4 Groundwater in hard rock – a literature review[1]

4.1 Introduction

Groundwater is the water that fills pores, fractures, and holes in soil and rock and which has a pore pressure that is higher or equal to the atmospheric pressure. Groundwater is the underground part of the water cycle in nature. The groundwater's volume and composition and its flow and flow paths are of principal importance for storage of nuclear waste, partly in a short term time span during construction and operations modes, partly in a long term time span for solution and transport of different substances, and partly for the delay and/or establishment of radio nuclei after possible damage to the depony. Knowledge about the groundwater conditions and the geological, hydrological, and chemical processes that effect the groundwater is therefore necessary for an optimal localisation and construction of a depony as well as for the long-term safety of this depony.

The following chapters will only cover the occurrence, flow, and nature of groundwater in hard rock, since the dictated method in Sweden for deposit of nuclear waste anticipate this type of bedrock. Crystalline rock, such as magmatic (from molten rock) and metamorphic rock (transformed), which are all or partly recrystallized rock types. Even though the silica-rich (acid) rock types are most common, for example different types

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of granites and acid gneisses, the silica-poor (basic) crystalline rocks such as diabase can locally be of great hydrogeologic importance. However, the term hard rock has a wider meaning and incorporates also the hard sedimentary and volcanic rock types, volcanic tuff among others. Volcanic tuffs are those rock types that are cemented to a degree that the hydrogeological meaning of the original porosity in the rock has lessened and the major part of the groundwater flow occurs in secondarily formed structures, mostly fractures, in the rock (Krásný, 1996b). Crystalline rock types on the other hand, can in superficial parts and in some zones be heavily porous and ”soft,” such as after far gone weathering, which is common in warmer climates. In Scandinavia, this weathering zone is often non-existent or very thin. Weathered crystalline rock occurs sometimes in fractures and in terrain locally protected from erosion. The chapter will not deal with the pore spaces that are created through chemical dissolution of limestones, so called karst, although rock types such as limestone can appear massively with low porosity. Karst hydrology is a very special knowledge area, which will not be dealt with here.

The chapter is a compilation of the great amount of knowledge that has appeared in recent years in science literature, sectorial report series (such as SKB’s report series) as well as experiences from construction projects. The International Association of Hydrogeologists (IAH) has three groups specialising in groundwater in hard rock where one group studies Scandinavia. Some more general compilations have also been produced. Lloyd (1999) deals with groundwater in hard rock in arid and semi-arid (dry) climates.

The chapter goes through in a relatively concise way today's knowledge on this area and some parts in question are not covered. Some areas such as mathematical modelling, investigation and sampling methodology, and groundwater chemistry should be separately investigated and reported.

The texts and figures are intended for readers who already have some knowledge, but who want to learn more. All the technical terms are explained in the text or in the glossary at the end of chapter 4.13. For other definitions or further reading, please see references in chapter 4.12.2.

4.2 Water in hard rock – different approaches

Groundwater appears in hard rock in different forms of pores. These pores are either primarily formed during the formation of the bedrock (for example, as pores between grains) or secondarily through later processes such as weathering, dissolution of minerals, or tectonics (deformation).

Depending on tradition, knowledge base, and practical experience, the scientific society sees availability of water and water flow in rock from different perspectives:

- Genetical tectonical perspective;
- Empirical descriptive perspective;
- Analytical mathematical perspective.

These perspectives often go hand in hand, but in many cases at least one is the dominant one.

In hard rock, the secondarily made porosity and flow guiding structures have as a rule mainly been created by tectonic processes such as folding, mineral orientation (foliation), fracturing and faults (displacement along fractures). In the genetical tectonical perspective, the groundwater conditions are assumed to generally be explained by the knowledge of the genesis of these structures (Larsson (ed), 1984, among others). This approach gives, above all, the possibility of achieving a qualitative analysis of groundwater conducting structures as well as developing capabilities of the set up of hydrological conceptual models.

An empirical and descriptive perspective describes the flow situation as it actually occurs without further analysis of the genetic conditions. For larger underground projects, mapping of structures and water inflows without focus on genetic conditions is usually made. Compilations of different geological conditions where water flow appears in hard rock have been done by, among others, Carlsson & Olsson (1977), Palmqvist (1990), and for the superficial rock by Olofsson (1993).

The analytical mathematical perspective has been dominant during the last decades. It is often based on direct hydraulic tests in boreholes and wells. These measurements are then the base for analytical flow calculations or mathematical modelling.

The first two perspectives have required good geological knowledge while the analytical calculations and mathematical modelling presume good understanding of physics and mathematics. Experiences from today (Olsson, 2000) have shown that it is important that analytical solutions and mathematical modelling are based on thorough geologic knowledge and hydrogeologic conceptual models in order to evaluate measurements and plan new ones, as well as to determine reasonable boundary values for the mathematical models.

4.3 Recharge of groundwater

4.3.1 The concept of groundwater recharge in rock

Groundwater recharge is defined as the downward directed flow of water that reaches the groundwater table in the system. Only a few studies have been conducted in the aim to directly study groundwater recharge in hard rock. Compilations by Olofsson (1994) and Bockgård (2000) are available. Knowledge of the amount of groundwater that is available for recharge is important information for the calculation of how much water that can continuously be extracted from rock wells as well as for calculation of the impact on groundwater conditions in soil and rock by for example tunnel construction. Furthermore, this information is important for the calculation of contamination spreading from soil to rock groundwater.

Groundwater recharge can be *direct*, formed through infiltrating precipitation, or *indirect* such as inflow from lakes and streams. In the humid climate area the direct groundwater recharge is dominant, while the indirect groundwater recharge is dominant in semiarid and arid areas, Figure 1.

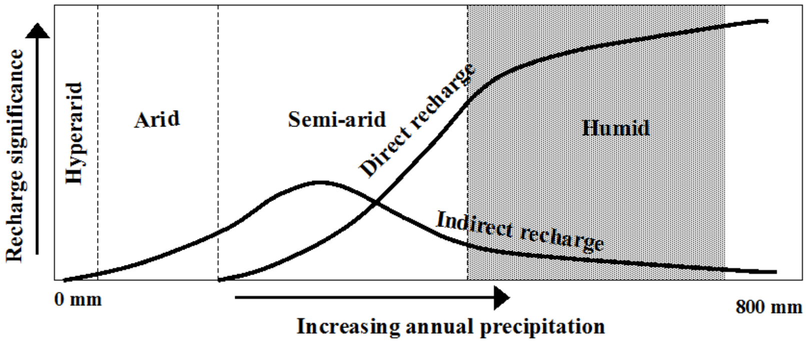

Figure 1. Principle-based diagram of the share of direct and indirect groundwater recharge in different climate areas. Scandinavia is marked in grey (based on, among others, Lloyd, 1994 and corrected for the humid, cold-temperate climate zone)

The dominating reason for groundwater flow is gravitation, which generates water flows from higher situated areas to lower. This is why the landscape in the humid climate area generally can be divided into *recharge* and *discharge* areas. The pattern is very complex, mainly due to the geology and variations in weather conditions. Groundwater recharge can in principle only happen when the flow gradient is directed downwards.

The groundwater recharge to rock is strongly dependent upon if the conditions are undisturbed groundwater conditions, that is, natural groundwater flow, or if the conditions are disturbed, such as is the case around underground constructions or pumping from wells. Most studies of groundwater recharge in rock have been made in conjunction with disturbances, for example pumping, which have accelerated and in some cases totally changed the natural flow conditions. A pumping test can therefore at best give an indication as to what the potential recharge is, while the real natural recharge is more difficult to quantify.

Another very important factor that affects groundwater recharge is whether water-saturated soil layers occur on top of the bedrock or if this bedrock is bare. In the former case it is not really the groundwater recharge to rock that is studied since groundwater most of the times already exists in the soil layers. Rather, it is the groundwater flow from soil to rock that is studied. For simplicity, no distinction will be made between these two concepts, since the soil layers can periodically be water saturated. Therefore, groundwater recharge to rock will include recharge of rock groundwater irrespective of the origin of the water.

The amount of precipitation and its distribution in time and space determine as a rule the size of the groundwater recharge. The part of the precipitation that can maximally be added to the groundwater (called net precipitation) can generally be calculated as the difference between the total precipitation in an area and the actual evaporation (which includes transpiration from soil organisms and plants). The net precipitation can run off on the land surface and/or infiltrate down into the soil. Direct surface runoff is during normal weather conditions not very usual in Sweden except for in discharge areas and areas of water saturated peat and clay. Direct surface runoff is common during heavy rainstorms in dry climate areas. In Sweden, a measure of the real groundwater recharge and surface runoff is given by the runoff in streams from a hydrologically well defined area *(drainage basin)*. A large part of this runoff water has on its way to the stream infiltrated and flowed through the ground. In flat and soil covered areas, a big part of the net precipitation will infiltrate but most of the infiltrated water will not reach the bedrock. Instead, it flows through the soil layers to close located springs, streams, and other discharge areas. In hilly regions with clearly defined recharge and discharge areas and quick groundwater flows, the infiltrated amount of net precipitation is smaller but the groundwater flow can reach deeper. The part of groundwater in surface streams has been studied with the help of oxygen isotopes, see for example (Rodhe, 1987) and showed to be 50–70 % even during heavy rains and snowmelts in tempered climates with permeable soils (Knutsson & Morfeldt, 1995).

4.3.2 Groundwater recharge in outcrop areas

In areas with many outcrops, the conditions are specific since a bare rock surface easily produces surface runoff. On archipelago islands where surface runoff can directly be transferred to the sea, the difference between net precipitation and real groundwater recharge can be considerable. In a study made in Sweden in the 1970's, outcrops were sprinkled with water. The study proved that more than 20 % of the amount of sprinkled water could find its way down into fractures in the rock (Bergman, 1972). Similar studies, although more focused on the regional scale under natural conditions in a granitic area in Canada, showed that the groundwater recharge in the outcrop area was only 5mm/yr or 1 % of the precipitation (Thore & Gascoyne, 1993). The direct infiltration into fractures on outcrops is very dependent on the fracture configuration, the fracture width and fracture fillings with mineral precipitation and soil. The superficial fractures can vary in width from microfractures up to a decimetre. These fractures often give a microtopography on the rock surface to where the runoff water flows. Open fractures are at the surface covered with soil and organic material. Water flow down such fractures can only occur if the soil material is at least partly saturated with water. Groundwater recharge to rock fractures can, however, occur during the year around since the small volumes of fractures quickly get saturated with water.

Steep open fractures favour the infiltration down into the rock. In order for the groundwater recharge to continue as the fracture fills up, a flow of the fracture water from the fracture is needed. The superficial groundwater flow is often favoured by flat rock fractures. These can be water collecting and are often open in the superficial bedrock, Figure 2. They can be a result of pressure release and temperature tensions (sheet joints) or may appear as flat laying shear fractures, often filled with soil material. Flat rock fractures appear generally in the upper parts of the rock mass (down to ten or twenty meters) (Olofsson, 1994). Water distribution in a specific fracture is determined by the pressure conditions in the fracture (the hydraulic gradient) and the other hydraulic properties of the fracture and its filling material.

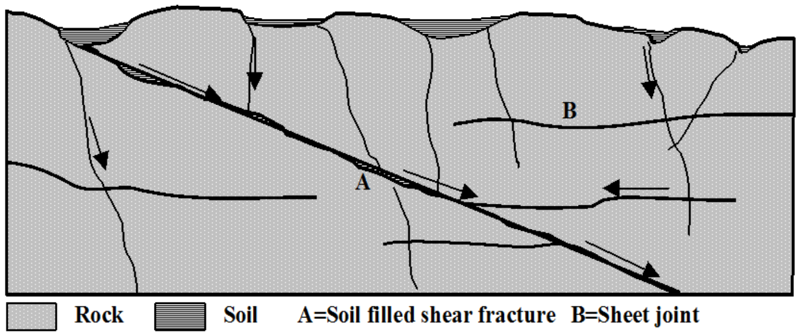

Figure 2. Example of flat and superficial draining rock structures

4.3.3 Groundwater recharge from soil to rock

A great deal of knowledge on flow processes from soil to rock has arisen from studies made in conjunction with disturbances of the flow conditions through pumping or construction in rock (Olofsson, 1991, 2000; Olsson, 2000). Some more detailed studies have also been carried out in Sweden that focus on the flow conditions in the contact zone between soil and rock (Olofsson, 1994).

The contact zone between soil and rock can be extremely varying depending on climate conditions and formation processes. In non-glaciated areas there is often a gradual transition between the soil layer, which is often locally formed *(in situ)*, via a heavily weathered and clayey zone, to a border zone with fractures and high porosity in the rock mass, down to the massive bedrock, where only the fracture porosity is of hydraulic importance, Figure 3.

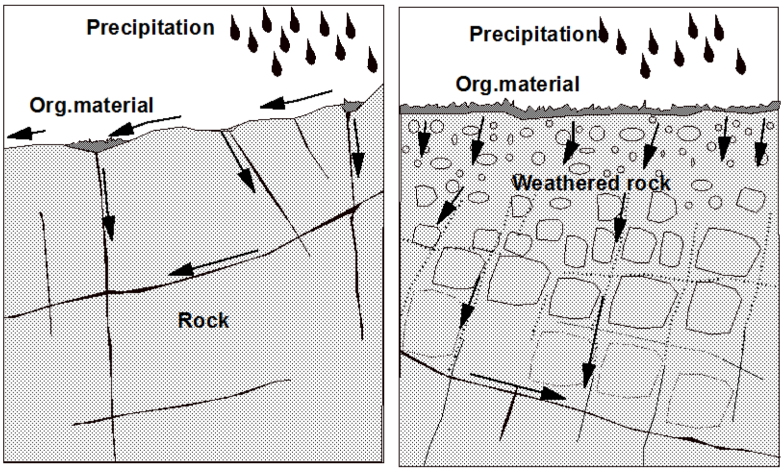

Figure 3. *Groundwater recharge in hard rock without soil layers as well as with residual deposits.*

The transition zone can often be tens of meters deep. The size of the groundwater recharge in such areas with hard rock, for example in Zimbabwe, Burkina Faso and Western Australia, have given values between 0,05-6,5 % of the precipitation (Lerner et al, 1990). In Scandinavia, Canada, and other regions with recently glaciation, the border zone often consists of a distinct border between a bedrock dominated by fracture porosity and more or less permeable soil layers.

Groundwater flow from soil to rock can only appear if permeable soil layers or permeable horizons in the soil are hydraulically connected to open or partly open structures in the rock (Olofsson, 1994). A thin layer of silt on the rock surface can effectively block the water flow (Knutsson, 1971). If the soil layers only consist of permeable, sorted sand and gravel, for example a glaciofluvial deposit that often is directly in contact with the rock surface, the flow to the rock will generally be channelled by the rock structures and their permeabilities. If, on the other hand, the soil layer consists of till, which is a type of glacial deposit dominant in Sweden the flow from soil to rock will be directly dependent on the till's grain-size distribution, structure, hydraulic heterogeneity and anisotropy, Figure 4.

The probability that a hydraulic connection may arise diminishes with increasing degree of hydraulic heterogeneity in the soil and the bedrock. Boulder-rich gravelly till may favour flow to the rock. Clayey till and clay may sometimes totally block the flow to the bedrock. It is therefore imperative to study the hydraulic characteristics of the soil layers in order to be able to make estimates on groundwater recharge in rock.

If the soil layers are thick and at depth saturated with water the year around, the flow from soil to rock may continue the year around with slight variations depending on the groundwater levels. Water saturated conditions may periodically be lacking in thin soils. During such conditions, groundwater recharge from soil to rock generally occurs during late fall and spring, when the soil water storage is refilled.

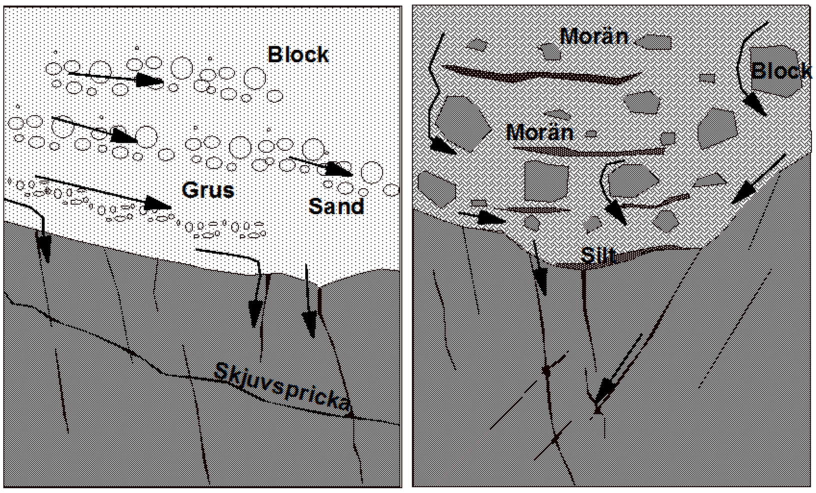

Figure 4. Example of flow from soil to rock in a glaciofluvial deposit (left; block=bolder, grus=gravel; skjuvsprica=shear fracture) as well as in till (right; morän=till)

4.3.4 Exchange with surface waters

In some cases a hydraulic connection may arise between rock groundwater and surface water bodies, such as lakes and streams. In arid and semi-arid climates such as on the Arabian Peninsula and in Southwest United States, groundwater recharge is mainly due to indirect infiltration. Intensive rain storms under short periods of time, result in surface runoff from high level areas and hillsides down to valleys and dried floodplains, in which the surface water can infiltrate and add to the groundwater if there are permeable soil layers and a low groundwater table.

Under disturbed conditions, such as during pumping or under ground construction, a similar process appears in humid climate areas, so called *induced infiltration*. Salt water intrusion in drilled wells near the coast is a common type of induced infiltration. In connection with a tunnel construction project in the city of Malmö in the 1970's a large amount of salt water was induced from Öresund. During the construction of the Äspö Hard Rock Laboratory, close to the town of Oskarshamn, southern Sweden, inflow of recent seawater was noted (Laaksoharju et al, 1999). In hilly near-shore terrain with little or no soil layer, for example in large parts of Sweden's coastal areas, it is probable that some of the water infiltrating into the rock fractures flows through the fractures in the rock straight out into the bottom sediments. Very few measurements of such systems have been conducted in Sweden. However, in Norway and on Svalbard low salt levels have been detected locally on the bottom of some fjords. It is believed that this is a result of submarine outflow of groundwater (Haldorsen & Lauritzen, 1993).

The natural flow to and from lakes through flow in the bedrock is generally not very high except for areas with challenging topography. A big study of an area with crystalline bedrock in New Hampshire, USA, has shown that even though the amount of outcrops was frequent, groundwater that had flowed through the bedrock constituted only 4 % of the flow to a lake, while the rest went through the soil layers and surface water bodies. Only 1 % of the outflow from the lake flowed through the rock even though groundwater flow constituted 40 % of the runoff (Rosenberry & Winter, 1993).

During disturbed flow conditions, for example by pumping in drilled wells, the induced infiltration from lakes and streams can increase and water quality changes may appear in the well water. Archipelago regions with lakes and wetlands have a better chance of avoiding salt water intrusion than areas without such water bodies. These areas also stand a lesser risk of getting fossil seawater (from earlier seas) in wells since close water reservoirs counteract drastic lowering of the groundwater levels. A method to increase the availability of fresh groundwater in wells, for example in the archipelago, is therefore to use plastic membranes or injection in soil (a subsurface dam) to screen off the natural outflow of groundwater through the soil layers and thereby assure water-saturated soil layer conditions year around in the vicinity of drilled wells.

4.4 Storage and flow characteristics

4.4.1 The bedrock porosity

The hard rock has certain *porosity* and is in this text defined as the total volume (%) of cavities and holes in the rock. In granite the primary porosity can amount to a few percent, for example in the form of microscopic fluid-filled holes (so called fluid inclusions). Hydrogeologically, these make up a closed system. Calculations in conjunction with SKB's earlier research in Stripa have shown that the amount of salt-rich fluid inclusions is large enough to theoretically explain even markedly high salt levels in the deep groundwater (Nordstrom & Olsson, 1987). The secondary porosity in hard rock is dominated by cavities and holes formed through fracturing. Often the term *fracture porosity* is used. It refers to the total volume (%) of holes present in the rock fractures. Fracture porosity is dependent on fracture frequency and fracture orientations, fracture width, and mineral fillings. It varies therefore strongly depending on the bedrock geological history, its composition, texture and structure, and from this its capacity to stand rock stress changes. Often the total fracture porosity may amount to around one percent, but bedrock with a very low frequency of open fractures may also occur. However, the fracture porosity does not give any direct information of the prerequisites for water flow since fractures may sometimes be disconnected and therefore not contribute to the groundwater flow, Figure 5.

The *kinematic porosity* (flow porosity) includes the amount of cavities and holes that can actually contribute to the groundwater flow. It varies greatly but may for Swedish crystalline bedrock be approximated to 0,0001–0,1 %. In porous media it corresponds to the *effective porosity*, which is the volume (%) of cavities and holes available for drainage. Soils fall in the range 3–30 % or a hundred to a thousand times larger than for hard rock.

