00E+00	3.00E+00	3.00E+00	2.98E+00	2.80E+00	1.50E+00	2.93E-03
Sb126m	1.71E+04	2.14E+01	2.14E+01	2.14E+01	2.13E+01	2.00E+01	1.07E+01	2.09E-02
I129	1.11E+00	1.12E+00	1.12E+00	1.12E+00	1.12E+00	1.11E+00	1.11E+00	1.07E+00
Cs134	5.51E+06	1.03E+06	2.30E+02	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs135	1.97E+01	1.97E+01	1.97E+01	1.97E+01	1.97E+01	1.96E+01	1.91E+01	1.46E+01
Cs137	4.72E+06	4.21E+06	2.36E+06	4.69E+05	4.36E-04	0.00E+00	0.00E+00	0.00E+00
Ba137m	4.48E+06	3.97E+06	2.23E+06	4.42E+05	4.11E-04	0.00E+00	0.00E+00	0.00E+00
Ba140	4.91E+07	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pm147	7.06E+06	1.94E+06	2.62E+03	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Sm151	1.13E+04	1.10E+04	9.10E+03	5.31E+03	5.18E+00	4.22E-30	0.00E+00	0.00E+00
Eu152	2.47E+02	1.90E+02	5.19E+01	1.36E+00	6.41E-21	0.00E+00	0.00E+00	0.00E+00
Eu154	2.01E+05	1.34E+05	1.79E+04	6.30E+01	1.69E-30	0.00E+00	0.00E+00	0.00E+00
Eu155	1.10E+05	5.25E+04	1.29E+03	4.06E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Total	4.93E+09	2.03E+07	8.18E+06	1.54E+06	8.64E+02	8.37E+02	6.56E+02	1.50E+02

Table B-2. Dose coefficients ingestion for radionuclides in spent nuclear fuel, applicable to adults older than 17 years. (See text for source).

Nuclide	Dose coefficient Sv/Bq Adult >17 y	Nuclide	Dose coefficient Sv/Bq Adult > 17 y
H-3	1.8E-11	Ra-223	1.0E-07
Be-10	1.1E-09	Ra-224	6.5E-08
C-14	5.8E-10	Ra-225	9.9E-08
Cl-36	9.3E-10	Ra-226	2.8E-07
Ca-41	1.9E-10	Ra-228	6.9E-07
Co-60	3.4E-09	Ac-225	2.4E-08
Ni-59	6.3E-11	Ac-227	1.1E-06
Ni-63	1.5E-10	Th-227	8.8E-09
Se-79	2.9E-09	Th-228	7.2E-08
Sr-90	2.8E-08	Th-229	4.9E-07
Y-90	2.7E-09	Th-230	2.1E-07
Zr-93	1.1E-09	Th-231	3.4E-10
Nb-93m	1.5E-09	Th-232	2.3E-07
Nb-94	1.7E-09	Th-234	3.4E-09
Mo-93	3.1E-09	Pa-231	7.1E-07
Tc-99	6.4E-10	Pa-233	8.7E-10
Pd-107	3.7E-11	U-233	5.1E-08
Ag-108m	2.3E-09	U-234	4.9E-08
Sn-126	4.7E-09	U-235	4.7E-08
Sb-126	2.4E-09	U-236	4.7E-08
I-129	1.1E-07	U-238	4.5E-08
Cs-134	1.9E-08	Np-237	1.1E-07
Cs-135	2.0E-09	Np-239	8.0E-10
Cs-137	1.3E-08	Pu-238	2.3E-07
Sm-151	9.8E-11	Pu-239	2.5E-07
Eu-152	1.4E-09	Pu-240	2.5E-07
Eu-154	2.0E-09	Pu-241	4.8E-09
Eu-155	3.2E-10	Pu-242	2.4E-07
Ho-166m	2.0E-09	Am-241	2.0E-07
Pb-210	6.9E-07	Am-242m	1.9E-07
Bi-210	1.3E-09	Am-243	2.0E-07
Po-210	1.2E-06	Cm-244	1.2E-07
Rn-222	3.5E-09	Cm-245	2.1E-07
		Cm-246	2.1E-07

Table B-3. RTI of 1 tonne of spent nuclear fuel (as Table B-1) at eight times (years) after unloading from the reactor /from Anttila 2005/.