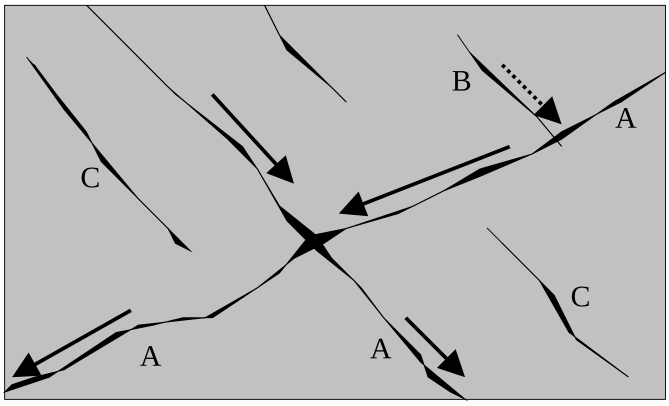

Figure 5. *Groundwater flow in rock fractures*
A=Drainable; flow
B=Drainable; No flow
C=Not drainable; No flow

4.4.2 Hydrogeologic significance of the type of rock

Test pumping and hydraulic tests in different rock types show some differences in median capacity between various rock types, even though the differences within one rock type can be even larger. The significance of the composition of the rock type is not completely investigated and in at least some rock type regions the differences are small between different rock types (Krásný, 1996a). The differences that do exist are thought to depend on the rock type's chemical composition and the structural and textural conditions. Rock types with a high quartz content (acid) such as granites and quartzites are often brittle and easily break when stress conditions change. Basic rock types such as gabbro and amphibolite are normally tough and can withstand fracturing. A large amount of capacity data from drilled wells in Sweden is stored at the Geological Survey of

Sweden (SGU) and has been compiled for the provincial hydrogeologic mapping project. A compilation based on nine counties in Sweden shows that the capacity is around 60 % higher for granitic rock types than for basic metamorphosed rock types, but the variation is large between the different counties, Figure 6.

The grain size for the rock type is also of great importance. Fine-grained rocks such as superficial volcanic rocks easily develop minor fractures, although not always connected. For the coarse-grained rock types, such as pegmatites the consolidation of grains can be weak and the grains can get loose during weathering and tectonic deformation.

The grain orientation and the mica content are of certain importance for the fracture formation and thereby indirectly also important for the capacity. In a massive rock type (one without any dominating grain orientation) fractures can form in many orientations depending on the tectonic processes during and after the rock's formation. Granite's distinctive feature is therefore the regular fracture pattern with two or three dominating fracture sets. In a capacity point of view, it is favourable if the pattern consists of two orthogonal and vertical fracture sets, cut off by near horizontal fractures, for example sheeting joints. These often appear in the youngest granite areas.

In rock with a particular mineral orientation and especially if the rock has important mica or graphite content, the tectonic deformation is taken up as movements along the mica planes. Mica rich gneisses therefore have usually fewer fractures than granite and these are also mostly oriented parallel to the gneiss orientation. The gneiss's mineral orientation (foliation) is therefore of big importance for the capacity. Steep foliation will usually give small discharge capacities, while more flat foliation is better suited for discharge of groundwater. Advanced gneissosity, where the material partly has melted (migmatite) lessen the importance of foliation.

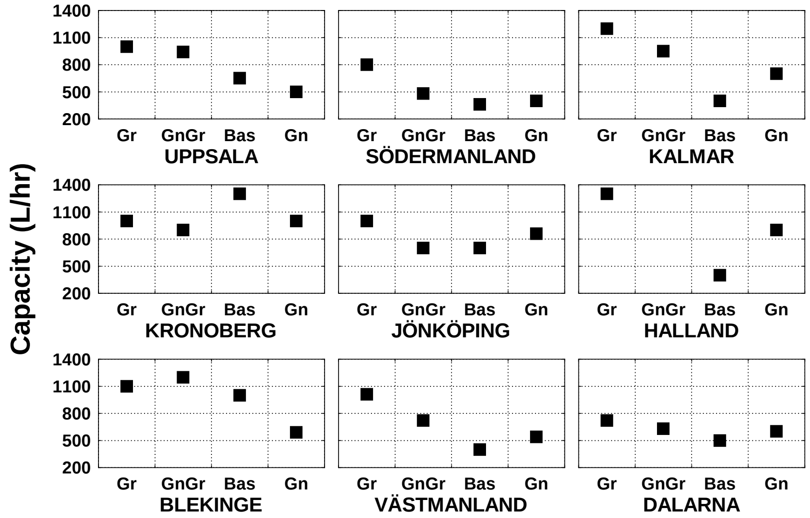

Figure 6. Median capacity for drilled wells in different rock types in nine counties in Sweden.
Gr=young granite, GnGr=gneiss-granite, Bas=greenstone, Gn=grey gneiss. (Based on data from SGU's hydrogeologic county maps)

Dyke intrusion of magmatic rock types which crosses the bedrock can be of great hydrogeologic importance. Diabase dykes can, for example, be hydraulically tight and prevent water flow and thereby build vertical dams for the groundwater. On the other hand, the contact zone to the surrounding rock, can be strongly fractured due to the contraction that occurred in the diabase during cooling. Weathering of some olivine-containing diabases or later deformation and foliation can give better flow conditions. Figure 7 shows an outline of *hydraulic conductivity* for a few rock types common in Sweden. The hydraulic conductivity, see definition in 4.13, is is in general higher in granites than in gneiss of sedimentary origin as well as in "greenstone", i.e. basic rocks as gabbro.

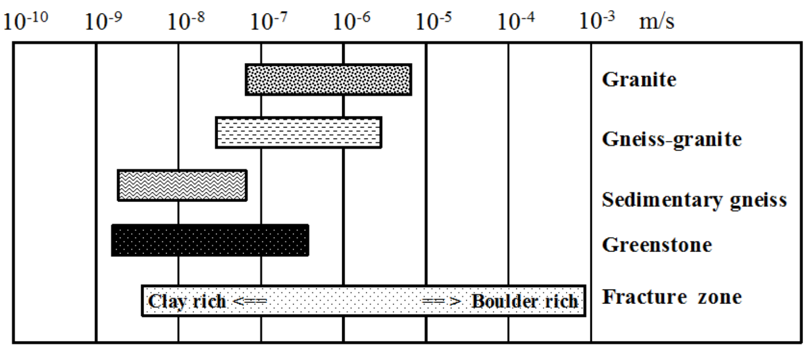

Figure 7. *Approximated regional hydraulic conductivity for some common rock types in Sweden. (Based mainly on compilation by Carlsson & Olsson, 1979 and SGU's hydrogeologic county maps)*

4.4.3 Significance of tectonics

The genetic meaning of fracture formation for groundwater flow at the local scale has during the last decade lost importance because later stress situations in the bedrock often regenerate older fractures, whereby their hydraulic conductivity changes locally. For regional hydrogeological studies, for example for finding locations for deep storage of nuclear waste, the genetic geological conditions are of some importance. See for example the SKB's provincial compilations (SKB, 1999, R99:17-35) and the hydrogeologic evaluation of larger structures in connection to the Hard Rock Laboratory at Äspö (SKB, 1996). A study on fractures in hard rock has previously been performed by KASAM (1998) and no further analysis will therefore be done here.

From a groundwater flow point of view, the most important fracture types are *extension fractures* (often formed from tension) and *shear fractures* (sometimes with displacements, i.e. faults). These fractures are often genetically coupled, Figure 8.

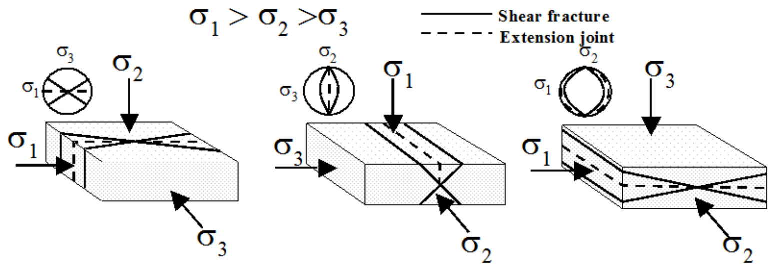

Figure 8. *From a flow point of view, the dominating fracture groups are related to the the main stress orientations (σ1) (after Park, 1989 and Lloyd, 1999)*

More important than the origin of the fractures are their length, orientation, complexity, mineral filling, and the roughness of the fracture surface. Even though extension fractures generally are partly open, they can be short, steeply dipping, and parallel ("en echelon") so that water flow between them gets more difficult. Shear fractures make up the most complex fracture group, from a flow point of view. This is because displacements along the fracture planes may have accomplished different degrees of crushing and mineral transformation.

Sometimes several parallel shear fractures occur with more or less crushed intervening rock. Such a *shear zone* can favour water flow, see Figure 7. However, heavy weathering or conversion to clay can in some cases totally block the groundwater flow, Figure 9. In some areas in Scandinavia as well as in dry climates, fractures and fracture zones are sometimes filled with clay minerals that swell when water enters (montmorillonite and bentonite). In spite of a tiny natural groundwater flow, these clays can be very problematic during construction in rock if the clay content is not observed, since it has sometimes led to collapse of the construction.

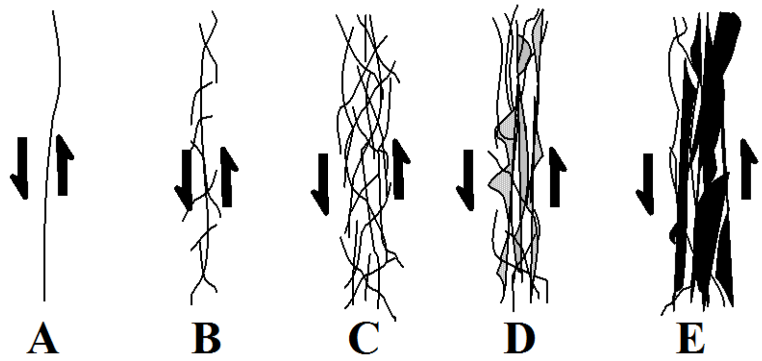

Figure 9. Shear fractures and their varying hydrogeological significance. A=Simple shear, water tight; B=Secondary fracturing, almost water tight; C=Complex shear, boulder formation, moderate permeability; D=Complex shear, crushing, high permeability; E=Complex shear, clay formation, low or moderate permeability (from Larsson (ed) 1984)

The fractures' orientation is also of importance. Steep structures gain vertical flow transport. Of greater importance for the total flow are flat shear fractures or shear zones, which collect and horizontally drain flows from steeply dipping structures. The intersection between flat shear fractures or flat and steeply dipping shear fractures can make effective drainage channels in the rock. Flat complex shear zones are found even at great depths (Talbot, 1990), although the frequency of flat fractures is highest down to a depth of about 50 m. A number of hydraulic studies of fracture zones (complex parts of fractured rock) have been done around the world, principally in connection with studies of storage alternatives for nuclear waste.

The hydrogeologic importance of fracture zones is not completely investigated, due to the complexity of these zones. A correlation between capacity data of wells and their vicinity to presumed fracture zones in hard rock has not been proven. This can be explained by that the wells generally are affected by the local fracture conditions at the well. High capacities in wells located in soil-covered pronounced valleys have been pointed out for a long time (Larsson (ed), 1984). Thick soil layers giving significant water storage capacities may in some cases explain this. In the Bohemian Massif in the Czech Republic, the transmissivity values i.e. the hydraulic conductivity multiplied by the thickness of the water bearing layer) are 2–4,5 times higher in valleys (representing discharge areas) than in the higher laying recharge areas (Krásný, 2000). Wells located in high permeability soil layers were taken away from the well population before the analysis so that the values only represent the hard rock. An alternative explanation may be that fine particles can clog the recharge areas , while the discharge areas may be flushed by the groundwater flow (Krásný, 2000). The fact that large fracture zones may give large flows has also been studied in connection with underground construction. There is a clear positive correlation between large fracture zones and the drainage of soils and rock (Olofsson, 1991, 2000), but there are also exceptions. Two big tracer tests performed in Germany have shown that the hydraulic conductivity in a fracture zone in quartzite and shale was tenfold higher than that of the surrounding bedrock (Maloszewski et al, 1999). In some cases (in Norway among others) well drilling in fracture zones has produced small water quantities which is explained by clay filled fractures which reduces the permeability (Banks et al, 1993). Fracture zones with high clay content and very low permeability have been the case in some underground construction sites such as the Bolmen Tunnel Project in southern Sweden (Olofsson, 1991).

4.4.4 Groundwater flow

The topography is of vital importance for the flow since gravitation is the predominant reason for groundwater flow. The flow goes from higher to lower places, see Figure 10, why the terrain can be divided into a complex pattern of recharge and discharge areas. The pattern changes because of seasonal variations of precipitation and evaporation. Terrain with high and steep slopes is theoretically promoting a deeper groundwater flow, although the flow in reality is also governed by fracture patterns, the characteristics of the fractures and also sometimes the interfaces of rock types, such as diabase dikes. In flat areas, the flow is therefore promoted in the shallow parts of the bedrock. In flat coastal areas, such as certain islands in the archipelago, and in connection to the Lake Mälaren, the groundwater level in relation to the sea or lake level decides the flow gradient. Below sea level in flat areas the flow in the bedrock, can be assumed to be very small and mainly related to a regional flow.

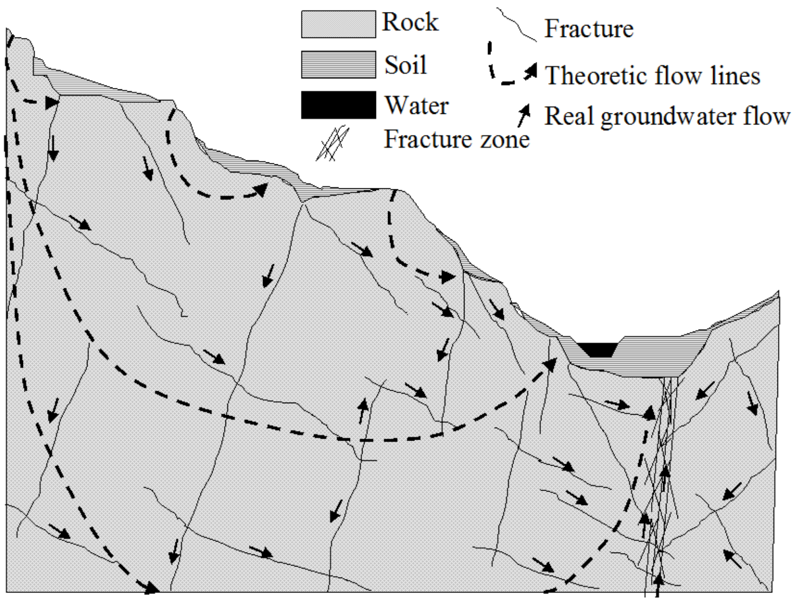

Figure 10. Principal flow pattern as a result of varying topography. In hard fractured rock the real flow pattern is far more complex than the theoretical

A large amount of measurements on hydraulic conditions in hard rock have been made, in connection to investigations for groundwater exploitation as well as before the location of a repository of nuclear waste. Hydraulic investigations in bore holes (such as injection tests) in sections screened off with packers, show that the hydraulic conductivity generally decreases with depth, which mainly is due to the rock stress increase, Figure 11.

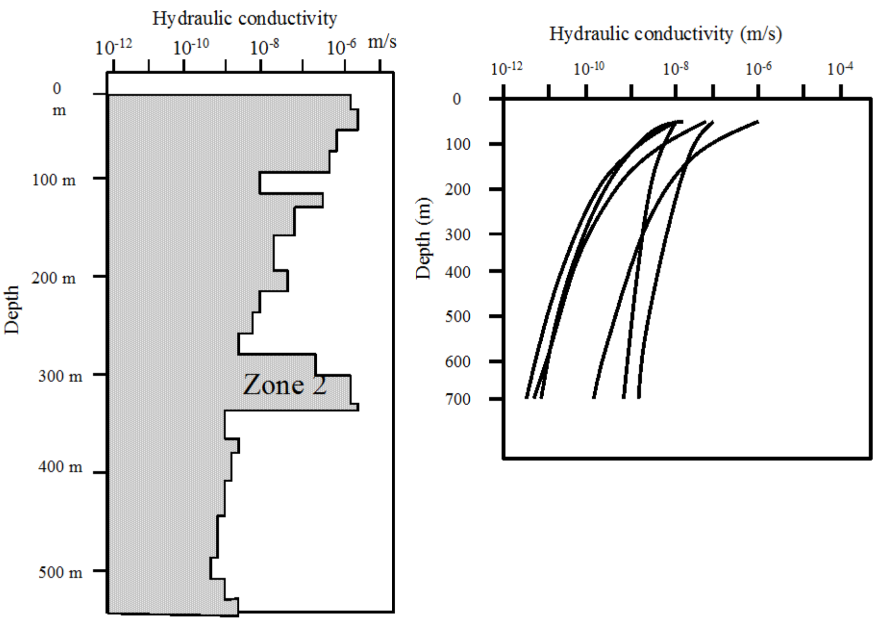

Figure 11. *Hydraulic conductivity from drill hole measurements (injection tests) in Finnsjön, Uppland, in relation to depth (25 m packer distance) (on the left) and calculated hydraulic conductivity for some rock types (on the right). Zone 2 is a flat fracture zone (after SKB 1992 and Knutsson, 1997)*

The fracture frequency (mainly flat fractures), decreases rapidly with depth in the shallow parts of the bedrock. The variability along the drill hole is however significant as a rule and highly permeable fractures and fracture zones can be found even at great depth. In the deep drill hole for investigation and possible gas occurrence in the meteorite crater structure north of Lake Siljan, fractures and fracture zones were found along the entire depth of the drill hole, 7 000 m (Juhlin et al, 1991). Flow measurements in drill holes conducted for instance in connection to the underground laboratory on Äspö show that the water flow is very reduced with greater depth (>300 m).

4.5 The importance of heterogeneity and anisotrophy

Hydraulic heterogeneity is a measure of the spatial variation of hydraulic conductivity in a material (Freeze & Cherry, 1979). The variation is a total statistical measure of all hydrogeologic factors influencing the flow conditions. During the last fifteen years, a big part of the quantitative research in groundwater flow has been aimed at getting a statistic description of the heterogeneity. In a homogeneous material the hydraulic conductivity is independent of the measuring position. Hydraulic tests in bore holes and wells have often shown that the hydraulic values have had a log-normal distribution (Gustafson (Krásný, 1993, Rhén et al., 1997). A homogeneous material can therefore be defined as a material where the variance of the hydraulic conductivity is under 0,5 log-cycles (Schultze-Makuch et al., 1999).

Hydraulic anisotrophy is a measure of the variation of flow possibilities in different directions. In fractured hard rock, where the fracture configurations govern the flow possibilities, the anisotrophy is generally high. Totally isotrophic conditions are rare in nature; even a sand or gravel deposit often shows some form of anisotrophy. Measurements on Äspö have shown that the hydraulically most conductive orientation has approximately 100 times higher conductivity than the hydraulically least conductive orientation (Rhén et al., 1997). Hard rock can therefore be called hydraulically heterogeneous as well as anisotrophic.

High hydraulic heterogeneity and anisotrophy generally lead to large difficulties to predict groundwater flow and in turn to considerable uncertainty regarding flow pattern and flow velocity. The hydraulic properties are highly heterogeneous even in a separate fracture. The most common reasons for heterogeneity in hard rock are variation in fracture frequency (the number of fractures per rock volume), fracture orientation, fracture length, degree of openness (for example, the average distance between the fracture walls), the roughness of the fractures (most fracture planes show irregularities), and mineral fill (many fractures have complete or partial mineral depositions or transported fine material that cover the fracture planes and reduces the conductivity) (Sharp, 1993). Because of the irregularity of the fracture plane, the flow in the fracture plane normally follows paths or channels. At sufficient depth, (500–1 000 m) most fractures have channel flow, due to the rock stress conditions (Tsang & Tsang, 1987). Strong shearing in the fracture or differing distribution of mineral deposits generally lead to an increased level of flow channelling (Hakami, 1995), Figure 12.

A number of tracer tests in separate fractures has been conducted in connection to research on nuclear waste, and they also show that channel flow is generally predominant (for instance Abelin et al, 1991). The first phase of comprehensive and detailed (block scale < 10 m) tracer test experiment on Äspö has recently been presented and shows a varying fracture width and evident flow connections along the fracture plane. Despite relatively hydraulic homogeneity along the fracture surface, the delay of the tracer substances was larger than in laboratory experiments, which is explained by an increased porosity in a transformation zone along the fracture surface (Winberg et al, 2000).

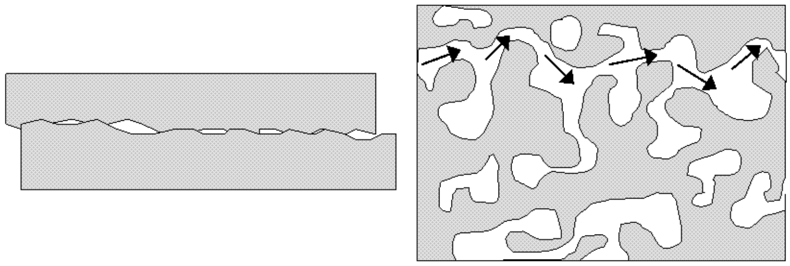

Figure 12. Principal sketch of channel flow. The cross-section shows how shearing promotes channelling

Substantial channel flow can also occur along cuts between fracture planes even if the pore volume in intersections is small compared to the total fracture porosity (Hakami & Stephansson, 1993). An important factor affecting the hydraulic heterogeneity is a more or less random variation of connections between the different flow paths. The hydraulic heterogeneity is therefore actually three-dimensional (Dijk & Berkowitz, 1999). With a flow pattern with several intersected fracture groups, and small channelling on the fracture surfaces, the total heterogeneity can be small, with strong shearing and formation of minerals it increases significantly. The geometry of the fracture surfaces is thus very important for the hydraulic heterogeneity (Kosakowski & Berkowitz, 1999). A description on different flow promoting structures in hard rock, mapped in connection to tunnelling, is shown in Figure 13.