Cooling time (y)	RTI Fission Products	RTI Actinides	RTI Activation Products	Overall RTI
1	2.80E+12	4.72E+12	3.17E+09	7.52E+12
5	1.74E+12	5.34E+11	1.66E+09	2.28E+12
30	8.45E+11	4.9E+11	8.87E+07	1.33E+12
100	1.57E+11	3.87E+11	1.56E+07	5.44E+11
1,000	8.92E+06	1.18E+11	1.13E+06	1.18E+11
10,000	8.72E+06	3.43E+10	6.22E+05	3.44E+10
100,000	7.15E+06	2.63E+09	2.09E+05	2.64E+09
1,000,000	3.01E+06	6.51E+08	1.25E+05	6.54E+08

Table B-4. Dose coefficients for naturally-occurring radionuclides (ingestion for adults older than 17 years).

Nuclide	Dose coefficient (Sv/Bq)
K-40 [1]	6.20×10^{-9}
Th-232 [2]	1.06×10^{-6}
U-235 [3]	1.97×10^{-6}
U-238 [4]	2.43×10^{-6}

[1] /Nagra 2002a/

[2] Dose coefficients for Th-232 +Ra-228 + Th-228 + Ra-224 from Table B-2

[3] Dose coefficients for U-235 + Th-231+ Pa-231+ Ac-227+ Th-227 + Ra-223 from Table B-2

[4] Dose coefficients for U-238 +Th-234 +Th-230 + Ra-226 + Rn-222 + Po-210 + Pb-210 from Table B-2

Figure 2-2 shows the RTI of the 8 tonnes of natural uranium from which the original fuel was derived. The details of this calculation are shown in Table B-5.

Table B-6 shows the radionuclide content of the natural uranium ores and the Olkiluoto tonalitic and granitic rocks along with the activity levels and the resultant RTI per m^3 of ore or rock. The details of the U and Th content of the Olkiluoto rocks and the Palmottu ore are from /Pitkänen et al. 2003/.

The RTIs for the natural materials shown in Figure 2-1 were based on the assumption that the deposition drifts of the KBS-3H repository were filled with the uranium ore. The volume used for this calculation was 2.46×10^5 m^3 /Autio et al. 2007/. This was also the figure used for the volume of Olkiluoto tonalite-granodiorite removed during excavation of the same drifts.

Table B-5. Activity and radiotoxicity indices of natural uranium /Nagra 2002a/.

Nuclide	kg element per kg material	Bq precursor per kg material	RTI per kg material	RTI per 8 tonnes uranium
U-235	1	5.6×10^5	1.1×10^4	8.8×10^7
U-238	1	1.2×10^7	3.1×10^5	2.5×10^9
Total			3.2×10^5	2.6×10^9

Table B-6. Activity and RTI of natural U ores and Olkiluoto rocks.

Nuclide	kg element per kg rock	Bq precursor per kg rock	RTI per kg rock	RTI per m^3 rock
		Cigar Lake U ore 55% (density 5,800 kg/m^3) [1]		
U-235	0.55	3.1×10^5	6.1×10^3	3.5×10^7
U-238	0.55	6.8×10^6	1.7×10^5	9.9×10^8
Total			1.8×10^5	1.0×10^9
		Cigar Lake U ore 8% (density 2,800 kg/m^3) [1]		
U-235	8.0×10^{-2}	4.5×10^4	8.8×10^2	2.5×10^6
U-238	8.0×10^{-2}	1.0×10^6	2.5×10^4	6.9×10^7
Total			2.4×10^4	7.1×10^7
		Palmottu ore 1,600ppm (density 2,700 kg/m^3) [2]		
U-235	1.6×10^{-4}	9.1×10^1	1.8	4.9×10^3
U-238	1.6×10^{-4}	2.0×10^3	4.9×10^1	1.3×10^5
Total			5.1×10^1	1.3×10^5
		Olkiluoto granite (density 2,600 kg/m^3) [2]		
K-40	2×10^{-2}	7.9×10^2	4.9×10^{-2}	1.3×10^2
Th-232	1.4×10^{-5}	5.7×10^1	6.3×10^{-1}	1.6×10^3
U-235	5×10^{-6}	2.8	5.7×10^{-2}	1.5×10^2
U-238	5×10^{-6}	6.2×10^1	1.5	4.0×10^3
Total			2.2	5.9×10^3
		Olkiluoto tonalite-granodiorite (density 2,600 kg/m^3) [2]		
K-40	2.5×10^{-2}	7.9×10^2	4.9×10^{-2}	1.3×10^2
Th-232	5.8×10^{-6}	2.4×10^1	2.6×10^{-1}	6.8×10^2
U-235	1.6×10^{-6}	9.2×10^{-1}	1.8×10^{-2}	4.7×10^1
U-238	1.6×10^{-6}	2.0×10^1	4.9×10^{-1}	1.3×10^3
Total			8.2×10^{-1}	2.2×10^3

[1] /Nagra 2002a/

[2] /Pitkänen et al. 2003/

B.3 Radiotoxicity index fluxes of repository releases and natural materials

An alternative indicator for hazard through ingestion is the radiotoxicity flux or RTF across a given interface (units: RTI/y, Nagra 2002a), which is defined by replacing the activity $A_j(t)$ in Eq. B-1 by the annual activity flux $F_j(t)$ across that interface (see Eq. B-2). The RTF can be used for direct comparison of radiological hazard of activity fluxes from the repository with that from natural activity fluxes.

$$RTF(t) = \frac{\sum_j F_j(t) D_j}{10^{-4} Sv} \qquad \text{(Eq. B-2)}$$

The releases from the repository are calculated in the form of a flux across the geosphere – biosphere boundary so are easily converted to the form of RTF by calculating the radiotoxicity index of the radionuclides released from the geosphere each year. This is the RTF plotted in Figure 6-3, both for the total annual release with time and for the main contributing radionuclides. The dose coefficients used for this calculation are those given in Table B-2.

The data and calculations of RTF for the groundwater fluxes shown in Figure 6-3 are given in Table B-8 but in this case the dose coefficients in Table B-7 for the naturally-occurring nuclides

Th-232, U-235 and U-238 do not include the full suite of daughter nuclides as, in the aqueous system, it is assumed that there will no longer be equilibrium between the parent and daughter nuclides due to chemical processes which can partition elements with different behaviour. Thus the dose coefficient for Th-232 includes only that for daughter Th-228 as it is assumed that isotopes of a heavy element are not chemically partitioned. Likewise the dose coefficient for U-238 includes only U-234.

The RTF for municipal drinking water on Figure 6-3 was based on the dose calculation shown in Table 6-1, which used the dose coefficients given in Table B-7 and data from /Pitkänen et al. 2003/ for the content of radionuclides in the drinking water.

Table B-7. Dose coefficients for naturally-occurring radionuclides dissolved in water (ingestion for adults older than 17 years).

Nuclide	Dose coefficient (Sv/Bq)
K-40 [1]	6.20×10^{-9}
Rn-222 [2]	3.5×10^{-9}
Ra-226 [2]	2.8×10^{-7}
Th-232 [3]	3.0×10^{-7}
U-235 [4]	4.7×10^{-8}
U-238 [5]	9.4×10^{-8}

[1] /Nagra 2002a/
[2] Dose coefficient from Table B-2
[3] Dose coefficients for Th-232 + Th-228 from Table B-2
[4] Dose coefficients for U-235 from Table B-2
[5] Dose coefficients for U-238 + U-234 from Table B-2

Table B-8. Concentration, fluxes and radiotoxicity of natural radionuclides in groundwater at the Olkiluoto.

Nuclide	Mean concentration (µg/L)	Elemental fluxes (kg/km²/y)	Activity flux (Bq/km²/y) [1]	RTI per km² per year (RTF)
Olkiluoto upper deep (–375 m) reference plane. Water flow rate: 1,680 m³/km²/y				
U	0.208	0.35×10^{-3}	8.9×10^{3}	2.2×10^{2}
Th	0.037	0.062×10^{-3}	5.1×10^{2}	5.6
Ra-226	5.4 Bq/L	9.1×10^{6} Bq/km²/y	9.1×10^{6}	2.6×10^{4}
Rn-222	52.5 Bq/L	89×10^{6} Bq/km²/y	89×10^{6}	3.1×10^{3}
K	10,100	17.0	5.2×10^{5}	3.2×10^{1}
Total				2.9×10^{4}
Olkiluoto Northern surface discharge area. Water discharge rate: 1.08×10^{5} m³/km²/y				
U	0.67	72.5×10^{-3}	1.8×10^{6}	4.6×10^{4}
Rn-222	24 Bq/L	26×10^{9} Bq/km²/y	2.6×10^{10}	9.1×10^{5}
K	6,710	726	2.2×10^{7}	1.4×10^{3}
Total				9.8×10^{5}
Olkiluoto Southern surface discharge area. Water discharge rate: 2.0×10^{5} m³/km²/y				
U	3.71	741×10^{-3}	1.9×10^{7}	4.7×10^{5}
Rn-222	267 Bq/L	53.3×10^{9} Bq/km²/y	5.3×10^{10}	1.9×10^{6}
K	7,130	1,420	4.3×10^{7}	2.7×10^{3}
Total				2.3×10^{6}