The hydraulic heterogeneity is strongly scale dependent. Modelling groundwater flow in fractured hard rock in Nevada has shown that an increase of the kinematic porosity with a factor 1.5 gave an almost 6 times higher variation of the maximal flow (Pohll et al, 1999). During the last decade, a lot of research has been carried out in order to explain the scale dependency. A comprehensive compilation of literature on the scale dependency of hydraulic conductivity shows that the geometrical average of the conductivity clearly increases with scale (Vidstrand, 1999). Quantitative measures of hydraulic properties must normally be conducted in a detailed scale in drill holes. The question is how these measures, that often only represent a single fracture or even single channel on a fracture plane, really can represent the hydraulic conditions in a larger volume of rock. The variance is largest for measurements at a laboratory scale and decreases gradually in field and regional scale. (Margolin et al, 1998). When doing hydraulic tests between packers in drill holes the packer spacing is of great importance for the variation of the hydraulic properties.

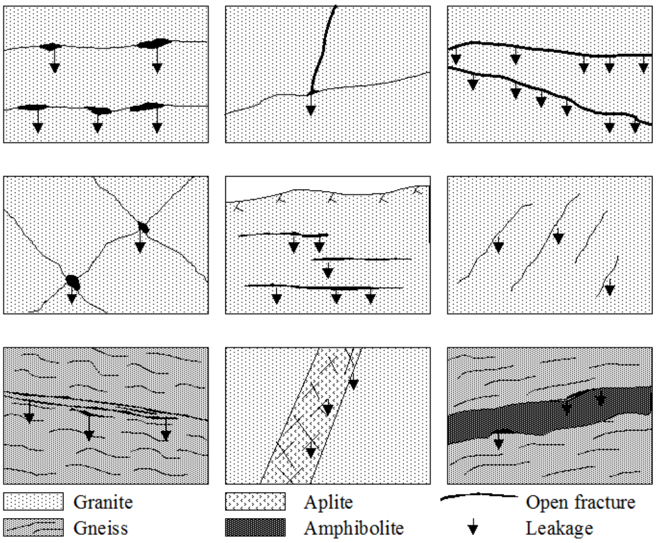

Figure 13. *Example of flow promoting structures in hard rock (Olofsson, 1993 after Carlsson and Olsson, 1997 and Palmqvist, 1990)*

With a packer spacing of only a few meters, the flow in every single fracture will be of importance. Hydraulic tests in drill holes on Äspö imply that more detailed measurements than over the block size 10 m, are hard to scale up to larger volumes (Vidstrand, 1999). The hydraulic conductivity is assumed to increase with the scale until it reaches a limit over which the conductivity is constant (Clauser, 1992, Hsieh, 1998). REV (Representative Element Volume) is a common measure of the volume over which the conductivity value does not change any more. Since the hydraulic conductivity has often shown to have a log-normal distribution, the REV is in practice very large, and it is doubtful if the concept is at all appropriate. At a regional scale and if larger inhomogeneities such as large fracture zones exist, these can possibly be treated separately with different hydraulic properties.

A really good measure of the heterogeneity in a fracture network does not yet exist, and especially not if it is meant to be applied on different scales. An attempt to describe the heterogeneity through so-called heterogeneity index, based on the variance of different fracture characteristics, has recently been presented (Cesano et al, 2000b).

4.6 Induced flow

Significant natural groundwater flow in hard rock mainly occurs in the shallow parts of the rock and in areas with large elevation differences, whereas it can be assumed to be small at great depth and in flat areas. Induced flow can either occur through hydraulic tests, such as test pumping in drill holes, or in connection to underground constructing. Almost every physical investigation of the groundwater conditions in rock will disturb the natural flow processes. A drill hole creates an important flow channel in itself which can lead to vertical flow paths (”hydraulic short-circuit”) that did not exist before. Underground constructions generally have great impact on the flow systems.

Studies of groundwater changes in soil and rock in underground constructions have substantially increased the knowledge on the complex flow conditions in rock, soil and in the border zone between soil and rock. (Olofsson, 1991, Cesano et al. 2000a). So far, groundwater modelling has not succeeded in describing these flow processes correctly, especially not concerning the impact of underground construction on groundwater in the soil layers (Olsson, 2000). Drainage of rock groundwater can in certain cases lead to an unsaturated zone between the groundwater level in the rock and the saturated soil layers above. This zone is often difficult to model. The groundwater flow is therefore strongly dependent on the spatially differing hydraulic conductivity of the soil layers in relation to the hydraulic properties of the flow governing structures in the rock. In that case, the hydraulic properties of the interface between these structures and the soil are of crucial importance. Very few studies have been made on this interface zone, partly because it is very difficult to study.

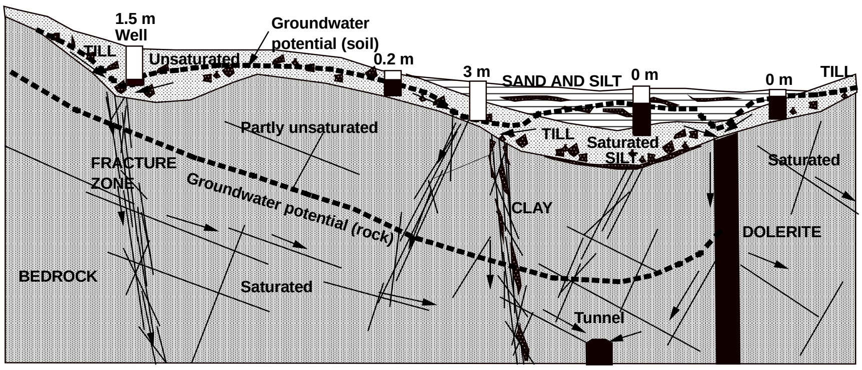

Figure 14. Principal sketch of drainage in the bedrock with a tunnel and how this can affect the groundwater conditions in soil locally. The conditions in reality are more complex locally (Olofsson, 2000)

Induced groundwater flow also leads to significant changes in the groundwater chemistry, shown in investigations on Äspö, where the inflow of saline water from the Baltic Sea and the mixing with water from old phases of the Baltic Sea were obvious (Laaksoharju et al., 1999).

4.7 The importance of rock stress changes

Changes in rock stress can be assumed to have a certain importance for groundwater flow in hard rock. Local as well as regional changes in rock stresses can lead to a reactivation of certain fracture orientations, which in turn can lead to a change in the flow possibilities of the separate fracture planes. The current rock stress condition affects the degree of openness and thereby also the hydraulic conductivity, whose variation in size as well as orientation has turned out to be equal to the local rock stress distribution (Carlsson & Olsson, 1979). Field tests with induced pressure in separate fractures have shown that the permeability increases with decreasing effective rock stresses (Alm, 1999). Rock stresses, vertical as well as horizontal, generally increase with depth, Figure 15.

The importance of rock stresses for flow can be assumed to decrease with the roughness of the fracture surface since the irregularities create a complex flow pattern. When constructing underground, the stress field will change locally around the construction; how large volume that is affected depends on the size of the construction. This also occurs at a detailed scale such as a drill hole, but the impact on the flow is then generally very small. Around the tunnels, the real effect will generally be negligible beyond a distance of approximately 1 tunnel diameter (Pusch et al. 1991). With careful blasting, like on Äspö, the impact can be much less, often less than 1 m, and with full boring technique the impact is only some decimetre.

However, the importance of rock stress changes for groundwater flow at a regional scale is unclear. Compilations of well capacities in drilled wells in hard rock (gneisses, granites and amphibolites) in southern Norway show a strong positive correlation between the capacities and the calculated land up-lift (Rohr-Torp, 2000). This is assumed to be a result of the young tectonic movements, caused by the land up-lift, which change the rock stress situation and reactivate existing fractures. In a limited study of well capacities and their relation to the current regional Scandinavian rock stress field, done in a test area east of Gothenburg, no connection could be detected (Wladis, 1995)

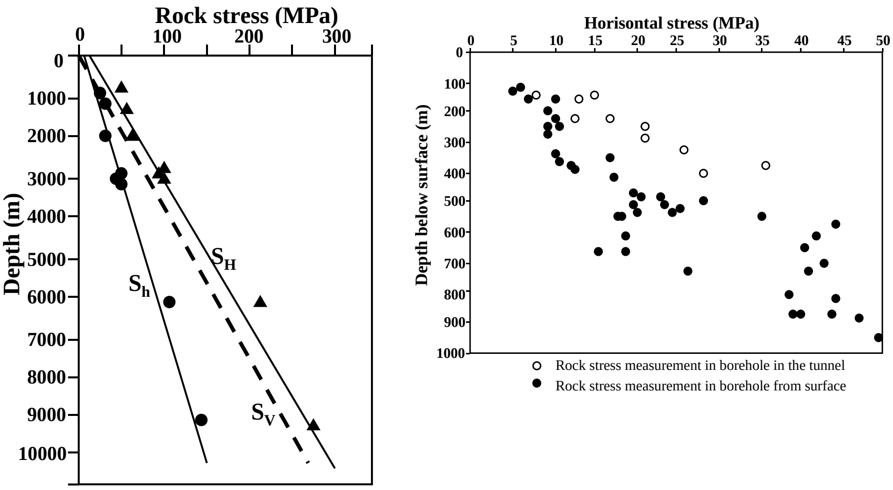

Figure 15. *Rock stress profile based on rock stress measurements within KTB (the continental deep drilling project in Germany). The rock stress measurements were done mainly in gneiss and amphibolite down to 9 km. The vertical stress (S_v) is calculated from density. S_H and S_h show the largest and the smallest horizontal rock stress respectively. The largest horizontal rock stresses in drill holes on Äspö are shown as a comparison (after Te Kamp et al., 1995: Stille and Olsson, 1996)*

In connection to the ice melting, quick changes in the vertical load in combination with a regional compression horizontally might have caused so-called *neotectonic* movements, situated mainly in the northernmost Scandinavia, for example on the Pärvie fault west of Kiruna. Whether such new or regenerated zones of disturbance increased the flow possibilities for groundwater is not clear.

The predominant maximal horizontal rock stresses in western Europe, and in Scandinavia, have an orientation in NW-SE, which is explained by tectonic stress from the Mid Atlantic Spreading Zone (Amadei & Stephansson, 1997). This orientation can however digress and varies vertically as well as horizontally, because of land up-lift and topography, especially in connection to larger shear zones.

4.8 Chemistry of groundwater in hard rock

4.8.1 Introduction

The soil through which the water percolates before reaching the rock gives its imprint on the chemistry of the water. The most important factors are the thickness of the soil layer, the texture and the mineralogical composition of the soil. The loose overburden in Sweden was mainly formed during the last ice age. Nevertheless, traces of many geological periods remain in the fracture systems in the rock; one example is at least three generations of calcite deposits. When percolating, the precipitation is mixed with older groundwater, for example groundwater from an old marine period. Finally, at great depth, very saline groundwater with an uncertain origin can be found. Most likely, the chemical composition of this water is a result of rock forming processes at depth. Many chemical reactions in the groundwater, even at great depth, take place due to microbial activity. Consequently, the chemistry of hard rock groundwater is not easily described.

4.8.2 Factors influencing groundwater chemistry

The following processes affect the groundwater chemistry, to different degree.

- Deposition of salts from the atmosphere;
- Weathering;
- Nutrient uptake by plants;
- Formation of secondary minerals;
- Oxidation and reduction of organic substances and mineral components;
- Ion exchange on fracture minerals;
- Mixture with older groundwater.

The deposition from the atmosphere consists mainly of sea salts but during the twentieth century, anthropogenic pollutants, especially sulphur from fossil fuels, has become a predominant part of the deposition of substances from the atmosphere. In Sweden, where a majority of the anthropogenic pollutants originates from Central and Western Europe, sulphuric acid, i.e. hydrogen ions and sulphate ions, has dominated the precipitation and resulted in a pH-value of around 4 in southern Sweden. Whereas the fall-out of sulphuric acid has decreased during the last decades, the fall-out of nitrogen as nitrate and ammonium is still high.

The weathering is a complex process, especially concerning silicate minerals. Most natural groundwater has hydrogen carbonate as negative ion. This indicates that carbonic acid is the weathering agent. There is however a number of complex processes behind this. Is has recently been discovered that mycorrhiza has a special ability to decompose silicates (Wallander & Wickman, 1999). Mycorrhiza is a sort of fungi that lives in symbiosis with trees, shrubs and herbs and helps them with the mobilisation of nutrients from the soil. These fungi secrete organic acids, for example oxalic acid and citric acid. The acids are later decomposed by bacteria and in this process carbonic acid is produced. The weathering is to a large extent ruled by the mineral composition of the soil. However, certain trees, such as the spruce, extract more base cations (calcium, magnesium and potassium) than other trees such as the pine (Pozwa et al. 2001). Thus, the vegetation also influences the weathering rate.

Nitrogen is the limiting nutrient in most eco-systems on Earth. This means that the nitrogen is normally recycled between vegetation and soil in a closed cycle. During the twentieth century, when the amount of nitrogen rich fertilisers increased, this pattern of recycling has changed. Today, 25 kg/ha/year leak from agricultural land, which is equivalent to approximately 10 mg of nitrate-nitrogen per litre.

Through weathering the minerals are dissolved, which may result in a deposition of new minerals when the concentration of the components reaches the solubility in the groundwater. It is mostly calcite, but also mica-like minerals such as illite and chlorite that have been formed in this manner in fracture systems in the rock. Minerals with a uranium content and other radioactive minerals are most frequent in acid granites. These minerals have been deposited in fracture systems in the rock in particular and they produce radon through radioactive decay. The amount of radon in groundwater is determined by the frequency of fractures and by the width of the fractures (Andrews et al, 1989).

Ion exchange takes place when there is a disturbance of the groundwater chemistry. In the Swedish bedrock this is mostly accentuated in connection to changes of the sea level. During the last 10 000 years, there have been two relatively saline seas covering large parts of southern and middle Sweden, the Yoldia sea and the Litorina sea. As saltwater has penetrated the rock, the favourable places for adsorption of positive ions have been saturated with sodium. As the sea has withdrawn, calcium in the groundwater is substituted for sodium and the result is a water containing sodium bicarbonate, like the mineral water from

Ramlösa in Scania. In drilled wells, especially close to the coast, this kind of groundwater is not rare.

In soil water and groundwater there are small amounts of organic material. This can be energy for bacteria that live in the ground and in the fracture systems of the rock. The oxidation can occur with the help of oxygen, just as when human beings assimilate energy from the food using oxygen. Oxygen has however a limited solubility in water, approximately 10 mg/l at +8°C, and this is easily consumed in a soil profile. Some bacteria can use other substances for their oxidation, such as nitrate, manganese oxides, iron hydroxides and sulphate. From these processes nitrogen gas, dissolved manganese and iron ions, and hydrogen sulphide are formed. Anaerobic conditions are very common in groundwater in bedrock. A good illustration of this is the fact that high iron contents is the most common quality problem in drilled wells and that groundwater with a smell of hydrogen sulphide is not unusual. These reactions are called reduction-oxidation reactions because they include the reduction of one specie and the oxidation of another. These so-called redox reactions involve the transfer of one or more electrons, for example oxidation of organic material with ferric hydroxide:

$$CH_2O + 4\ Fe\ (OH)_3 + 8\ H+ \ \text{->}\ CO_2 + 11\ H_2O + 4\ Fe^{2+}$$

Redox-reactions are vital for deposition of radioactive waste since radionuclides are present in forms, which are less soluble in reducing conditions than when oxygen is present.

A mixture with older saline groundwater is common in the bedrock near the coasts (see Fig. 16). The Litorina sea was more saline than the Baltic Sea, close to Stockholm, is today. As a consequence, it is not unusual to find groundwater in the bedrock more saline than the seawater in the Baltic. Furthermore, water has been found at depth, approximately below 1 000 m, that is more saline than the sea water but whose origin is uncertain. Unlike the sea water it is a Ca-Cl-water and not a Na-Cl-water. Except in Sweden, this kind of water has been found in an old copper mine in Outokumpu, Finland, and in the Canadian shield.

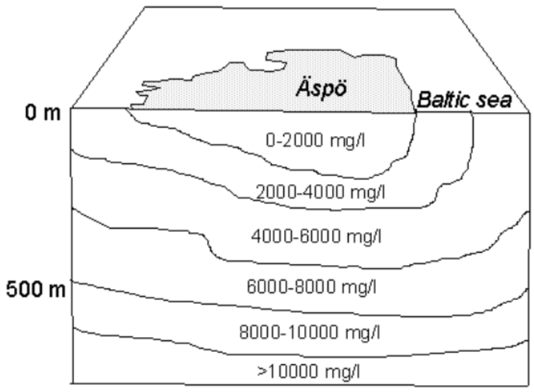

Figure 16. *Fresh water pillow under the island Äspö before the construction of an underground laboratory caused a cone of depression. In the mainland, groundwater with a chloride content higher than in the oceans has been found at a depth of more than 1 000 m (after Laaksoharju et.al. 1999)*

4.8.3 The stability of the groundwater chemistry

In connection to deposition of waste in the bedrock, the stability of groundwater is of great importance. Two conditions are especially important, the pH of water and the redox conditions. The stability is described with the concept buffering. There are different buffers. pH, i.e. the concentration of protons (H+) is buffered by the carbonic acid system in most natural waters. Carbonic acid is a weak acid, which means that only a part of its protons is set free from the anion, the negative ion. The carbonic acid is an acid with two protons and can release two protons when pH rises in the water:

$H_2O + CO_2 \rightarrow H_2CO_3$
$H_2CO_3 \leftrightarrow H^+ + HCO_3^-$
$HCO_3^- \leftrightarrow H^+ + CO_3^{2-}$

Weathering consumes protons and produces hydrogen carbonate. If further hydrogen ions are added, for example from acid rain, the hydrogen carbonate will react with these and the change in pH will be small.

$H^+ + HCO_3^- \rightarrow H_2CO_3 \rightarrow H_2O + CO_2$

Groundwater in the bedrock has a high content of hydrogen carbonate and is thus well buffered.

Redox buffering is a similar process but for electrons and not protons, as earlier described. A weak redox buffering can for example result in one redox system changing into another. If low contents of oxygen are present in the water, then the redox condition easily changes into oxidation of organic material with ferric hydroxide, which can give a high content of ferrous iron in the water. Groundwater in bedrock at larger depth usually has this redox level and is rather stable, since there are large quantities of ferric oxides in the bedrock minerals. If the redox conditions would change into sulphate reduction, then hydrogen sulphide will form. This is harmful to the copper capsules that are planned to enclose the radioactive material.

4.8.4 The groundwater chemistry and the turnover of groundwater

As earlier mentioned, the renewal of water varies with depth below the ground. This has a great impact on the groundwater chemistry since the chemical reactions, including silicate minerals, are very slow. The acidic deposition has affected surface waters, supplied by shallow flow through the ground. Superficial groundwater, pumped from dug wells a few metres deep, has in some cases been acidified. Groundwater in drilled wells has only occasionally been acidified, for example on the West Coast of Sweden, where the soil layers are thin.

Many attempts have been done to date the deep groundwater. If the age is in excess of a few hundred years, ^{14}C is a possible method. ^{14}C has a half-life of 5 700 years. Such measurements have shown that water at 5–600 m depth can be several thousands of years old. However the method has been questioned since carbon exists in for example very old carbonates and this carbonate may be remobilised. Organic material, such as humus, in the water has also been used to date the water with ^{14}C. The result has often been a lower age, but of the same order of magnitude. A humus and carbonate dating at Fjällveden in Sörmland of groundwater at 400 m depth gave a result of 4 200 years from the carbonate and 1300 years from the humus. ^{36}Cl has been used to date very old water. ^{36}Cl is formed in the atmosphere, like ^{14}C , but has a half-life of 300 000 years. There are however uncertainties with this isotope as well.

A fact recently found is that a mixing of different groundwater in the bedrock is common. Groundwater has been found containing tritium (^{3}H), with a half-life of 12,4 years, but with a very high ^{14}C -age. This implies a mixture of very old water with very young. This relationship is used in a new concept of modelling which describes the mixture chemically using reactions for dissolution and formation of minerals. It has in an excellent way described the mixture of rainwater, so-called biogenic water (water that has passed the soil zone with its weathering reactions), brackish water from the Baltic Sea, ice-age water and the very saline water described above. (Laaksoharju et al., 1999).

4.8.5 Microbial activity in groundwater

As earlier mentioned, organisms play an important role in groundwater chemistry. Mycorrhizal fungi mediate weathering to variable extent. Most redox reactions take place through the activity of bacteria. Many inorganic reactions are very slow. Due to this they do not react even though the energy levels are sufficient. Bacteria reduces this slowness dramatically with enzymes. What is possible thermodynamically is realised by the bacteria while purely inorganic reactions often are inhibited.

Bacteria exist on all levels in the rock but two sorts can be distinguished: those living on organic material produced by photosynthesis at the surface, and those living at great depth using energy from hydrogen gas which diffuses from great depth (Pedersen, 2000) (see Figure 17).

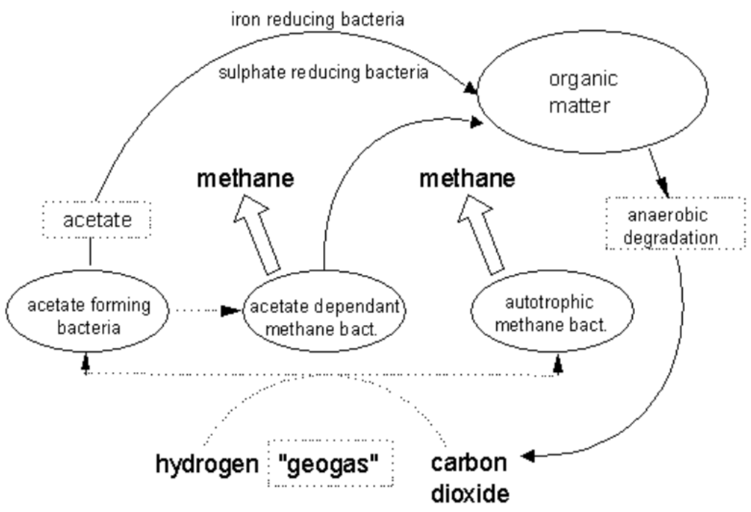

Figure 17. Bacterial flora in the bedrock. The bacteria get their energy from hydrogen gas and have carbon dioxide as a source of carbon. (after Pedersen, 2000)

4.8.6 New knowledge

The presence of microbial life at great depth in the rock has only recently been discovered. The understanding of how different groups of bacteria live on each other's metabolic products is also completely new. Very saline water at great depth is also a recent discovery in the Scandinavian shield, whereas this has been known for a long time in Canada. The development of modelling the mixture of different water has contributed with a new possibility to estimate the turnover of groundwater at great depth. This also gives an opportunity to explain the puzzling findings of groundwater with both ^{14}C ages of several thousands of years and tritium, indicating a very young water. The microbial life at great depth in rock that incorporates carbonates and carbon dioxide into organic material make dating with ^{14}C uncertain.

4.9 Investigation methods

Investigation methods and analysis methods for studies of groundwater in rock have developed much during the last two decades. A lot of new knowledge and new methods have been developed in connection to studies for deposition of nuclear waste. This section gives a brief description of different investigation methods for rock and groundwater conditions.

The methods can be divided into groups depending on the *scale* applicability of the method and what kind of data that is generated, either *qualitative data*, for example interpreted air photos, or *quantitative data*, for example direct measures on water content. Another distinction is between methods giving *direct* information on processes of groundwater flow and methods where the measured results need advanced analysis in order to be used, thereby only providing *indirect* knowledge, Table 1. Of course it is sometimes difficult to differ between these groups.

Table 1. Different types of investigation methods for groundwater in hard rock (modified after Cesano, 1999)

Mainly indirect information	Mainly direct information
Studies of earlier investigations in the area	Hydraulic tests
Remote sensing and air photos	Hydrometeorologic measurements (run-off, precipitation, temperature)
Bedrock and tectonic mapping in the field	Flow measurements in bore-holes etc.
Airborne and ground geophysical measurements	
Geochemical sampling of soil-, rock and groundwater	
Rock stress measurements	

Direct quantitative measurements have been the most common source of input data to mathematical modelling, especially results from hydraulic tests. The indirect qualitative measurements are however very important for correct boundary conditions.

Table 2 shows a summary of different methods of investigation and their suitability at different scales. The table is mainly based on experience from groundwater prospecting as well as from larger construction projects. (Larsson (ed), 1984, Andersson et al., 1993, Rhén et al., 1997, Stanfors et al. 1997, Cesano, 1999, Lloyd, 1999). Investigations for depositories of nuclear waste are especially complicated since the repository most likely will be situated at great depth (400–600 m). Most measurements at surface, i.e. all measurements except borehole measurements, will thus always be performed at least 500 m from the intended deposition site which in turn leads to increased uncertainty in the measurements. Furthermore, the airborne and ground geophysical measurements will give values integrated over large volumes. The accuracy for these will decrease rapidly with depth and at great depth only the geological and structural main features will be identified. These features are often of little interest for an estimation of the local flow.

Table 2. *Some investigation methods and their relevance for studies of the occurrence or flow of groundwater at different scales. The scales refer to the length of sides of squares or to a cube with the same side length (for drill hole investigations).*
D=direct information on groundwater occurrence or groundwater flow. I=interpretation in order to get relevant information. Increasing number of signs indicates increased relevance.

Method	Can show for example	100 000 m	1 000 m	100 m	10 m	1 m
Map studies	Topography, fracture zones, geology	II	II			
Remote sensing, satellite images	Fracture zones, soil moisture, discharge and recharge areas	III	II	I		
Remote sensing, air photos	Fracture zones, discharge and recharge areas	III	III	II		
Mapping of rock types and fractures in the field	Rock types, fractures (frequency, fracture type, orientation, mineral filling)		I	II	III	I
Airborne geophysics	Structures, geology	III	II	I		
Ground geo-physics	Structures, geology, groundwater level		I	II	III	I
Borehole geo-physics	Fractures, geology, water content			I	II	III
Inventories of wells	Capacities, groundwater levels			D	DD	
Flow measure-ments in bore-holes	Groundwater flow				D	DDD
Injection test in borehole	Hydraulic parameters				DD	DD

Method	Can show for example	100 000 m	1 000 m	100 m	10 m	1 m
Interference test in borehole	Hydraulic parameters			D	DD	
Bore hole TV	Fractures, rock types				I	III
Rock stress measurements	Current rock stresses			I	II	I
Analysis of rock cores	Fractures, mineral filling, rock types				I	II
Regional chemical water sampling in wells	Regional water chemistry			I	II	I
Water sampling in bore holes, disturbed conditions	Change in water chemistry over time, genesis of the water and chemical reactions			II	I	
Stable isotopes	Genesis of the ground-water		I	II	I	
Radioactive isotopes	Age and mixture of the groundwater		I	II		
Tracer tests in borehole	Flow paths, flows			D	DD	D

As is evident from the table, there hardly exist any methods of measurement that give direct quantitative information on groundwater occurrence or groundwater flow at a more general scale. The hydraulic tests, often performed in groundwater investigations, seldom give a sufficient general information to determine boundary conditions for mathematical modelling. For this reason, a combination with a more comprehensive indirect method is often needed in order to establish a conceptual geologic section.

It is also important to understand what is being measured. A groundwater level can easily be measured in a drill hole but in hard fractured rock no real groundwater table exists. Two adjacent drill holes will display different water levels simply because they are furnished with water from different fractures, with different water pressure. In order to get a representative average level, different amount of drill holes are needed, depending on the hydraulic connection between the fractures, which in turn depends on the fracture configuration, degree of openness, mineral filling etc.

The measurements have a spatial dimension, difficult to represent with a correct hydrogeological model, and a time dimension, i.e. when the measurements were made is important information since flow is a dynamic process. Groundwater occurrence and groundwater flow changes constantly due to seasonal variation in groundwater recharge and groundwater temperature (mainly dependent on changes in the viscosity of water). Because the volumes of water in hard rock are very small, small changes in air pressure and gravitation (for example the gravity of the moon) gives measurable differences in water level, for example in drill holes.

Generally, a groundwater investigation produces a lot of different data with different levels of detail. It is therefore important to have functional systems of the storage, integration and treatment of the data in order to achieve the specific goal of the investigation best. Specific and complex databases have generally been created in connection to the nuclear waste repository programs in the world. In the case of groundwater exploitation it is common that measurement data are compiled in a map form for example in a GIS (Geographic Information System), where all position-associated information can be included. (Gustafsson, 1993, Lloyd, 1999) At the same time, a GIS make treatment of different data possible and nowadays it can also be integrated with or connected to mathematical models (for input and output of data). A strong limitation with the most common GIS today is that most of them only handle two dimensions, i.e. they miss the vertical component. There are certain three-dimensional GIS but 3D standard programs for data storage and visual presentation are mostly used instead (so-called CAD-programs) in studies of deposition of nuclear waste. The fourth dimension, time, is much more difficult to present visually together with the three spatial dimensions, as this generates enormous amounts of data.

4.10 Groundwater modelling

4.10.1 Introduction

The word model is used in many different ways in hydrogeology (analogy model, conceptual model, laboratory model, mathematical model, etc.). It is therefore important to state, in each context, what is referred to, which question to be addressed, and what type of model that is to be considered. A common feature for all models is that they give a more of less simplified picture of an often very complex reality such as a groundwater system. Studies and calculations of hydrological and hydrochemical processes in models must therefore be interpreted carefully. In what follows only conceptual and mathematical models are treated.

4.10.2 Conceptual model

A conceptual model is a qualified estimation, in the form of a generalised description or a figure, of how the entire water system (Figure 18) or groundwater system (Figure 14) functions. It is based on existing data and assumptions on for example the type of system, the size and boundaries, the recharge and flow of water, and the chemical composition of the water. Furthermore, sources of pollution and risks of pollution can be mentioned.

A conceptual model is thus a sort of system analysis of the groundwater system and its relation to the surface water within the catchment area. It is appropriate to establish a conceptual model as an introduction to a groundwater investigation. A geological model is a valuable basis for such a model. It can be a tectonic model of the hard bedrock showing the open and prevailing fracture zones, linked to a soil type geologic model, showing the soil type distribution and storage possibilities.

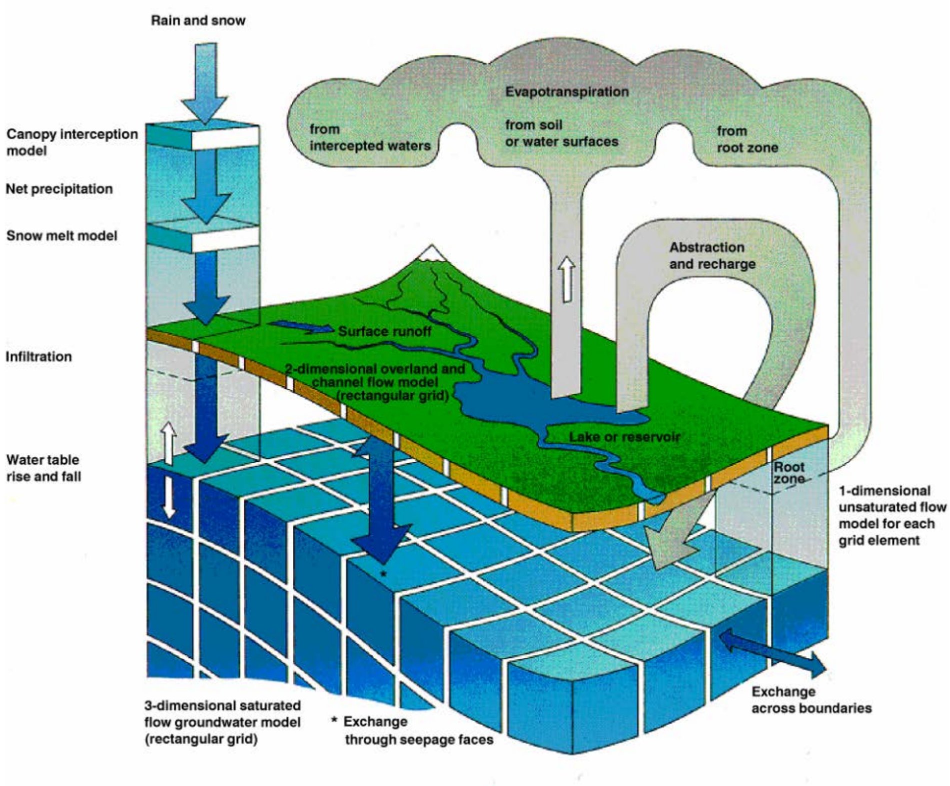

Figure 18. *Conceptual model over the entire hydrological system and information on how it can be modelled mathematically (after the so-called MIKE-SHE model).*

To be able to continue the work with other types of models, for example mathematical models, it is useful to start with a conceptual model, which is then further simplified.

4.10.3 Mathematical models

A mathematical model expresses a conceptual model in mathematical form, for example with a number of differential equations for flow and transport. The equations, usually partial differential equations, are derived from the basic equations of mass, energy and momentum together with equations describing the material and transport properties.

Different coefficients are introduced such as values for storage capacity (S) and hydraulic conductivity (K). Assumed values of these parameters are often used in the beginning. The conditions must be simplified since the field conditions are often too complex. This is especially true if the mathematical model is to be solved analytically, which may be possible when the aquifer is assumed to be homogenous and isotropic and the boundary conditions are simple. If more realistic conditions are to be treated, such as a heterogeneous, anisotropic aquifer, it is more suitable to solve the mathematical model numerically with the help of a computer. This has created the expression computer model. With such a numerical solution the differential equations are transformed into algebraic equations and the calculations are limited to a number of points or subareas.

The development of computer models has been fast, and the usage has increased rapidly during the last decades. This is partly because powerful personal computers have been common, but also due to the development of more and more advanced software. A common limitation is the lack of sufficient input data. The big advantage of a computer model is that once the model and its computer program is developed, a large number of calculations of different examples can be made rapidly. Thereby two purposes with the modelling work can be fulfilled, such as:

- To use the model in order to understand how the system works and to evaluate the importance of different input parameters by sensitivity analysis (see below). This can give guidance to further investigations, which then can be focused on the most important parameters and directed towards the areas where the information on these parameters is most needed (for example optimal sites for observation boreholes).

- To use simulations and predictions in a finishing stage where the models are well calibrated with for example observed data. This gives the possibility to study a number of possible scenarios such as different abstraction rates of water or, different loads of pollutants.

An important stage in modelling is the so-called successive sensitivity analysis. Normally this is performed by successive alteration of each parameter in order to see which parameters that are the most important. With an advanced variant of sensitivity analysis the model equations are reformulated so that the sensitivity coefficients can be calculated, when the values for the resulting parameters have been obtained. As an example of an advanced variant could be mentioned the so-called adjoint sensitivity method.

The most developed type of a computer model is the physically based one. Such a model is based on a number of equations and assumptions on how the hydrologic system functions. The equations are basically valid for small scaled, homogenous systems, but yet they are often assumed to be applicable to large scale heterogeneous systems. The scale dependency of different parameters, such as hydraulic conductivity, is currently subject to research. Physically based models are most appropriate for for small scale process studies, where the physical parameters can be measured and controlled, and do not have too high a variability. Consequently, a considerable amount of field studies is often necessary to get input data for the model.

The most common numerical models within hydrogeology are the so-called finite difference models and finite element models. There are many variants both concerning the mathematical formulation as well as the numerical methods used to solve the different algebraic equation systems. These equation systems arise as a consequence of the assembly of the algebraic equations resulting from the spatial and time integration of the flow and transport equations. Both methods lead to a large system of nodes that are distributed in a grid over the considered area.

The boundary element model can be mentioned as an example of a special variant of the finite element method. It differs from the other methods in that the calculation area is only discretisised along the boundaries of the study area. In this way, the number of elements or nodes is reduced a lot. A disadvantage of the boundary element method is that the material properties in the considered flow domain must be homogeneous.

In finite difference models, the nodes are placed within a rectangular grid, or at the intersections of the grid. There is however an interesting variant of finite difference models usually called the integrated finite difference model. With this method the equations are approximated at the interfaces between the squares. In this way the grid cells need not be rectangular. The grid can then be better adapted to the material and geometrical properties of the aquifer (see for example Patankar (1980)). In finite difference models, the properties of the aquifer are assumed to be constant in each cell.

In finite element models, the area of investigation is divided in a number of elements. In two-dimensional models these are triangles or rectangles, and in three-dimensional models they are shaped as prisms, tetrahedrons or hexahedrons. Different element types can occur in the same area, provided that the basis functions are continuous along the element sides. The basis functions, or he interpolution functions, are used to define an approximate solution to the piezometric head solution. The flow can be described for each element by the groundwater levels in each node. The condition that the groundwater level (or the piezometric head) must be continuos in each node gives a system of equations.

Finite difference models were used first. They are relatively easy to understand and to program, especially in connection with the programming of parallel computers, often used nowadays to solve large problems. Finite difference models are suitable for solving regional flow problems in relatively uniform aquifers, but also in multiple layer systems. They are however limited because they are based on regular cells and because heterogeneous conditions must be described with the shape of these cells.

Finite element models are now the most commonly used to deal with hard rock flow problems. They are more difficult to understand and to program, but they are on the other hand more flexible since the elements can have irregular shapes. Therefore, they are more suitable for studies of more heterogeneous groundwater systems on a medium scale, such as the impact on the groundwater levels at a planned abstraction of water or on flow paths from a planned landfill.

An alternative to finite difference as well as finite element models is the so-called analytical element method, developed by Strack (1989). It uses analytical functions as solutions of the differential equations for the groundwater flow together with the associated boundary conditions. Each function represents a certain element in a groundwater system (such as esker material, clay and rock) and different boundary conditions (such as rivers, lakes and fracture zones). The model is calibrated with measured groundwater levels and known fluxes. The model has been used in Sweden to calculate the effect of groundwater abstraction from wells and regional flow patterns. An advantage with the analytical elementary model is that the calculation times, even for large problems, are usually much shorter than that with the numerical models. For this reason, the analytic element model is a very interesting alternative to numerical models for ordinary porous media for example in order to do a preliminary calculation of the groundwater flow in an unconsolidated deposit.

4.10.4 Models for fractured rock

The Swedish crystalline hard rock is a very complicated geologic environment for calculations of flow and transport. A number of factors has to be taken into account such as the irregularity of the fractures, unclear connections, varying surface structures, size, different levels of mineral filling and the different geochemistry and variations in rock stress. Traditional continuum models for porous media are therefore not suitable except for very large fractured areas. The following ways to model flow and transport in fractured rock have been tested:

- In a double porosity system of fractures and blocks. Such a system is dealt with as two overlapping continua. A special type is the model for so-called "double-porosity flow pipes".
- In networks of discrete fractures or channels (Figure 19).
- In a so-called stochastic continuum, where for example the K-value is considered to be a spatially irregular variable. (Figure 19).

The spatial variability in for example the hydraulic parameters is often dealt with by means of geostatistical methods, such as kriging. Moreover, the model calculations are realised by means of so-called conditioned simulation. This means that a number of input data fields (realisations) are created with the aid of Monte-Carlo simulation in such a way that the observed or assumed statistical properties are maintained. The result of the model calculations is a statistical comparison of flow and transport and flow patterns for different cases with the same likelihood. An uncertainty analysis of both input data and results is a stage in the model work. This type of model is an example of a stochastic model. They have been developed to give the probability distribution of values of the resulting parameters as a consequence of the uncertainty in the input parameters in a model.

The following models have been applied in studies of groundwater in hard rock, mostly in association to the work with SKB.

HYDRASTAR is a stochastic continuum model developed for SKB. The model is based on finite difference approximation and solves a groundwater equation.

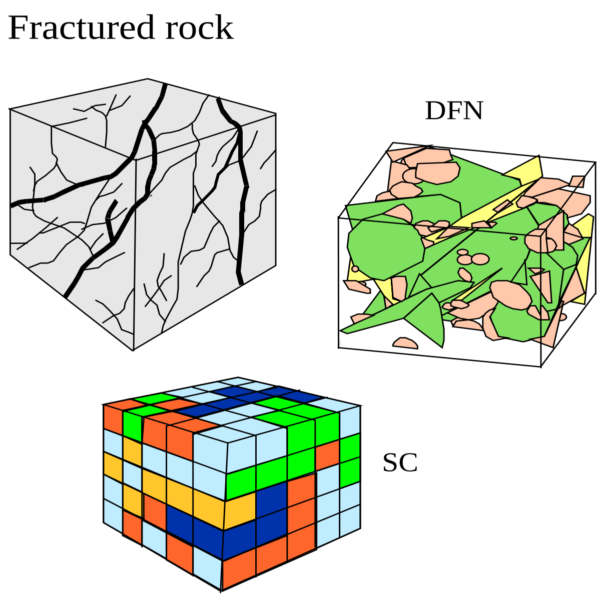

Figure 19. The two most common model concepts at numerical modelling of groundwater flow and transport through a body of rock with fractures; discrete fracture network modelling (DFN) and stochastic continuum modelling (SC) (from Follin, 2000).

PHOENICS is a general model for flow and transport problems. The model has been applied in large scale modelling for example in connection to the Äspö laboratory (Svensson, 1999). This model is an example of a model of the type integrated finite difference methods. Larger fracture zones can be explicitly ("directly") modelled with this model. It has also been modified to implicitly account for fracture patterns.

FRACMAN/MAFIC is a model developed by Golder Associates (Dersowitz et al., 1995). It is a so-called discrete network model. The fracture patterns are described directly in the model. MA-FIC treats interaction with the mass of rock (the matrix).

The models mentioned above are all based on different assumptions and consequently they have different areas of application. It is therefore not possible to directly compare results from different methods, instead they answer different questions. The following advantages and disadvantages are worth mentioning.

Continuum models are in principle only applicable to ordinary porous media. The suitability of continuum models as a tool to describe fractured rock is thus dependent on if, and to what extent, the rock can be regarded as a porous medium. For large-scale calculations mainly of groundwater flow, it may be possible to treat the rock body as a porous medium, but this is more doubtful for transport calculations of solutes in the groundwater. The description of porous media is traditionally based on the so-called REV-principle (se section 4.5). Studies with a stochastic continuum model (see Holmén (1998)) indicate that the REV for groundwater flow in fractured rock should be of about 1000 m. Thus, a REV size that is hardly applicable to transport calculation.

The stochastic continuum models can be regarded as a type of model for treatment of a heterogeneous rock body on a field scale somewhere between continuum models and discrete network models. A very important aspect of stochastic continuum models is that the problem with uncertainty in the calculations is accounted for.

The HYDRASTAR model is, as earlier mentioned a so-called continuum model. This means that apart from typical statistical parameters such as the mean and variance, the spatial correlation must be defined., This is done with the help of a so-called variogram. The HYDRASTAR model is designed for the handling of detailed flow patterns. The model can also be used for so-called inverse modelling. This means that the parameter values, such as the K-values, are calculated with help of a number of observations of the groundwater level. The model can be applied to both steady and nonsteady state flow problems. It is worth pointing out that time dependent inverse problems require enormous amounts of data. HYDRASTAR is probably one of few existing models that has been applicable to stochastic continuum inverse modelling of a rock body in three dimensions.