[1] Elemental specific activities for the natural isotopic composition (Bq/g): K: 30.4, U: 2.53×10^{4}, Th: 8.14×10^{3}

The RTF shown in Figure 6-3 for the erosion of rock from the 1 km^2 above the repository was based on rock nuclide (U, Th) contents given in /Pitkänen et al. 2003/. The erosion rate was taken as 10 m in 10^6 years thus, over the 1 km^2; this is equivalent to 10 m^3 of rock per year. A "mixed" rock type composed of 60% muscovite-gneiss migmatite, 20% granite and 20% tonalite-granodiorite was assumed, based very approximately on surface exposure (ignoring current superficial deposits). The calculation of RTF is shown in Table B-9.

Table B-9. **Radiotoxicity flux arising from erosion of U,Th-bearing rocks over 1 km^2 at the surface above the Olkiluoto repository.**

Nuclide	kg element per kg rock	Bq per kg rock	RTI per kg rock	RTF per 10 m^3 rock per y
Mixed rock: 60% muscovite-gneiss migmatite, 20% granite, 20% tonalite-granodiorite (density: 2,600 kg/m^3)				
Erosion rate: 10 m per 10^6 years = 10 m^3 per km^2 per year (see Section 4.4.1)				
Uranium	3.84	9.73×10^4	2.43×10^3	6.31×10^7
Thorium	9.89	8.08×10^4	8.85×10^2	2.30×10^7
Total			3.31×10^3	8.61×10^7

Appendix C

Main differences between the swedish and the finnish regulatory system

The KBS-3H safety studies are based on the Finnish regulation YVL 8.4. Swedish regulations relevant to the long-term safety of nuclear waste repositories are described in Section 1.4 and in Appendix A of SR-Can Main Report /SKB 2006/. The two more detailed regulations are:

- The Swedish Radiation Protection Institute (SSI) Regulations concerning the Protection of Human Health and the Environment in connection with the Final Management of Spent Nuclear Fuel or Nuclear Waste (SSI FS 1998:1) /SSI 1998/.
- The Swedish Nuclear Power Inspectorate (SKI) regulations concerning safety in final disposal of nuclear waste (SKIFS 2002:1) /SKIFS 2002/.

SSI has also issued a guidance document concerning the application of SSI FS 1998:1 which gives more detailed information regarding the above /SSI 2005/. These regulations and guidance are reported in their entirety in Appendix A of SR-Can Main Report /SKB 2006/.

The most relevant differences between the Swedish regulations and the Finnish regulation YVL 8.4 /STUK 2001/ are the following:

- According to the Swedish regulations, protection of human health shall be demonstrated by compliance with a risk criterion that states "*the annual risk of harmful effects after closure does not exceed 10^{-6} for a representative individual in the group exposed to the greatest risk*". Harmful effects refer to cancer and hereditary effects. The risk limit corresponds, according to SSI, to a mean annual dose constraint of about 1.4 x 10^{-5} Sv/yr. This, in turn, corresponds to around one percent of the natural background radiation in Sweden. The Finnish regulation has no equivalent risk criterion.
- In Sweden, SSI's guidance requires that the quantitative risk criterion be applicable for approximately 100,000 years with a more detailed assessment for the first 1,000 years following repository closure. Finnish regulations do not give a prescribed time frames for regulatory compliance but distinguish between the "environmentally predictable future" (lasting "several thousand years"), during which conservative estimates of dose must be made, and the "era of large-scale climate changes" when periods of permafrost and glaciations are expected, and radiation protection criteria are based on constraints on nuclide-specific activity fluxes from the geosphere.

Finnish regulation (YVL 8.4) includes nuclide-specific constraints for the activity releases to the environment as follows:

- 0.03 GBq/y for the long-lived, alpha emitting radium, thorium, protactinium, plutonium, americium and curium isotopes.
- GBq/y for the nuclides Se-79, I-129 and Np-237.
- 0.3 GBq/y for the nuclides C-14, Cl-36 and Cs-135 and for the long-lived uranium isotopes.
- 1 GBq/y for Nb-94 and Sn-126.
- 3 GBq/y for the nuclide Tc-99.
- 10 GBq/y for the nuclide Zr-93.
- 30 GBq/y for the nuclide Ni-59.
- 100 GBq/y for the nuclides Pd-107 and Sm-151.