Recent studies have revealed the importance of accounting for the geologic structure and fracture patterns in the rock mass. As a consequence, the discrete fracture network model FRACMAN has been developed mainly in order to take the geometrical properties of the fracture patterns into account. Thus, this model can make more realistic descriptions of the flow and transport in a fractured rock mass. This requires knowledge of structural geology and good input data. Another problem is that the calculation time grows rapidly with increasing rock volumes, implying that the sizes of problems possible to treat will be limited by the amount of data that needs to be stored and the calculation time.

A general conclusion is that both analysis and modelling of groundwater in fractured rock is dependent on the access to a number of different model types. Thus, the different models complement each other, have different areas of application and often treat different physical and chemical problems connected to the actual groundwater flow as well as to the transport of solutes in the groundwater. It should therefore be observed that the above mentioned models are chosen from a large number of existing models. The chosen models only represent different type models.

Among the rest of the models, sometimes used for description of flow and transport in fractured rock on two or three dimensions, MODFLOW and MIKE-SHE could be mentioned.

Both were used in the investigations of the Hallandsås tunnel. MODFLOW is part of the model package Groundwater Modelling System (GMS), which also contains a number of models for geostatistics, visualisation and GIS-communication. MODFLOW is probably one of the most commonly used groundwater models in the world. MODFLOW solves a general groundwater flow equation in three dimensions. The model has become very popular because of the "modular" structure of the computer code. There are a number of variants of the model, and it contains routines for evapotranspiration, open and closed drainage. As earlier mentioned, it can also interact with transport models of different kinds, such as models for solute transport and LNAPL transport.

The MIKE-SHE model is a relatively complete hydrological model system (see Figure 18). Apart from a description of the groundwater, it also contains a description of the water in the unsaturated zone as well as the surface water, i.e. watercourses of different kinds. The area of application for the model should mainly be modelling of groundwater problems related to the catchment area. The equations are formulated in a way that implies that the model is hardly more applicable to fractured rock than any other porous medium model, although certain types of fractured media could be accounted for (see the MIKE SHE manual).

4.10.5 Model verification

An essential prerequisite of a mathematical model is that the model is functioning and that it gives reasonable results. In order to ensure this, the following stages should be an integral part in the modelling work.

Calibration	To show that the model can reproduce measurements (such as time series) after a certain adaptation of the parameters.
Verification	To show that the calibrated model can reproduce other results (for example a different part of a time series) without adaptation of the parameters. Or to show that a computer model in some sense is a true representation of a conceptual model with defined boundaries or areas of application and corresponding accuracy.
Validation	To show that a computer model has a sufficient level of accuracy within the area of application, i.e. that the model results fulfil the aim of the model. However, in a strict sense a model can never be completely validated, only validated to a certain degree.
Sensitivity analysis	To show how the main parameters, such as hydraulic conductivity, boundary conditions, etc., will affect the results.
Uncertainty analysis	To calculate the uncertainty of the result, with statistics or percentage. Or to describe what impact model simplification, scale problems and measuring accuracy has on the uncertainty.

4.10.6 Models used as decision system support when managing groundwater problems

One of the most important purposes with groundwater modelling is to use the results as basic data for different kinds of decisions. The issue can be to evaluate different alternative locations of a repository, a road or a waste plant or to optimise the management of a plant, where there are different equipment alternatives, etc. The decision-maker usually has a multitude of different factors or criteria to consider. The results from the groundwater model are thus not sufficient as basic data, but has to be assessed together with other factors or results from other models. Economical values, as well as different kinds of environmental and cultural values can be of great importance. Even sheer subjective opinions can be an important factor. It does not take a large number of factors to make a problem very complicated and difficult to grasp. Within hydrology there is a large number of systems, all developed for different applications. Hannerz and Lindström (1999) identified around 90 different systems in their literature survey. One of the first methodologies for handling groundwater problems was proposed by Freeze et al. (1990, 1992).

Decision support systems can be designed in many different ways. The two central parts in a decision-making system are some type of database, containing the information, and a model that extracts and processes the data from the database. The data are then treated and formed into a decision aid, that facilitates the evaluation and comparison of advantages and disadvantages of different alternatives. Scientific models, i.e. models of the type described above, are used to calculate or predict consequences of scenarios or input data.

Systems especially interesting for SKB in connection to the choice of a site for a depository, is probably systems that can handle results from models as well as sheer subjective judgements. Traditional cost-benefit-analysis has been used by environmental economists, but is now regarded as an insufficient tool for the formation of bases for decisions. Instead, a number of different variants of Multiple Criteria Decision Aid models (MCDA) have been developed. NAIADE can be mentioned as an example (Munda, 1995), developed at the Joint Research Centre, Ispra, Italy. The MCDA is illustrated in Figure 20, how it is used in the decisionmaking process handling groundwater-related problems.

The MCDA–method has recently been applied to a groundwater problem when choosing a route for an European highway from four alternatives, close to a municipal water supply plant in a large esker in southeastern Sweden (Eliasson, 2001).

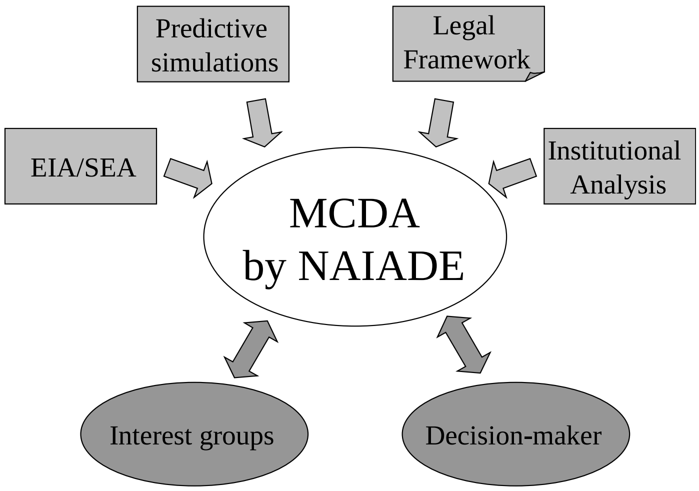

Figure 20. Exemplifies how MCDA, with the use of NAIADE, can be used as a tool in water management (Linde et al., 2000).

4.11 Need for complementary knowledge

During the last decade a large amount of knowledge on groundwater in hard rock has emerged. Generally, this knowledge has been obtained in connection to studies of deposition of nuclear waste in different parts of the world, and there has thus existed a defined goal. However, these studies provide an important general knowledge on hydrogeology, which can be used in other areas of interest such as for example the prospecting of groundwater resources. The focus has been on a quantitative, usually statistic, description of hydraulic relations, and the calculations have often been based on hydraulic borehole tests in certain points. A large amount of measuring equipment has been developed and tested for different scales and purposes. The following problems have been treated intensively, yet without finding a satisfactory solution:

- *The scale problem.* To what extent can detailed measurements and hydraulic tests on a small scale describe groundwater occurrence and flow in large test volumes and on a regional scale?
- *Measurement problem.* Closely connected to the scale problem. How is it possible to let hydraulic measurements represent larger test areas (for example 100m/side)? How can qualitative analysis like bedrock analysis and tectonic analysis, be designed to enable quantitative measurements?
- *The problem of heterogeneity and anisotropy.* How to best describe the hydraulic heterogeneity and anisotropy?
- *Modelling problem.* Which mathematical models can best describe the groundwater flow in a fractured body of hard rock, on different scales and which input data and measurements are necessary for this?

Other important problems have not been treated as thoroughly and significant scientific work is needed.

- *Groundwater recharge in hard rock.* Very little research has so far been done on this subject, which is central for both groundwater exploitation and construction in hard rock. This field of science also includes natural and induced groundwater flow from soil to rock. Basic research on the contact zone between soil and rock is a prerequisite for a successful combination of soil and rock water modelling.
- *The hydrogeology of the superficial rock.* A substantial part of the flow in hard rock takes place in the superficial part (-50 m). The research has nevertheless been almost completely concentrated on deeper parts. The Äspö Foundation has, however, invested in some projects where the hydrology and hydrochemistry of the superficial groundwater in rock is studied. The importance and occurrence of superficial fractured rock and the importance of superficial flat tectonic structures for the groundwater flow is still not investigated.
- *Flow through unsaturated rock.* The subject basically treats the superficial rock, is also relevant to flow in rock that has been aired, for example when the rock is refilled with water after drainage. It is still unclear how the water percolates through unsaturated fractures or if the flow requires local saturation.
- *The kinematic porosity of the rock.* There is a lack of general values on the kinematic porosity of the rock under different hydrogeological conditions, related to quantitative measurements (such as geological and geostructural). This type of measurements is important for the estimation of water resources in areas with a lack of fresh water, for example the Stockholm archipelago, and for the groundwater flow to tunnels and rock cavverns.
- *The occurrence of short-circuit-flows* in rock is still unknown, especially concerning groundwater chemistry. Extensive scientific work is needed. This is linked to the problem of heterogeneity.
- *An active transport* of gas has been noticed in ore prospecting projects in northern Sweden. This phenomenon is not yet evaluated in connection to the repository issue.

- *Tools for better decisions.* The models used in the work with SKB are meant to form a scientific basis for different decision making. Judging from what has been presented so far in reports from SKB, a deficiency is that no *existing tools for decision making* have been further developed or applied. The principle *of a decisionmaking support system* is to provide a decisionmaker, faced with a complex and unstructured problem, with a basis for his decision, such as a location of a repository. The idea is to quickly compile and handle large amounts of information with the help of the computer. This will give a better basis, hopefully at a lower cost. It is important to stress the fact that a decision support system is only meant to be a support for the decisionmaker, and not a decisionmaking system. For simpler, less complex problems, so called expert systems can be applied. It should be pointed out that the responsibility always lies with the decision-maker, no matter if he uses a decision-making support system.

4.12 Literature

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4.13 Definitions of important terminology

Anisotrophy	The variation of physical properties in different directions- opposite to isotrophy.
Aquifer	A body of geologic material which can yield water in significant quantities.
Buffering	Resistance to chemical change, in pH, i.e. the amount of free hydrogen ions, as well as the oxidation condition of the ions (redox).
Computer model	Numerical description and calculation of a mathematical model with the aid of a program-language such as Pascal.
Conceptual model	General description and/or a figure in principle of how for example an aquifer functions.
Continuum model	Model based on the mechanics of deformable media.

Discharge area	Area where the groundwater flows out from the system, often in lower parts of the terrain, in springs, rivers, lakes and seas.
Discrete fracture network model	Model with separate description of fractures concerning their geometrical and physical properites.
Discretisation	Transfer of a continuum model to a numerical model for instance by dividing a flow area into a grid.
Evapotranspiration	The amount of water that evaporates from soil and water surfaces and disappear from vegetation through transpiration.
Groundwater	The water that fills pores, fractures and cavities in soil and rock and which hydrostatic pressure is higher than or equal to the atmospheric pressure.
Groundwater recharge	Flow of water directed downwards and that reaches the groundwater table in the system. It can be direct through the infiltration of precipitation or indirect through inflow from rivers and lakes.
Hard rock	Rock that is so hard and dense that the main part of the groundwater exists and flows in secondary structures, mainly fractures. Crystalline rock types, such as granite and gneiss, is predominate but certain hard and dense sedimentary

and volcanic rock are also included in the concept hard rock from a hydrogeological point of view.

Hydraulic conductivity The groundwater flow per time unit through a unit area perpendicular to direction of the groundwater flow, when the hydraulic gradient is equal to 1 and the properties of the water is taken into consideration (density, viscosity). The hydraulic conductivity is K and it is measured in m/s. Compare with permeability.

Hydraulic gradient The difference in water levels per length unit in the direction of the groundwater flow.

Hydraulic heterogeneity Spatial variation of hydraulic conductivity in a material.

Induced recharge Inflow of surface water to the groundwater due to lowering of groundwater levels, which in turn is caused by pumping or drainage for instance connected to a tunnel.

Interception Part of the precipitation that does not reach the ground but get stuck in the vegetation and then evaporates.

Mathematical model Model that expresses a conceptual model in a mathematical form with the help of a number of differential equations for flow and transport.

Permeability	The material specific properties of soil or rock to transmit water under pressure per time unit (without regard to the properties of the water). Is signified with k and measured in m^2. Compare with hydraulic conductivity.
Pietzometric pressure level (or p. head)	Level to which the groundwater rises when for example an impermeable layer, under which the groundwater system is isolated from the atmosphere, is penetrated.
Porosity	Pores, fractures or cavities in soil and rock, filled with gas, air and/or water. Total porosity is the total volume of all types of cavities. Effective porosity or kinematic porosity is the volume of all the cavities where the water can flow freely. Fracture porosity or secondary porosity, is the volume of the cavities that exists in the fractures.
Recharge area	Area where groundwater is recharged through direct infiltration of precipitation, most often in higher parts of the terrain.
Redox process	Transfer of electrons from one chemical specie to another. This often leads to a change in the solubility of one of the two species/ions, for example Fe^{3+} is difficult to solve and forms hydroxides while Fe^{2+} is rather soluble.

Rock stress	The stress transferred through contact between grains in a rock-type or between blocks of rock is called the effective stress. The stress calculated on the component perpendicular to the surface towards which the force is acting is called normal-stress.
Stochastic continuum	A domain whose physical or geometrical properties are treated as randomly distributed – for example hydraulic conductivity in a rock mass.
Stochastic model	Model containing at least one random component- the same data can thus generate different results.
Storage coefficient	Coefficient that give the groundwater reservoirs capacity to yield or store groundwater, expressed in water volume per unit area, when the groundwater level changes one unit. It is represented by S and is dimensionless.
Tectonics	The structural construction, deformation and development through history of the bedrock. The most common tectonic processes are folding, faulting and jointing.

5 The function of bentonite as a barrier in the deep repository for spent nuclear fuel[1]

5.1 Introduction

The Swedish nuclear power programme is expected to generate some 13 000 m^3 of spent nuclear fuel. Levels of radioactivity in excess of the natural background will persist in the waste material for 100 000 years. Thus the method chosen for isolating the spent nuclear fuel must guarantee that radioactive, high-level waste (HLW) will not be released to the environment over this time scale (KASAM, 1998a).

International consensus suggests that disposal, in a deep repository in stable bedrock, is the only viable alternative for nuclear waste disposal presently available (KASAM, 1998c). On this basis, the Swedish Nuclear Fuel and Waste Management Company (SKB) has devised the so-called KBS-3 concept, founded on a series of natural and engineered barriers to inhibit the spread of radionuclides from the repository. The components of this proposed containment strategy are illustrated in Figure 1, each playing a part in preventing breaches in the systems' integrity (KASAM, 1998b). They consist of the fuel contained in zircalloy tubes, a steel insert in a copper canister, bentonite buffer and the surrounding bedrock.

The spent fuel consists of extremely insoluble phases, mainly uranium dioxide (UO_2), enclosed in chemically resistant

[1] This chapter was prepared at the Division of Inorganic Chemistry, Luleå University of Technology, by Dr. Douglas C. Baxter and Prof. Willis Forsling (member of KASAM).

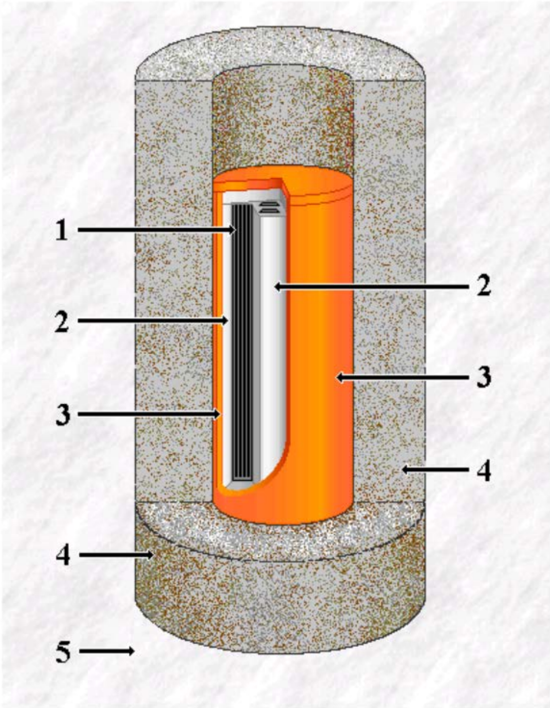

Figure 1. Schematic diagram (not to scale) showing the barrier system in the KBS-3 concept. (1) Uranium dioxide nuclear fuel pellets enclosed in zircalloy tubes. Neither the spent fuel pellets nor the zircalloy casing is easily dissolved, which retards the leaching of radionuclides in the event of contact with water. (2) Steel insert to provide radiation shielding and mechanical stability. Delays the penetration of water should the copper canister be breached. (3) Copper canister, extremely resistant to corrosion under the chemical conditions prevailing in the deep repository. (4) Bentonite buffer which limits the rate of groundwater transport to the canister. The bentonite will also hinder radioactive material from being dispersed in the environment. (5) Bedrock. Should radionuclides escape from the buffer, transport in groundwater through fissures in the bedrock would be limited by precipitation reactions and sorption on the surfaces of rock and mineral particles. Thus the bedrock will prevent radionuclides from reaching the biosphere.

zircalloy tubes. As such, these factors alone constitute considerable barriers to the release of radioactive material to infiltrating groundwater (Quiñones *et al.,* 1998; Curti & Hummel, 1999). The fuel elements will be supported in a cylindrical, steel insert to provide mechanical stability and radiation shielding. Once sealed, a copper canister with a welded base plate and lid encloses the steel insert. The copper canister will be resistant to corrosion under the chemical conditions prevailing in the repository (see section on ”Pore water chemistry”, below). Each canister is placed in a deposition hole, surrounded by bentonite and thus isolated from direct contact with the bedrock.

The bedrock itself, hosting the deep repository, will also provide a natural barrier to the dispersion of radioactive material escaping from the bentonite. Radionuclides entering the groundwater present in fissures in the bedrock may be immobilised by precipitation as insoluble minerals or by sorption onto the surfaces of microfractures. Sorption processes cause material initially present in solution to adhere to the surfaces of solid particles, and are described in more detail in the section on ”Surface processes”.

The low porosity of bedrock also limits diffusion. In fluids, i.e., gases and liquids, material moves from zones of higher concentration to more dilute regions. In this way, the material concentration will gradually become constant throughout the fluid and so-called concentration gradients will disappear. The combination of sorption processes and slow diffusion will retard migration of radioactive material to the biosphere (Samper *et al.,* 1998; KASAM, 1999).

The bentonite buffer around the HLW canister has a key role for repository safety. It must hold the canister in place in the centre of the bore hole. The buffer will also have to conduct the fuels' residual power, i.e., the energy emitted when radioactive material decays and which is converted to heat in the canister and its' contents. The bentonite will hinder groundwater, possibly containing various corrosive substances, from flowing freely to the canisters' surface. A crucial prerequisite for the

KBS-3 concept is that the repository will avoid oxygen encroachment during its' operational lifetime. In an oxidising environment, corrosion processes would be accelerated and the solubilities of fuel components would be enhanced. Bentonite thus has an important role to play in preventing corrosive oxidising agents from reaching the canister. Should the copper be penetrated, corrosion of the steel insert will require a supply of water through the breach. One of the purposes of the bentonite packing material is, in such a situation, to limit the rate of water transport to the canister, again to prevent corrosion. The buffer will also prevent leakage of radioactive gases and water soluble compounds to the surrounding bedrock.

Bentonite is a naturally occurring clay found in deposits in many countries world-wide. Clay is the collective name for various earthen materials having particle diameters less than 0.002 mm, that is to say < 2 µm. (Note that bentonite is actually a mixture of several different kinds of clay and various other compounds, as shown by the list of structural components in Table 1.) When such small particles are compressed, the gaps between particles, called pores, are also minute. Channels between particles are extremely narrow and tortuous. Therefore, well-packed clay is practically impermeable for water, *i.e.,* water finds it extremely difficult to flow through bentonite.

Water molecules, as well as ions (charged species formed when salts dissolve) and molecules dissolved in water, have a certain mobility, called Brownian motion, allowing them to flow into any available spaces. This motion is disordered, without any defined direction in the absence of an external force. If a pressure difference exists between the groundwater in the bedrock and the pore water in bentonite, then water molecules will be forced to flow in the direction of the pressure drop. Resistance to this flow is great, however, due to the extremely narrow pores in bentonite, and so the rate of water transport (hydraulic conductivity) is low.

Table 1. *Typical composition of sodium bentonite with data taken from Dultz & Bors (2000), Eriksen et al. (1999), Hemingway & Sposito (1989), Ochs et al. (1998) and Wanner et al. (1996)*

Component	Chemical formula	Content (mass %)
*Clay minerals**		
• montmorillonite**	$Na_{0.6}[Si_8]Al_{3.2}Fe_{0.2}Mg_{0.6}O_{20}(OH)_4$	65-80
• illite**	$Na_{0.75}[Si_{6.8}Al_{1.2}]Al_3Fe_{0.25}Mg_{0.75}O_{20}(OH)_4$	1.4
• kaolinite	$[Si_4]Al_4O_{10}(OH)_8.n\ H_2O$ (n = 0 or 4)	< 1
Accessory minerals		
• quartz	SiO_2	10-15
• feldspar***	$(Na,Ca)(Al,Si)Si_2O_8$	5-8
• calcite	$CaCO_3$	0.7-1.4
• gypsum	$CaSO_4{\cdot}2\ H_2O$	0.34
• pyrite	FeS_2	0.3
Miscellaneous		
• organic matter		0.05-0.5
• sodium chloride	NaCl	< 0.01
• unspecified		< 1

* Elements given in square brackets reside in the tetrahedral layers (see Figure 2).
** Idealised compositions pertaining to sodium bentonite are given.
*** Chemical formula shown is for the plagioclase mineral series often found in bentonite.

Other than low hydraulic conductivity, a property that is common to all clays, the bentonite particles also have the capability to bind water molecules, as well as positively charged ions, cations, to their surfaces. (Negatively charged ions are called anions.) This is a result of the structure of montmorillonite, the major clay component of bentonite. Montmorillonite consists of three molecular layers (see Figure 2), forming flat, sheet-like crystals. The distribution of electric charges in the crystals is such that primarily cations, but even electrically neutral water molecules, are attracted. Water molecules (H_2O) have an angular structure ($_H/^O\backslash_H$), the oxygen atom (O) acting like a negatively charged electrical pole and the hydrogens (H) behaving as positive poles, forming a so-called electric dipole.

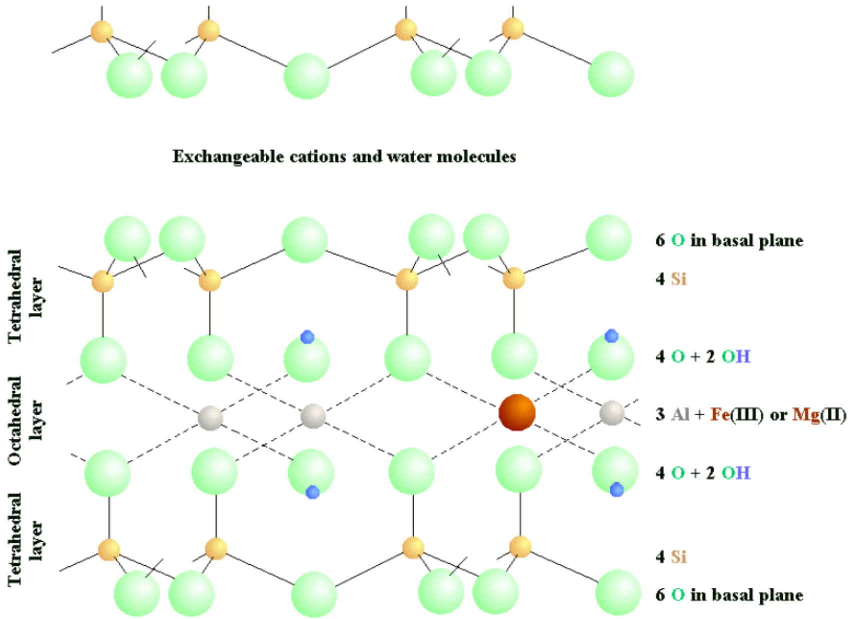

Figure 2. The structure of montmorillonite, the major clay mineral present in bentonite. A portion of the lower tetrahedral layer of an adjacent mineral particle is shown in the upper part of the figure. Water molecules and cations reside between the particles, the cations being strongly sorbed on the surface due to the negative charge arising from partial substitution of divalent magnesium for trivalent aluminium in the octahedral layer. Adapted from Andrews et al. (1996) and Hemingway & Sposito (1989).

When a compressed block of moist bentonite is saturated with water, layers of water molecules will grow around each and every crystal, causing them to swell and pushing the particles away from each other. If the bentonite adsorbs water in a confined enclosure, in the present context in a repository bore hole, then swelling is limited to the space available around the buffer and the internal pressure in the block will rise. The bentonite will then reach its' maximal capacity as a barrier against the transport of gases and dissolved compounds.

This chapter deals with the aforementioned properties of bentonite and the state of the art concerning specific aspects of how bentonite interacts with groundwater, gases and compounds dissolved in the pore water. In general, the data discussed in this report has been taken from the international scientific literature. Attempts are also made to identify areas of uncertainty regarding the behaviour of bentonite both in the initial stages of deployment and from a long-term perspective.

Selection of bentonite for use in the Swedish deep repository is in accordance with decisions made in most countries currently contemplating HLW disposal. As the repository is planned to be located in hard bedrock below the groundwater level, the bentonite buffer must limit the water flux and restrict the release of radionuclides into the host rock in the event of canister failure. The required properties of the buffer can therefore be summarised as (Cho *et al.*, 2000):

- low gas permeability, *i.e.*, the ability to prevent gases formed by chemical and nuclear reactions from escaping to the bedrock and ultimately dispersing in the environment;
- low hydraulic conductivity, *i.e.*, the ability to limit the flow rates of groundwater, potentially containing corrosive substances, to the canister surface and of pore water, perhaps contaminated with radionuclides from a leaking canister, to the bedrock;

- high radionuclide retardation capacity, *i.e.,* the ability to remove dissolved radioactive material from pore water by sorption on the surface of the bentonite particles; and
- high swelling potential, *i.e.,* the ability to adsorb large quantities of water such that the bentonite expands and seals any cracks in the buffer to hinder material transport to and from the canister.

5.2 Characteristics of bentonite

For the Swedish deep repository, a commercially available form of bentonite, Volclay MX 80, mined in Wyoming and South Dakota, USA, has been proposed as the buffer material. This particular bentonite consists to 65–80 % of the smectite mineral montmorillonite, with particle sizes less than 2 μm. Montmorillonite is an aluminosilicate clay mineral with a layered structure, as illustrated in Figure 2 (Andrews *et al.,* 1996; Hemingway & Sposito, 1989).

The central layer has an octahedral geometry, and consists of aluminium (Al), oxygen (O) and hydroxyl (OH) groups. Two tetrahedral sheets sandwich this layer, and are composed of silicon (Si) and oxygen, some of the latter being shared with the central layer in which some of the Al is replaced by trivalent iron, Fe(III), or divalent magnesium, Mg(II) (i.e., the 4 O between the 4 Si and the 3 Al + Fe(III) or Mg(II) rows in Figure 2). The term divalent means that, e.g., a magnesium atom looses two of its' electrons when it forms a chemical compound with another element, e.g., oxygen. These electrons are taken up by the other element, thus magnesium becomes the positively charged cation Mg^{2+}, bonded to the negatively charged anion O^2-, in the chemical compound magnesium oxide, MgO.

Generally, cations are adsorbed onto clays to balance the negative charge on the aluminosilicate structure caused by substitution of silicon or aluminium by lower valent cations in the tetrahedral or octahedral sheets, respectively (*e.g.,* trivalent aluminium for tetravalent silicon in the tetrahedral sheet). Broken bonds on the edges of crystallites may also cause development of a small charge.

In montmorillonite, some of the trivalent aluminium atoms have been substituted by divalent magnesium. In this way, a net negative layer charge is developed predominantly in the octahedral sheet. This is balanced, in MX 80 bentonite, by incorporation of monovalent sodium ions (Na^+) in, or on cavities in the basal plane of oxygens (refer to Figure 2) in the tetrahedral layers. Significant amounts of water are present between the sheets and, therefore, the incorporated cations do not actually come into direct contact with the basal oxygen framework until the interlayer is dehydrated.

Additional major components typically found in bentonite are described in Table 1. Although a minor component, calcite is important for the determining the nature of the chemical environment within the buffer. Calcite is a basic mineral, capable of neutralising strong acids and thus of maintaining alkaline conditions (pH>7) in the bentonite barrier. High pH values in water in contact with the copper canister surface are favorable in terms of preventing corrosion.

In this section, gas permeability, organic matter in bentonite and microbial corrosion is discussed. Further characteristics, viz., hydration (*i.e.,* the process by which water is taken up or adsorbed by the buffer), cation effects and smectite to illite conversion, which conceptually could also have been included here, have been relegated to the following section, due to their intimate connection with the pore water chemistry of bentonite.

5.2.1 Gas permeability

An important requirement of the buffer is that it should prevent gas migration to and from the canister. It is worth pointing out that bentonites have successfully contained natural gas deposits in the North Sea on a geologically significant time scale, demonstrating the admirable capabilities of the proposed type of buffer material as a gas barrier (Pusch, 1998). In the deep repository, it is generally assumed that oxygen will be flushed out by water or consumed in chemical reactions in connection with, or following, sealing of the shafts and tunnels. Dissolved oxygen, $O_2(aq)$, can be consumed in a heterogeneous reaction, *i.e.,* a reaction involving more than one of the phases – gas (g), liquid (l) and solid (s). Reaction occurs with the accessory mineral pyrite, $FeS_2(s)$, present as an impurity in bentonite (see Table 1)

$$4\,FeS_2(s) + 15\,O_2(aq) + 14\,H_2O(1) \rightarrow 4\,Fe(OH)_3(s) + 16\,H^+(aq) + 8\,SO_4^{2-}(aq)$$

Note that this reaction leads to formation of sulfuric acid, shown as dissolved protons, $H^+(aq)$, and sulfate ions, $SO_4^{2-}(aq)$, in the above formula. The acidity is neutralised by immediate reaction with calcite, thus avoiding any substantial lowering of the pore water pH. All oxygen is effectively removed from the bentonite in this way, thus the conditions in the deep repository are described as anoxic, *i.e.,* free from oxygen.

Under such anoxic conditions, the canisters are expected to have a substantial lifetime (Horseman *et al.,* 1999). Should the copper sheath be penetrated, corrosion of iron (Fe) in the steel insert will be mediated by water reduction (Gallé, 2000)

$$3\,Fe(s) + 4\,H_2O(l) \rightarrow Fe_3O_4(s) + 4\,H_2(g)$$

leading to the formation of magnetite (Fe_3O_4) and hydrogen (H_2) gas. Additional gaseous products will be generated by radioactive decay of the spent fuel (such as radon gas) and by radiolysis of water. Radiolysis means that the chemical bonds linking the oxygen and hydrogen atoms in liquid water are broken, by energy emitted by nuclear decay processes in the spent fuel, producing $H_2(g)$ and $O_2(g)$.

In a worst case scenario, accumulation of a high gas pressure could compromise the stability of the bentonite by inducing crack formation and propagation through the buffer (Horseman *et al.,* 1999). This would, in turn, provide a convenient escape route for dissolved radionuclides present in solution.

At lower gas pressures, migration is only possible via diffusion of gas molecules in the buffer pore water. Diffusion is a slow process, and as more gas is evolved, the solubility limits of gases in the pore water will ultimately be exceeded, allowing a build up of pressure at the canister. In order for gas to penetrate the buffer, the gas pressure must exceed the total pressure exerted by the swollen, water saturated bentonite and the groundwater (Gallé, 2000; Horseman *et al.,* 1999). The swelling pressure of bentonite increases with the density of the dry, pre-compacted material. For densities greater than 1.6 g/cm^3 (the density of water is about 1 g/cm^3), the swelling pressure will exceed 7 MPa. The groundwater pressure at a repository depth of 500 m will be about 5 MPa. Thus the gas pressure generated by anoxic corrosion, radioactive decay processes and radiolysis of water must be greater than 12 MPa before substantial cracking of the buffer can commence.

In the absence of such cracks, water saturated bentonite is completely impervious to gas. It has also been established that such damage is not permanent, as clay is self-sealing when the gas-filled voids are gradually flushed with inflowing groundwater. However, as long as gas-filled pockets persist in the bentonite, the threshold pressure for gas breakthrough will be significantly lower than the initial value (Horseman *et al.,* 1999).

At present, it is still impossible to accurately predict the risks for gas breakthrough. A simple model has been used to estimate the hydrogen gas pressure generated by anoxic corrosion (Gallé, 2000), indicating that this is of the same order of magnitude as the clay barrier breakthrough pressure. However, the model described an envisaged scenario in the type of deep repository currently being evaluated in France and several other countries, and the conclusions reached are not directly applicable for risk assessment of the KBS-3 concept, for two major reasons.

Firstly, a European bentonite (Fo-Ca clay) is under consideration for the buffer material in the French HLW disposal strategy. As noted above, this is a calcium bentonite and differs from MX 80 in its' water adsorbing capacity and permeability (Hoeks *et al.,* 1987). An ideal buffer should be highly adsorbant and impermeable, MX 80 being better on both counts. Secondly, and certainly of greater importance, is the fact that the proposed French canister consists of steel alone, without the copper cladding of the KBS-3 design. This means that anoxic corrosion will proceed over the entire surface of the steel canister in the former, rather than at discrete points where the copper has been penetrated in the latter. Consequently, the rate of gas generation and accumulation will be limited by physical constraints in the KBS-3 canister design, providing an additional safety margin.

It should also be noted that, as long as the buffer is completely water saturated, manufacturing joints have no effect on gas permeability (Horseman *et al.,* 1999). This is important because, as long as the $H_2(g)$ generated is confined by the bentonite at the point of production, the anoxic reaction will reach equilibrium before sufficient gas pressures to break through the buffer are realised. At equilibrium, no further corrosion can take place unless the gas is removed. The copper sheath around the steel insert is pliable and any gaps between the canister and the bentonite will be sealed by swelling of the buffer during water uptake. This will prevent loss of generated $H_2(g)$, stopping the corrosion process. The gas produced will fill any breaches in the copper cladding, hindering further transport of water to the steel, again reducing the rate of corrosion.

Nevertheless, the importance of gas migration in the bentonite barrier as a potential limitation of the buffer should not be underestimated (Gallé, 2000). The need for full-scale experiments to reconcile areas of uncertainty concerning gas breakthrough in bentonite has been expressed (Horseman *et al.*, 1999). The necessary resources should be made available to ensure that these uncertainties are adequately resolved in the near future.

5.2.2 Organic matter in bentonite

Clays contain small amounts of organic matter, residues of dead biological material remaining after mineralisation. The organic matter in clay minerals consists of approximately 50 % by mass of carbon, so-called humic substances constituting 70–80 % by mass of this material (Cho *et al.*, 2000). Humic substances are weak acids that are largely dissociated, forming negatively charged organic anions, at the neutral to alkaline pH values found in the pore waters of saturated bentonite. As clay surfaces are, themselves, negatively charged, humic substances are bound through multiply charged cations, such as calcium (Ca^{2+}), that form a bridge between the organic anions and the mineral substrate.

The remaining organic matter is not soluble in water, but binds hydrophobically to the surface complexed humic substances. That is to say, the hydrophobic (literally meaning water fearing) organic matter ”sticks” to the humic substances, which are, in turn, attached to bentonite particle surfaces. Humic substances are also a form of organic matter, but contain acidic groups, rendering them water solubility. They therefore behave in similar fashion to soaps.

Cho *et al.* (2000) studied the dissolution of organic matter present in calcium bentonite of Korean origin, in the temperature range from 20 to 80°C, under the influence of irradiation. As expected, the concentrations of dissolved organic matter increased with temperature. Irradiated bentonite samples exhibited a greater tendency to release organic matter to the aqueous phase, as a result of partial decomposition to water-soluble products. Most, though not all, radionuclides are present as positively charged ions in solution and are therefore sorbed on negatively charged mineral surfaces, retarding their migration through the bentonite (see ”Radionuclide transport”, below). Being negatively charged, dissolved organic acids can also form soluble complexes with many radionuclides. These so-called organically complexed radionuclides will not interact as efficiently with mineral surfaces, thus their rates of transport towards the bedrock as neutral species may be enhanced.

It appears that the concentrations of dissolved organic matter in bentonite pore water are lower than those typically present in groundwater. Consequently, there will be an influx of dissolved organic matter from the groundwater into the bentonite over time. This means that measured diffusivity values, based on experiments made using organic-free water, possibly represent lower limits. It can be concluded that insufficient data are currently available to assess the importance of radionuclide transport as organic complexes through bentonite. On the other hand, there is ample evidence that humic substances form very strong complexes with numerous actinides (thorium, uranium, americium, curium) and fission products formed during the decay of the spent nuclear fuel (Tipping, 1998).

5.2.3 Microbial corrosion

In groundwaters of the composition likely to be found in a Swedish deep repository, sulphide ions and residual dissolved oxygen are probably the only two substances capable of corroding copper. It is anticipated that the concentrations of both will be low in the immediate vicinity of the canister. Nevertheless, the possibility that that sufficient sulphide ions could be generated, by sulphate-reducing bacteria, to induce copper corrosion at an accelerated rate must be considered (Pedersen, 1999). (Note that it is the extremely toxic gas, hydrogen sulphide, that is actually produced, rather than sulphide ions directly.)

Sulphate-reducing bacteria have been encountered in deep geological formations in Sweden. However, it has been established that hydrogen sulphide, generated by microbes in the bedrock, was inefficiently transferred across the interface to the bentonite. Water saturated bentonite also offers considerable resistance to diffusional gas transport from the bedrock to the canister (or vice versa, see ”Gas permeability” section). Sulphide sources outside the buffer are therefore unlikely to pose a major problem.

Another possibility is microbial sulphide production inside the bentonite. Again, the buffer presents a considerable hurdle since, in highly compacted bentonite, water is strongly bound to particle surfaces and is thus unavailable for active lifeforms. Bentonite is an exceedingly hostile environment for microbes due to limited moisture, nutrient and energy sources. Accumulation of hydrogen sulphide, owing to the poor, diffusional, gas transport characteristics of saturated bentonite, also contributes to making the buffer uninhabitable, as this gas is toxic to microbes (Pedersen, 1999).

It should be recognised that the gaseous impermeability and the water binding properties of montmorillonite, constituting the buffer material, are instrumental in inhibiting microbial corrosion processes. In the event of conversion of montmorillonite to illite (see section on ”Illitisation”), it seems likely that these protective features of the bentonite would be partially, or completely, lost. Therefore, the ability of illite to sustain the sulphate-reducing microbial strains found in deep geological formations representative of a HLW repository warrants study.

5.3 Pore water chemistry

In hydrated MX 80 bentonite, the ions initially present are predominantly sodium (Na^+), calcium (Ca^{2+}) and chloride (Cl^-), with minor contributions of potassium (K^+), magnesium (Mg^{2+}), sulfate (SO_4^{2-}) and hydrogen carbonate (HCO_3^-). The existence of charged species in solution confers electrical conductivity, which is proportional to the ionic strength of the water, a function of the concentrations of all ions present and their charges. When bentonite is exposed to groundwater having a different chemical composition to that of the pore water, ions will be transported into and out of the material, in order to equalise the concentrations and eliminate ionic strength gradients. An understanding of the pore water chemistry and ion transfer in bentonite is vital in the assessment of repository behaviour. Over time, the pore water chemistry will evolve, reflecting interactions between the species present at the time of closure, and ions entering with the groundwater during saturation (Muurinen & Lehkoinen, 1999).

5.3.1 Hydration

In the repository setting, the bentonite will continue to adsorb groundwater seeping through the bedrock until it becomes completely saturated. This is called hydration, and will cause the bentonite to swell, closing up construction joints and providing a dense barrier between the bedrock and the canister in the deposition hole.

The self-sealing properties of bentonite are evidenced by a series of experimental observations (Oscarson *et al.*, 1996). It was found that both hydraulic conductivity and diffusivity values for bentonite samples containing fabrication joints and defects were identical, within the limits of experimental uncertainty, to those of ”perfect” material following water saturation. Diffusivity values are used to describe, mathematically, the transport of material induced by a concentration gradient. Ideally, diffusivity values should be as close to zero as possible, as this would mean that transport of radioactive material, by diffusion through the bentonite buffer, would be extremely slow. In other words, radionuclides released from a damaged canister would not reach the outer environment within the time scale required for them to decay to non-radioactive daughters.

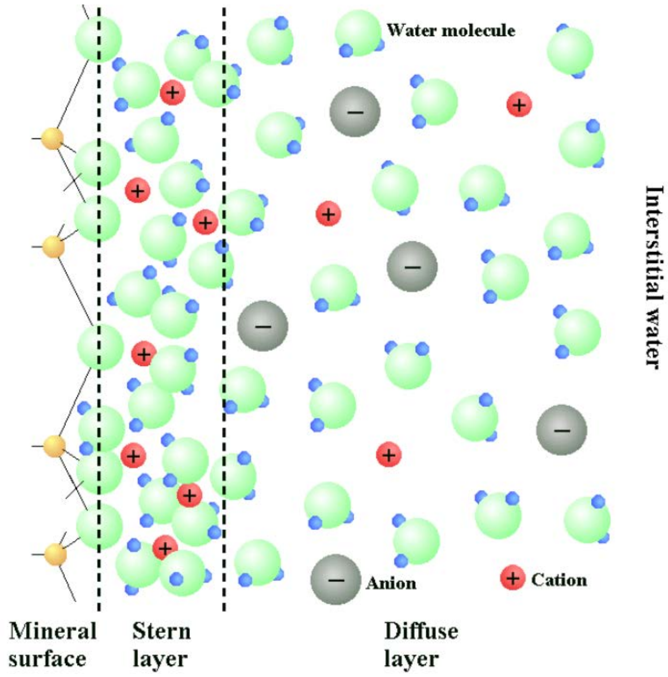

Figure 3. *Schematic illustration of the diffuse double layer. The montmorillonite mineral surface is negatively charged due to the partial substitution of divalent magnesium for trivalent aluminium in the octahedral layer (see Figure 2). Water molecules and cations are strongly sorbed to the mineral surface, forming the Stern layer from which anions are excluded. In MX 80 bentonite, the proposed buffer material in the KBS-3 concept, sodium ions (Na^+) are responsible for balancing the negative surface charge. In time, the influx of calcium rich groundwater will lead to replacement of Na^+ with calcium ions. The chemical potential in the Stern layer decays gradually, through the diffuse layer, with increasing distance from the surface of the clay particle. Interstitial or bulk water exists between particles, out of range of the attractive forces exerted by the montmorillonite surfaces. In highly compacted bentonite, the particle surfaces are so close together that there is little or no interstitial water present. Note that the anions and cations present in the diffuse double layer are not necessarily singly charged.*

In highly compacted clay with low porosity, water sorbed at pore surfaces is immobile (Cho *et al.*, 1998). Bulk water, on the other hand, which does not experience the electrostatic attraction in the Stern layer (see Figure 3) and is thus mobile, may flow in the presence of a sufficiently high pressure gradient. Radioactive decay of nuclear waste in the canister will generate a temperature-, and thus pressure-gradient in the buffer. At the elevated temperatures close to the outer canister walls, the hydraulic conductivity in water saturated bentonite will increase. This will result in an enhanced rate of transport of water and dissolved species from the canister, towards the bedrock, through the buffer.