These constraints apply to activity releases which arise from the expected evolution scenarios and which may only reach the environment after several thousand years. These activity releases can be averaged over a period of up to 1,000 years. The sum of the ratios of the nuclide-specific activity releases and the respective constraints shall be less than one /STUK 2001/.

No such nuclide-specific constraints for the activity releases to the environment are given in the Swedish regulations. In SR-Can, SKB uses the Finnish activity release constraints as alternative safety indicators.

Section 2.4 of YVL 8.4 states that, whenever practicable, estimates of the probabilities of activity releases and radiation doses arising from unlikely disruptive events impairing long-term safety should be made. These probabilities should be multiplied by the calculated annual radiation dose or activity in order to evaluate the importance to safety of an event. In order to satisfy regulatory requirements, the expectation value should remain below the radiation dose or activity release constraints (as given above). If, however, the resulting individual dose implies deterministic radiation impacts (dose above 0.5 Sv), the order of magnitude estimate for its annual probability of occurrence should be 10^{-6} at the most.

The differences between the Swedish and the Finnish regulatory systems imply that additional steps are undertaken in the Swedish SR-Can's safety assessment structure and methodology. These steps are described in Chapter 2 of SR-Can Main Report /SKB 2006/. In particular, the scenario selection and disaggregation (see below) as well as the risk summation steps in SR-Can are not undertaken in the KBS-3H safety studies because they derive from the risk criterion in the Swedish regulatory system.

C.1 Scenario selection and disaggregation in SR-Can

The following text is based on the description of scenario disaggregation in Section 2.9.2 in SR-Can Main Report /SKB 2006/. In principle, the product of dose consequences and likelihood of all possible future evolutions of the repository should be weighed together and presented as a time-dependent risk. The spectrum of possible evolutions is, however, very wide and cannot be captured in a detailed sense. This is also recognised in SSI's regulations and associated general guidance.

The usual approach taken in safety assessments, and also in SR-Can, is to work with scenarios and variants that are designed to capture the broad features of a number of representative possible future evolutions. Together, these are intended to give a reasonable coverage of possible future exposure situations. Conditional risks are calculated for each scenario and variant and these are then weighed together using the probability for each scenario/variant. Furthermore, each variant, represented by a specific calculation case, may be evaluated probabilistically in order to determine the mean exposure given the data uncertainties for the particular variant. The approach of calculating risk as a weighted sum over a number of scenarios constrains the way in which scenarios are selected and defined. It must be possible to explain logically the determination of probabilities.

In short, the scenarios should be mutually exclusive and the set of scenarios comprehensive in the sense that all relevant future evolutions are covered. A "normal evolution" scenario with a high probability of occurrence must, for example, contain initially defective canisters and other barrier insufficiencies, if such are likely when the entire ensemble of canisters and deposition holes in the repository is considered. Furthermore, in evaluating less likely scenarios treating disruptive events during the course of repository evolution, the consequences of these need to be superimposed on those of the normal evolution scenario. This does not mean that the calculation case for the latter must include also the normal evolution but it must be possible to superimpose the two in order to correctly represent the disruptive scenario in the final risk calculation.

In practice, after a reference evolution of the repository system is defined and analysed, a set of scenarios for the assessment is selected in accordance with SKI regulations SKIFS 2002:1. The main scenario is closely related to the reference evolution. The selection of additional scenarios is focused on the safety functions of the repository and the safety function indicators form an important basis for the selection. For each safety function, an assessment is made as to whether any reasonable situation where it is not maintained can be identified. If this is the case, the corresponding scenario is included in the risk evaluation for the repository with the overall risk determined by summation over such scenarios. The set of selected scenarios also includes e.g. scenarios explicitly mentioned in applicable regulations, such as human intrusion scenarios, and scenarios and variants to explore the roles of various components in the repository.

A similar methodology is followed in the scenario selection in the KBS-3H safety studies, as described in the Radionuclide Transport Report /Smith et al. 2007/. The KBS-3H scenario selection methodology and the "difference analysis approach" undertaken for the KBS-3H safety studies (see Section 1.3.1) result in a more limited set of scenarios and calculation cases than that in SR-Can; the scenarios and calculation cases selected have the purpose to investigate the differences between a KBS-3H and a KBS-3V repository.