Water molecules residing in the Stern layer may be thermally desorbed, *i.e.,* released from the bentonite particle surfaces, at higher temperatures. This would increase the permeability owing to an increase in the effective flow channel volume. Both the viscosity (a fluids' resistance to flow) and the density of water decrease as the temperature is raised, which also contribute to an increase in the hydraulic conductivity. Of these factors, Cho *et al.* (1998) concluded that the effect of viscosity was the most important. It was suggested that, as long as the temperature of the buffer material was maintained below 100°C, the hydraulic conductivity would be low enough to prevent significant radionuclide transport by advection from the nuclear waste to the bedrock.

At elevated canister outer surface temperatures, steam could be generated. Investigations have shown that steam markedly decreases the swelling capacity and increases the permeability of loosely compacted bentonite. These effects could adversely affect the buffers' transport properties. However, the influence of steam on the highly compacted bentonite to be used in a deep repository appears to be of minor impact (Oscarson *et al.,* 1996). Higher temperatures would also cause the buffer near the canister to dry, shrink and crack due to heat-induced moisture migration along the temperature gradient away from HLW container. Crack formation can also be initiated by hydraulic fracturing or gas accumulation (see below). These points all emphasise the importance of the self-sealing properties of the buffer, and indicate that bentonite is as ideal a candidate as can be hoped for. In the KBS-3 concept, the canister is designed to limit the temperature at the outer copper surface to a maximum of 100°C during the initial storage period. As time passes and the fuel cools, the potential for thermally-driven water transport will drop.

The long-time effectiveness of the clay barrier may be influenced by some short-term processes. One of these processes is the accumulation of solutes transported through the bentonite during the hydration step. Another one is the eventual formation of cemented zones (precipitation) or the influence of pore water composition on the hydromechanical properties of bentonite (Martin *et al.,* 2000). Furthermore, these processes are probably influenced by the thermal gradient from the canister and outwards in the repository, which can give rise to a concentration gradient.

The transport of solutes (dissolved substances, mainly salts) into, and within, the clay barrier is expected to have special relevance for the clay surface properties. The arrangement of clay particles is changed, and the intracrystalline swelling capacity can be deteriorated, by high saline concentrations (Abdullah *et al.,* 1999). As a consequence, the permeability may increase in compacted natural clays. In addition, salinity is an accelerating factor in the processes of corrosion of the metallic components of the canister, and of degradation of interfaces between bentonite and cement. It is therefore important to describe the process of migration of salts in compacted bentonite, to assess the risks for salt accumulation in specific areas, particularly at the canister walls, and to obtain transport parameters for the conditions of the thermal gradient imposed in the compacted bentonite.

Once the bentonite is saturated, a concentration gradient is generated as the pore water strives to come to equilibrium with the groundwater, solute transport being reported to be mainly diffusive (Martin *et al.,* 2000). However, under such conditions, water may also be transported by osmosis. In this process, water migrates away from the region of lower electrical conductivity (lower dissolved salt concentration) in the bentonite pore water, as a result of the chemical gradient. After some time, the rate of change of differential hydraulic pressure drops to a minimum when the salt concentrations in the pore- and groundwaters have been equalised. Subsequently, groundwater will start to flow into the bentonite again, causing an increase in the hydraulic conductivity and swelling of the buffer.

The drop in the differential hydraulic pressure is neither a result of collapse of the diffuse double layers due to increased ion concentrations within the bentonite, nor failure of the clay to perform as a semi-permeable membrane. If either were true, the ionic gradient would diminish rapidly (Keijzer *et al.,* 1999). Osmosis will result in transient and dynamic water and solute exchange in the period immediately following closure of the repository. It appears unlikely, however, that any profound effects on the performance of the buffer will result.

The apparent diffusion coefficients of ionic species generally increase with temperature. The mobilities of cations such as magnesium, calcium, strontium, sodium and potassium decrease when saline water is present at low temperature. Zinc ions appear to represent an exception, showing higher mobility in a low-temperature, saline environment (Martin *et al.,* 2000). Ions with higher mobilities, notably Cl^- and Ca^{2+}, are strongly influenced by the rise of temperature. It was observed that Na^+, also being a highly mobile ion, exhibited a smaller temperature effect. Its' mobility is constrained because Na^+ is the major cation in MX 80 bentonite, and the one that compensates most of the negative surface charge. Diffusion of Na^+ is highly dependent on the mobility of anions such as the divalent sulfate ion, whose apparent diffusion coefficient is relatively independent of temperature and salinity, and is an order of magnitude below those of Cl^- and Ca^{2+} (Martin *et al.,* 2000).

The anions and the cations that compose the salts present in bentonite pore waters do not have the same diffusion coefficients. If the anion is the more rapidly diffusing of the two, a negative charge is generated in the direction of the flow. This delays the rate of diffusion of the anion and increases that of the cation. The electric potential gradient increases until the mobility of both species is the same.

5.3.2 Cation effects

We now turn to the mechanisms responsible for bentonites' ability to adsorb water. As water molecules are dipolar, they are electrically bound to the tetrahedral sheets'outer surfaces in diffuse double layers, as illustrated in Figure 3. These water layers also accommodate cations that are attracted by the negatively charged clay mineral surfaces. The various types of interaction between clay particles depend on the cations present in the exchange complex. Competing kinds of interaction are generated through the repulsive nature of overlapping diffuse double layers and the attractive van der Waals' forces, leading to various types of interparticle associations. In bentonites with primarily potassium (K^+) and calcium (Ca^{2+}) ions providing surface charge compensation, rather similar moisture-dry density relationships are observed. Both exhibit higher maximum dry densities and smaller optimum moisture contents as compared to sodium bentonite (Abdullah *et al.*, 1999).

Monovalent cations, particularly Na^+, are less strongly attracted than divalent cations, such as Ca^{2+}. The diffuse double layer is compressed drastically due to the presence of divalent calcium ions in the exchange complex leading to a dominating effect of attractive van der Waals' forces between clay particles. The net attractive force leads to face-to-face aggregation, *i.e.*, small crystals lump together and form larger particles and, consequently, particles grow due to this "stacking" tendency (Abdullah *et al.*, 1999). As the thickness of the diffuse double layer is greater in the presence of Na^+ and Ca^{2+}, assuming equal

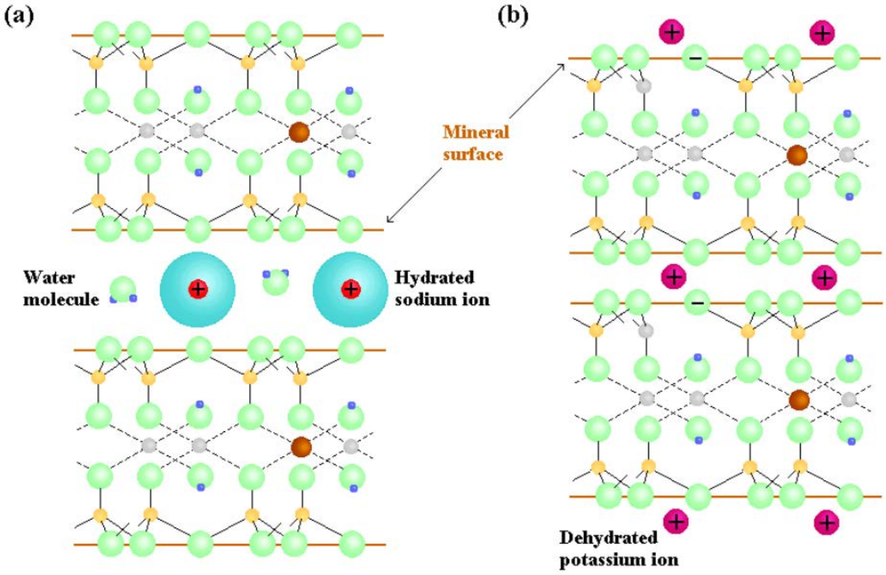

Figure 4. *Comparison of (a) smectite (montmorillonite) and (b) illite clay minerals. In MX 80 bentonite (a), hydrated sodium ions are sorbed on the surface to neutralise the surface charge. The large ionic radii of these ions help to maintain the spacing between the smectite layers, allowing water molecules to penetrate. This accounts for the large swelling- and water adsorbing-capacity of sodium bentonite. As more water is adsorbed, the spacing between mineral particles grows, and the diffuse double later shown in Figure 2 develops. Should illitisation occur, trivalent aluminium will, to some extent, replace tetravalent silicon in the tetrahedral layers (see Table 1), leading to a more pronounced negative surface charge (b). With the ingress of potassium rich groundwater, surface charge neutralisation will be made by potassium ions. These ions can relatively easily shed their waters of hydration. The resultant, dehydrated, potassium ions are effectively incorporated in the spacing between the tetrahedral layers of adjacent mineral particles. Illite sheets form closely-packed stacks, the spacing being insufficient to allow entry of water molecules. Thus illitisation, in combination with a supply of potassium ions from infiltrating groundwater, would cause shrinkage of the buffer material.*

concentrations of both, more water molecules can be accomodated between the stacked particles. Sodium bentonite therefore exhibits a greater swelling capacity than the calcium form. Sodium ions are the dominant cations in MX 80 bentonite, which appears to be one of the most important reasons for its' preference as a buffer material by SKB. The water adsorbing capacity of European bentonite is typically two- to threefold inferior to that of MX 80, as a result of the much higher calcium content of the former (Hoeks *et al.*, 1987).

Potassium bentonite behaves in a similar fashion to calcium clay, owing to aggregation. Aggregation means that the tetrahedral sheets of neighbouring clay particles are tightly packed, cations between the particles being bonded to both (conceptually similar to the case for illite particle stacking illustrated in Figure 4). However, the mechanism differs from the calcium clay. As a consequence of the substitution of trivalent aluminium by divalent magnesium (Figure 2), a high electric polarisation exists near the clay particle surface, especially within the Stern layer (i.e., the component of the diffuse double layer closest to the tetrahedral sheet, see Figure 3).

Electric polarisation coupled with the low hydration energy of the potassium cations (8.56 kJ/mol) causes the hydrated K^+ to shed all its' waters of hydration. As a result, K^+ (with an unhydrated radius of 133 pm, 1 pm = one billionth of a mm; hydrated radius 430 pm) enters hexagonal holes in the tetrahedral sheets (having a radius of 132 pm) and provides a strong linkage between clay particles. This explains the tendency of potassium-clay particles to aggregate and form stacks.

The potassium-linkage is stronger than the weak (van der Waals') forces exerted between calcium bentonite particles. Stacks are much larger than the discrete clay particles; the larger the particles, the smaller the elasticity of the potassium clay. That is to say, the ability to self-seal cracks in the structure is reduced. Potassium-linkage causes a drastic, approximately fourfold, decrease in the specific surface area of the clay and a significant deterioration of the water adsorbing capacity (Abdullah *et al.*, 1999). A reduction in the cation exchange capacity also results, meaning that the buffer materials' ability to sorb radionuclides released from a damaged canister will be impaired.

For the sodium clay, the thick diffuse double layer gives rise to an important repulsive force that overwhelms the van der Waals' attractive forces between particles and causes net repulsion. This situation is conducive to a large swelling potential or water absorbing capacity. Clearly, the type of cation associated with the bentonite will have a pronounced effect on the buffer behaviour.

Although sodium bentonite produces a thick diffuse double layer, high concentrations of Na^+ in pore waters actually reduce its' thickness. The sodium ions become electrostatically bound to, or attracted to the surfaces of, neighbouring particles. This causes the clay to contract, similar to the situation for illite illustrated in Figure 4 and discussed below. When subjected to highly saline groundwaters, bentonite will therefore lose its' high water adsorbing capacity and some of its' elasticity.

5.3.3 Smectite to illite conversion

Among the chemical forces to which bentonite will be subjected in the deep repository, exchange of Na^+ by K^+ and Ca^{2+}, as described above, will be one. A particular feature of montmorillonite is that, in the tetrahedral layers, there is negligible substitution of tetravalent silicon (Hemingway & Sposito, 1989). However, in the repository environment, which will initially reach temperatures approaching 100°C in the immediate vicinity of the canister, conversion of the mineral phases present may be thermally induced. Dissolution of most solids is enhanced at higher temperatures, leading to the formation of hydrated, dissolved species that may crystallise in cooler regions as new mineral phases. This combination of dissolution and crystallisation processes in the repository may lead to the conversion of montmorillonite to an illite mineral (Hökmark *et al.*, 1997).

Illites are characterised by a substantial degree of substitution of trivalent aluminium, for tetravalent silicon, in the tetrahedral layers (see Table 1 and Hemingway & Sposito, 1989). Negatively charged sites are thus present in the basal plane (compare montmorillonite and illite structures in Figure 4), permitting the formation of ionic bonds between the tetrahedral sheets and cations in the pore water.

The dimensions of the tetrahedral layers in illites are such that dehydrated potassium ions are readily incorporated and extremely tightly bound (Figure 4). Bound potassium ions also constitute linkages to neighbouring illite particles, leading to aggregation and formation of stacks. As such, the ability to bind water molecules in a diffuse double layer is more or less eliminated following conversion of smectite to illite. Consequently, illites lack the water adsorbing capacity and elasticity of montmorillonite. Illitised bentonite would thus be incapable of self-sealing cracks induced by excess gas pressure release or mechanical shear stresses caused by seismic activity in the bedrock.

Substantial conversion of montmorillonite to illite in the buffer will require a considerable time scale. Even in unfavourable circumstances with respect to the bentonites' longevity, including a steady supply of potassium-rich groundwater, rapid reaction rates and elevated temperatures, complete conversion is not expected to occur over the period of canister heating by the exothermic fuel decay processes occurring within (Hökmark *et al.*, 1997). The limitation for illitisation is provided by restrictions in the rate of potassium transport from its' source in the surrounding groundwater or the repository backfill.

It must, at this point, be mentioned that conclusions concerning this conversion are based on a model of the process. Considerable uncertainties remain regarding the kinetics of the process and the concentrations of potassium in the backfill. So further investigations of the illitisation process are required.

Questions remain regarding hydraulic conductivity, gas permeability and radionuclide transport in illitised bentonite.

5.4 Radionuclide transport

The final role of bentonite as an engineered barrier comprises its' ability to retard the transport of radioactive material escaping from a canister completely penetrated through the copper cladding and steel insert to the spent fuel. However, bentonite must also serve to contain radioactive products formed in the steel through neutron activation of the construction material (Adeleye *et al.*, 1995). Two such products are radioactive chromium (^{51}Cr) and cobalt (^{60}Co). Fortunately, radionuclide transport through bentonite has been quite thoroughly investigated. Thus there is a wealth of data available on the transport mechanisms, as well as the effects of dry clay density, type of cation associated with the bentonite and pore water chemistry.

5.4.1 Diffusion pathways

The cation incorporated in the bentonite for neutralisation of surface charge is decisive for the rate of radionuclide transport. In sodium bentonite, the proposed buffer in the KBS-3 concept, pores in the material have been shown to be smaller than in calcium bentonite (Choi & Oscarson, 1996). Diffusion pathways in the calcium clay are therefore less tortuous and radionuclides can flow more easily as they are less likely to interact with the mineral surfaces.

At this point it is worth distinguishing between the two, distinct mechanisms of diffusion at work in compacted clays.

Firstly, there is diffusion in the interstitial water. Interstitial water experiences little or none of the surface charge on the bentonite, existing outside or on the outskirts of the diffuse layer (refer to Figure 3). In clays of lower densities and larger pore sizes, diffusion in the interstitial water is the dominant pathway for radionuclide transport.

Secondly, there is surface diffusion, leading to transport in the Stern layer. As water molecules are densely bound to the surface in the Stern layer, the viscosity is higher than in interstitial water, retarding mobility. In addition, cationic species present will be sorbed on the surface, further reducing their mobility. In compacted sodium bentonite, surface diffusion of cations is the most important mechanism of radionuclide migration (Cheung, 1989; Cheung, 1990).

Viscosity is one factor determining the rate of radionuclide transport through the bentonite. The viscosity of bulk water is known to decrease monotonically with increasing temperature, and thus the rate of diffusion will be enhanced, as borne out by experimental evidence (Martín *et al.,* 2000). Temperature effects are also coupled with changes in hydraulic conductivity (Cho *et al.,* 1998) and sorption efficiency.

5.4.2 Surface processes

Clay minerals are one of the most common types of ion exchange materials. *Ion exchange* is a process whereby equivalent exchange takes place between ions in solution and those on the material surface, *i.e.,* the law of mass action applies. For example, release of two monovalent sodium ions from the surface of bentonite is balanced by exchange with one divalent calcium ion from solution. *Surface complexation,* on the other hand, is a process in which ions are bound to the surface by chemical forces, and does not cause displacement of other ions in equivalent quantities. *Surface precipitation* is a third process, where a chemical reaction occurs on the surface because its' conditions differ from those in solution. When the precise nature of the actual process responsible for the transfer of ions existing freely in solution to the surface is neglected, ion exchange, adsorption and surface precipitation can be commonly described as sorption (Nagy & Kónya, 1988).

Cation exchange can be classified as being selective or non-selective. In the former case, certain cations, which only weakly bind waters of hydration, such as K^+, rubidium and cesium (Cs^+) ions, are preferentially sorbed from solution by the clay mineral surface. In non-selective exchange, strongly hydrated cations, such as Na^+, Mg^{2+} and Ca^{2+}, are sorbed in proportion to their relative concentrations in solution. Various spectroscopic techniques have been used to probe the cation exchange sites in clay minerals. Results for the naturally occurring, non-radioactive $^{133}Cs^+$ isotope indicate that there are primarily two surface environments capable of exchange (Weiss *et al.,* 1990; Kim *et al.*, 1996). One results in relatively tight bonding in the Stern layer, the other being considerably weaker, retaining Cs^+ in the diffuse layer. Most of the sorbed Cs^+ is found on the basal clay particle surfaces, although edge sites are also important. Referring to Figure 2, it can be seen that the oxygen at the upper left of the basal plane is shown as being bonded to only one silicon atom. This oxygen will have a single negative charge and is an edge site. (Note that the radioactive cesium released from the Chernobyl reactor incident consisted of two, non-naturally occurring isotopes, ^{134}Cs and ^{137}Cs.)

Surface complexation implies the formation of a strong, covalent, chemical bond between the ion and a reactive surface group. Hydroxyl groups (-OH) are the most important reactive moieties in the structure of montmorillonite, although additional groups are also available in bentonite, as a result of the presence of some accessory minerals (see Table 1). Since the pore water in bentonite is alkaline, surface precipitation of radionuclides as hydroxides is also important for the retention and retarding capabilities (Ochs *et al.,* 1998).

It may be difficult to distinguish between the various sorption mechanisms, although some exceptions are to be found in the literature. Adeleye *et al.* (1995) concluded that chromium (Cr^{3+}) is sorbed by surface complex formation. Higher valent cations, such as Cr^{3+}, are more difficult to displace following sorption. It was also pointed out that the efficiency of sorption of a specific ion depends on the presence of other charged species in solution. This synergism was explained by surface complexation predominating over simple ion exchange. Eriksen *et al.* (1999) provided an explanation for the behaviour of Co^{2+} in light of their finding that complete immobilization of this ion occurred on compacted MX 80 bentonite. With increasing concentrations of Co^{2+} and at higher pH values, surface complexes with hydroxyl groups and hydroxide precipitates are formed. This is certainly in line with the expected chemistry of cobalt in aqueous solutions.

Chemical considerations are of great importance for radionuclide transport in the repository environment. So far in this review, most attention has been devoted to cations, but there are a number of elements whose chemistry suggests that they will exist, to varying extents, as anionic species.

Anions migrate exclusively through diffusion in the interstitial water. Therefore, among the radionuclides present in spent fuel or generated through fission processes, anionic species have the greatest potential to reach the biosphere before decaying to insignificant levels. The element iodine is of particular concern, as one of its' radioactive isotopes, ^{129}I, has such a long half-life (1.7×10^7 year).

In the repository, iodine will exist primarily as iodide (I^-) ions, which are strongly repelled from the clay surface and therefore transport can not be hindered by surface -sorption and –diffusion (Hoh *et al.,* 1992). It is the latter two effects that are operative in slowing the progress of cations through the bentonite buffer. There is also general agreement that anion exclusion limits the rate of diffusion. Being negatively charged, anions resist entering the buffer unless adequate flow channels are available. As noted above, pores in sodium bentonite are very small and the amounts of bulk water (*i.e.,* water present outside the diffuse layer shown in Figure 3) are relatively low, which means that the transport rates of anions are limited. Although diffusional transfer will increase as sodium bentonite is gradually converted to the calcium form by ion exchange with Ca^{2+} present in infiltrating groundwater, the changes are not large enough to seriously compromise the efficiency of the barrier. Experimental data suggest that diffusion coefficients are less than ten times greater in calcium bentonite (Choi & Oscarson, 1996).