To assess compliance with the Swedish regulatory risk criterion, in SR-Can each scenario is classified as "main", "less probable" and "residual scenario". The main scenario is based on a realistic initial state of the repository and a credible evolution of external conditions over the assessment period. The main scenario is split in two variants: the "base" variant and the "greenhouse" variant, the latter addressing a warmer climate evolution. Scenarios leading to the loss of safety functions of the repository guide the selection of less probable and residual scenarios. Less probable scenarios are those for which there is an appreciable probability of occurring and they are included in the risk summation. For the main scenario and less probable scenarios, risk contributions are estimated. The SSI regulations provide a risk conversion factor between effective dose and risk of 0.073 Sv^{-1}, based on ICRP's probability coefficient for cancer and hereditary effects. An annual risk limit of 10^{-6} thus corresponds to an effective dose limit of about 1.4 x 10^{-5} Sv/yr. The risk contributions from the main scenario and less probable scenario are summed up to calculate the overall risk. The result of this risk summation is shown in Figure C-1. No such figure is included in the KBS-3H safety study because of the absence of the risk criterion in the Finnish regulation YVL 8.4.

Other scenarios are classified as "residual" and are not included in the risk summation. Risk contributions from independent scenarios are added if combinations do not lead to higher consequences than the individual scenarios. Since SSI's general guidance states that the risk criterion concerns a repository undisturbed by man, scenarios involving direct intrusion into the repository are excluded from the risk summation. Also human actions that disturb the immediate environment of the repository, e.g. the local groundwater flow field, are considered in the treatment of future human actions but excluded from the risk summation.

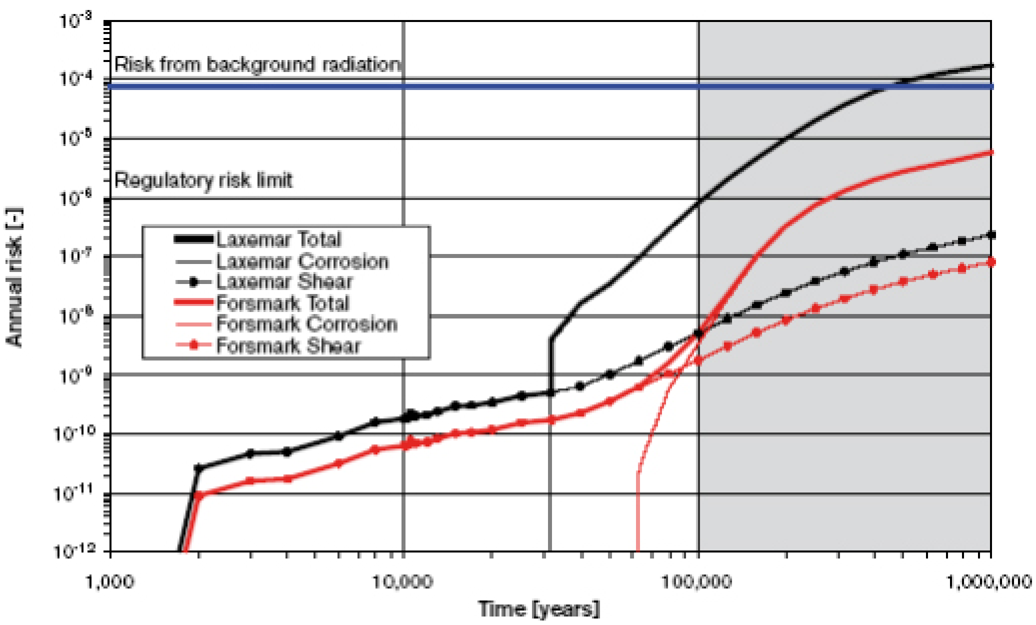

Figure C-1. *Risk summation, expressed as annual individual risk for the Laxemar and Forsmark sites in Sweden. (Figure 12-20 in SR-Can Main Report, /SKB 2006/).*

The main scenario identified in SR-Can is canister failure due to advection/corrosion. Another scenarios included in the risk summation was buffer advection. Canister failure due to shear movement was included in the risk summation, weighted by a low probability factor. Buffer freezing and buffer transformation (e.g. illitisation) were not propagated to the main scenario and defined "residual" scenarios. Canister failure due to isostatic collapse was excluded from the risk calculation due to its low probability.

From the total annual risk curves (Figure C-1), buffer colloid release/erosion process, which could occur when the buffer is exposed to glacial melt waters of low ionic strength, is the main factor affecting the calculated risk at the two Swedish sites (in particular at Laxemar).

C.2 References

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