Recently, the effects of canister corrosion products on processes occurring in the deep repository have come under scrutiny. The presence of magnetite produced during anoxic corrosion of steel enhances the formation of colloids, which could have two possible effects on radionuclide transport. The first is that larger colloids, incorporating radioactive isotopes, will be deposited in pores in the bentonite. This should effectively reduce transport of all species by limiting the number of pathways available for radionuclide flow. The second is that small, uncharged colloids will be able to migrate, relatively unhindered by electrostatic effects, freely through the buffer, enhancing the rate of radionuclide migration. Fortunately, it appears that the first effect is dominant (Inagaki *et al.,* 1998; Idemitsu *et al.,* 1998).

5.5 Conclusions

Bentonite would appear to fulfil the criteria placed on the buffer material in a deep repository for high-level nuclear waste. In order to function correctly at the time of closure, it is important that the bentonite contains two accessory mineral impurities, namely pyrite and calcite. Pyrite will consume any residual oxygen remaining in the repository, thus providing the desired anoxic environment conducive to maintaining the integrity of the copper canister. However, strong acid is produced in this process, requiring that sufficient calcite is present for complete neutralisation. Therefore it is important that the bentonite material meets certain quality specifications.

The presence of organic matter in bentonite is unlikely to cause any problems during the initial stages of HLW storage. Bacterial strains which could utilise the organic matter as a source of energy appear to be incapable of surviving in the hostile environment posed by bentonite. Ongoing research on the threats from microbial corrosion should clarify remaining uncertainties in this area. On the other hand, organic matter could play a significant role in facilitating radionuclide transport through the buffer. Neutral complexes migrate at a rate limited by the hydraulic conductivity of the buffer and could enhance the dispersion of radioactive material in the repository. Although the formation of neutral organic matter – radionuclide complexes definitely occurs, the sizes of such species are decisive for their transport in the densely compacted bentonite.

The attractive features bestowed by bentonite would be lost in the event of large scale conversion of the smectite clay (montmorillonite) to illite. Therefore, data on gas permeabilty, hydraulic conductivity, radionuclide transport and microbial migration in illite should be compiled and evaluated. At our current level of understanding, there are indications that illitisation will not occur to such an extent that repository safety will be compromised over the anticipated lifetime. On the other hand, there is little reason to doubt that sodium bentonite will be converted to the calcium form by the influx of calcium-rich groundwater. Therefore, greater consideration should be given to the properties of calcium bentonite.

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6 Extrapolation of canister material properties[1]

6.1 Background

Before the spent nuclear fuel repository reaches a state of permanence in its rock environment, it must pass through several stages of evolution:

- deposition and simultaneous rock engineering work to construct new areas;
- a possible observation period when tunnels and shafts are kept open and
- a transition period after closure when the original conditions in the bedrock are restored.

It is well known that the canisters' capability of isolating the fuel from the groundwater in the rock is important for the long-term safety of the repository. It is less known that the canisters are also of considerable importance as a barrier while spent fuel is being deposited and during the subsequent transition period, when the groundwater is still oxygenated by the open tunnels. Furthermore, this is the time when stresses on the canisters are the greatest.

When the canister is assembled, a millimeter-wide gap is provided between the copper shell and the insert to allow the insert to be lowered into the copper shell. After deposition, the copper shell is pressed against the insert by the swelling ben-

[1] This chapter was written by KASAM member, Professor Rolf Sandström, Materials Technology, Royal Institute of Technology, Stockholm. Nils Rydell MSc. Eng., Senior Technical Advisor to KASAM also participated in the preparation of this chapter.

tonite buffer. The copper undergoes creep and, at the same time, there is also some oxygen remaining in the groundwater. These load conditions can cause metals, such as steel and copper, to crack. This phenomenon has been studied for some time now in pressurized steel and, more recently, in copper including copper which is to be used for spent fuel canisters. It takes a few hundred years for the shell compression and reduction of oxygen to occur. In relation to the long future of the repository, this is a very short time but, nevertheless, a very long time compared with the time in which materials investigations can be conducted in laboratories.

This chapter describes the problems of extrapolating data from such short-term measurements to hundreds of years or more. The state of the canister after the initial period of shell compression will be its initial state on the geological time scale and is therefore of decisive importance for the analysis of repository safety.

6.2 Major challenge

Industrial process facilities are normally designed for a lifetime of 10 to 25 years. A number of different types of materials data are required for the construction of such facilities. Since laboratory experiments are often limited to a few months or, in certain cases, to a few years, laboratory data cannot be directly used. Furthermore, previous operating experience is not reliable. The rapid pace of industrial development means that new types of materials are being developed all the time. For example, steels are usually replaced after three to five years on average. This means that conducting laboratory experiments for timescales exceeding a few years on materials that are likely to be replaced within the same period of time is dubious since the investigated materials will no longer be in use.

The way of handling this problem is to extrapolate results from laboratory experiments conducted in the short or medium term. Performing an extrapolation is always a challenge. The results can seldom be verified by experiment or through operating experience before it is too late. A special methodology must be followed and this will be illustrated below. It must also be ensured that the controlling mechanisms in the laboratory experiments are relevant on industrial timescales.

In the case of the materials in the canister that SKB is planning to use for the final disposal of spent nuclear fuel, the extrapolation of materials data from laboratory experiments is a considerable scientific challenge. The shell of the canister, which will be manufactured from pure copper, will be exposed to active corrosion and creep for a few hundred years. Therefore, extrapolations must be conducted that are at least one order of magnitude greater in time than in conventional industrial applications.

Systematic methods for data extrapolation have, above all, been studied for application to creep investigations. On the basis of an ISO standard, it was previously recommended to only allow extrapolations by up to a factor of three in time, in the case of safety-related components such as pressure vessels [1]. However, over the past few years, a method has been developed which allows data to be extrapolated for much longer timescales [2], [3].

The purpose of this chapter is to present an overview of different types of procedures used for extrapolation and how these can be applied to materials for spent nuclear fuel canisters.

6.3 Time independent and time-dependent properties

To design components that are exposed to mechanical stresses, data are required for different properties, including materials strength, malleability and corrosion resistance. Basic materials strength properties such as fracture stress (the highest stress to which a material can be exposed before it ruptures) and fracture strains (the highest strain to which a material can be exposed) are essentially time-independent. The same applies to physical properties such as electrical and thermal conductivity, density and the thermal expansion coefficient. If the microstructure of the material does not change with time, these properties will remain unchanged.

Other properties, such as resistance to fatigue, are highly dependent on the number of load cycles and, thereby, on time. The same applies to corrosion resistance. Corrosion increases with time and corrosion rates must be determined. These types of properties are time-dependent.

The time-dependent properties of the greatest direct relevance to the canister material are corrosion and creep. Examples of corrosion mechanisms that can occur are uniform (or general) corrosion (an even corrosion of the metal surface) and localized corrosion (pitting corrosion on the surface of the material, where an aggressive solution is concentrated).

Creep is a long-term deformation that occurs when a load of adequate size is applied to an alloy or other material types. Unlike ordinary plastic deformation, the material continues to creep without the need for increased loading. In pure copper, this occurs significantly above 75 °C, while in steel and cast iron, the temperature must be above 450 °C for this to occur to any significant extent.

6.4 Premises and requirements for the canister material

The central materials requirements for the outer canister shell is that the corrosion resistance of the material should be adequate and that material deformation can occur without cracking [4]. Pure copper has therefore been chosen since it has a large thermodynamic stability range in oxygen-free water. This means that the material is immune to corrosion. Oxygen-free conditions are expected to prevail during most of the final disposal period. If dissolved sulphides are present in the groundwater, pure copper will no longer be immune. The material will corrode through the formation of copper sulphide and hydrogen gas. Since the supply of dissolved sulphides is very limited in the groundwater, the corrosion attack will be determined by the concentrations of dissolved sulphide in the canister's near field.

In many respects, the corrosion properties of pure copper are well documented. The occurrence of pure copper in nature as well as the existence of archeological materials that are hundreds of years old confirm the chemical stability of copper in many types of groundwater. The oxygen in the repository is expected to be used up after a few hundred years. After this initial phase, further corrosion will completely depend on dissolved sulphide in the near field of the canister. In this case, long-term corrosion behaviour can be described with the diffusion of dissolved sulphides to the copper surface, for which well-established models exist.

The intention is to place the canister in granite bedrock, at a depth of about 500 meters, in the presence of groundwater. This will result in a hydrostatic pressure of about 5 MPa. Furthermore, a pressure component must be added from the swelling of the bentonite clay buffer which is to be placed around the canisters, estimated at a maximum of 7 MPa giving a total of 12 MPa. The canister can be considered to be a pressure vessel under external pressure and the well-founded design principles that exist for this type of component can be used. The Swedish Nuclear Fuel and Waste Management Co (SKB) has established a safety factor of 2.5 which adequately satisfies the pressure vessel standards. The result is a design pressure of 35 MPa. Furthermore, the pressure from the largest expected glaciation during the final disposal period should be taken into account. This will increase the design pressure to about 45 MPa.

The requirement on the mechanical strength of the canister material is largely dependent on the selected canister design. In the KBS-3 concept, SKB has decided that the shell of the canister must be loadbearing. The external pressure is absorbed by a cylinder of nodular iron. Nodular iron is a type of highly ductile and malleable cast iron. It is planned to design the canister as a cylinder with a flat lid and bottom.

Pure copper is considered to be the most suitable canister material from the standpoint of corrosion. Under certain conditions, copper alloys can have far better corrosion properties. However, there is a risk of stress corrosion occurring, since the canister is subject to mechanical loads. Furthermore, there is a risk that alloying elements could leach out during the very long timescales involved. For this reason, pure copper is the natural candidate, due to its thermodynamic stability in oxygen-free groundwater.

For manufacturing-related reasons, it is suitable to leave a gap of between 1-2 mm between the cast iron insert and the copper shell [5]. With the gradual increase in hydrostatic pressure, the copper will be deformed and be supported by the insert. To prevent the copper from cracking, the copper must be sufficiently malleable and ductile. The total deformation is expected to be about 4% before the copper canister reaches the pressure-bearing nodular iron.

The deformation will partly occur through creep. This means that the copper material must have an adequate creep ductility. Creep testing shows that pure oxygen-free copper can have a very low ductility at temperatures over 160 °C. Data for these temperatures are needed to be able to predict properties for long periods of time. However, phosphorus-alloyed copper does not have this limitation. Therefore, SKB has selected this type of material. The reason for the low creep ductility of oxygen-free copper is that the material is highly sensitive to small concentrations of sulphur. Surface analysis has shown sulphur enrichment at grain boundaries and that the risk of cavity and microfracture initiation is greatest at these locations. In order to limit the effect of sulphur enrichment, the grain size of the material should not be too large.

The canisters must be sufficiently robust to withstand the external overpressure that can arise due to the fact that the bentonite is not saturated and does not swell at the same rate throughout its entire volume. The canisters must also withstand the increased external pressure to which they are exposed when the ground outside the repository is covered and pressed downwards by a thick continental ice sheet. If the canisters meet these materials strength requirements, they will also withstand a certain deformation in the deposition holes which could be caused by bedrock movements in connection with an earthquake in the vicinity or due to creep movements in the surrounding bedrock fracture network. Materials strength requirements can be established for glacial conditions since a continental ice sheet has a limited thickness. However, requirements that can handle all possible movements in the bedrock cannot be formulated in a similar manner.

6.5 Extrapolation of creep data

Results from creep experiments are often presented in creep rupture diagrams, where the stresses at which the experiments were conducted are plotted as a function of time to rupture, see Figure 1. A loglog diagram is usually used. As shown in the diagram, the time to rupture t_R is highly dependent on the stress σ applied in the experiment. Temperature T also has a considerable impact. The following approximate relationship is obtained

$$t_R = C\sigma^{-m}e^{-BT} \qquad (1)$$

where C, m and B are positive constants. The value of m is usually between 5 and 20. Over a broader temperature range, the value for m decreases with an increase in rupture time and temperature.

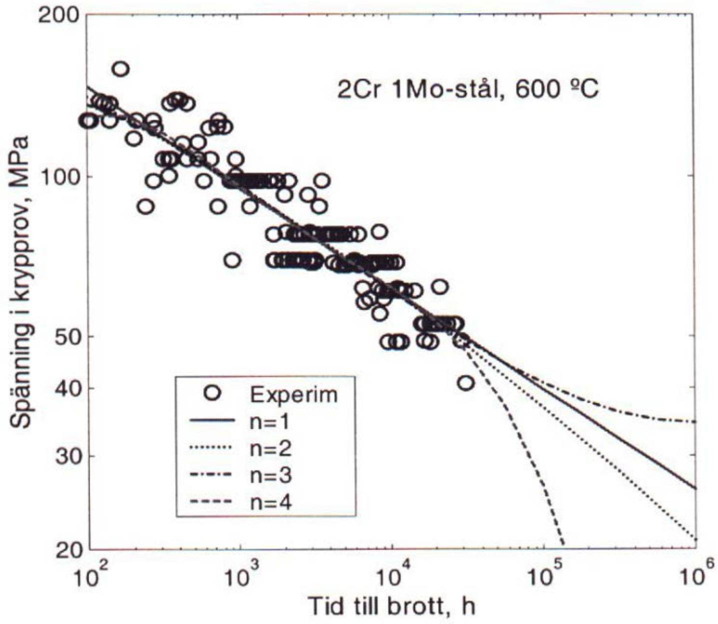

Figure 1. Stress as a function of time to rupture during creep in 2.25%Cr 1%Mo-steel at 600 °C. This description is called a creep rupture curve. Polynomials of degrees n = 1 to 4 have been adapted to the experimental data. Even if the polynomial shows a good fit with the data, they are of no value for data extrapolation involving long periods.

6.5.1 Simple Curve Fit

Figure 1 illustrates the most straightforward procedure for extrapolation, which we have called the "Off-the-cuff method" (OTCM) [6]. With this method, a polynomial is fitted to the empirical data points and the polynomial is then used to derive the extrapolated values. The creep data in the Figure applies to a high-temperature steel comprising 2.25%Cr and 1%Mo at 600 °C. The steel is commonly used for steam lines in combined heat and power plants. The data were taken from a European steel database. Smelt from several steelworks is included [6]. The creep experiments were conducted at more than ten laboratories. The spread is typical for those obtained when smelt of different origins are used. The total spread interval is about a factor of 2 in the stress.

Polynomials of n from 1 to 4 have been fitted to the experimental data in Figure 1. The four polynomials result in an equal fit to data. On the other hand, the results differ considerably for times greater than the longest test time. For $n = 3$, the curve slopes upward for long times, which is not physically possible. This curve should be completely ignored. The difference between the other curves gives an impression of the uncertainty of the results. A more precise measure of the error is obtained if the analysis is revised, but without the data points for the longest test times and if the results are then compared after an extrapolation by a factor of 3 (at $3\,t_{R\mathrm{max}}$). It is common to eliminate specimens that have a longer rupture time than $t_{R\mathrm{max}}/3$ where $t_{R\mathrm{max}}$ is the longest rupture time in the original analysis [7]. This type of procedure is called a "screening analysis" since some of the experimental data are eliminated, in this case with the longest test time. The result is shown in Figure 2.

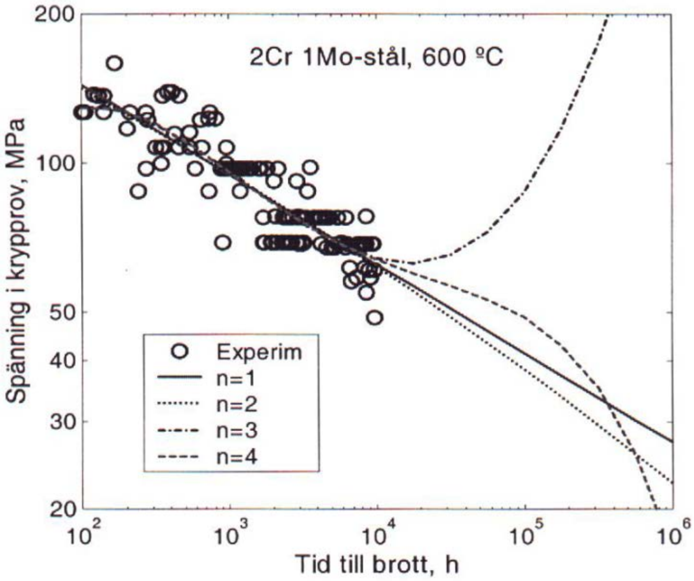

Figure 2. *The same as Figure 1, but without the rupture times that are longer than one-third of the longest time in Figure 1. An analysis without the longest experimental data is "screened". If the extrapolated results are to be considered to be acceptable, the difference between the original and the screened analysis must be insignificant, which is far from the case in Figure 1 and 2.*

In Figure 2, both of the polynomials for $n = 3$ and $n = 4$ show a physically unreasonable behaviour and these curves cannot be used. For $n = 1$ and $n = 2$, the predicted stress is about 30% higher than that for the unscreened data at 3 $t_{R\max}$ (90 000 hours), which is far too great a variation. A rule of thumb is that 10% is specified as an acceptable error.

It can also be observed that application of the OTCM results in significant errors, even in the case of extrapolation for a limited time. The risk of obtaining completely unphysical results is also significant. It is important to observe that the above example is not in any way rigged. The spread of the data is normal as has been stated above. The data are also extensive. The total testing time of the experiments for the data points in Figure 1 is no less than 110 years. An adequate adaptation of the model to data is a prerequisite for a good result in connection with extrapolation. However, this is in no way a sufficient condition in itself.

6.5.2 Use of the Time-Temperature Parameter

As early as in the fifties, one extrapolation method, based on the work of Larson and Miller [8] became popular and the procedure still continues to attract considerable attention. The method is based on a time-temperature parameter *PTT*. The most common method of extrapolating creep data is to use such a parameter. A family of rupture curves from different test temperatures is compiled using a temperature-compensated time axis. In this way, the rupture curves can be superimposed on a single master curve. The function of time and temperature used to construct the master curve is known as the time-temperature parameter. The first, which was proposed by Larson and Miller, was derived from an Arrhenius equation. Most of the parameters used nowadays have an exclusively empirical background. This is the case, for example, for the parameter proposed by Manson and Haferd [9] and upon which the ISO standard is based [1]. These empirical parameters have been developed by thoroughly analyzing the experimental data. The time-temperature-parameter proposed by Larson and Miller is as follows

$$PTT_{LM} = T(C + \log t_R) \qquad (2)$$

where T is the temperature in Kelvin. If the rupture time t_R is expressed in hours, the constant C usually has values between 10 and 25.

The use of a time-temperature parameter to combine data from different temperatures is illustrated in Figure 3, where the test stress is plotted against PTT for six temperatures from 500 to 625 °C. By assigning a suitable value to C, the experimental results can be positioned along a curve without the spread in the data being larger than at individual temperatures.

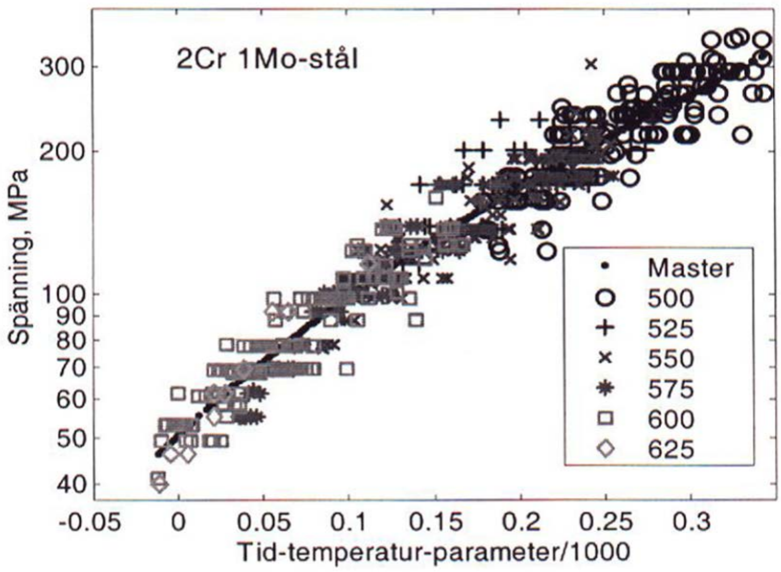

Figure 3. Stress as a function of the time-temperature-parameter in equation (3) for a 2.25%Cr 1%Mo-steel at 500 to 625 °C. This is known as a master curve since data from different test temperatures can be represented in the form of a relationship. The master curve allows stresses for long time-periods at low temperatures to be derived from test data at high temperatures.

The master curve is then used to derive the extrapolated stress values. A generalized form of *PTT* has been used in Figure 3

$$PTT = v(T) + w(T)\log t_R \qquad (3)$$

where $v(T)$ and $w(T)$ are polynomials in the absolute temperature. The selected PTT function can adapt easily to experimental data at the six temperatures, see Figure 4. Larson-Miller, Manson-Haferd and a number of the other classical time-temperature parameters are special cases of equation (3). If the screening test is applied and only test times $\leq t_{R\max}$ /3 are included, where $t_{R\max}$ is the longest rupture time, the extrapolated values at 3 $t_{R\max}$ are well within the difference of 10% in relation to the analysis when the entire experimental data are included. The stability of the selected form of *PTT* is thereby illustrated.

The most important alternative to the time-temperature parameter is the use of graphical methods. For an overview, see [11]. Based on these methods, break points, changes in the slope of individual rupture curves, can be studied. The difficulty is that the person conducting the analysis must make a number of non-trivial judgements. The method is also difficult to computerize entirely, which means that considerable manual work is involved. Time-temperature parameters are without doubt the most commonly used methods. The reason is probably due to the fact that they are easy to use and the fact that a single master curve can summarize all of the rupture data. One advantage is also that statistical processing can be easily conducted.

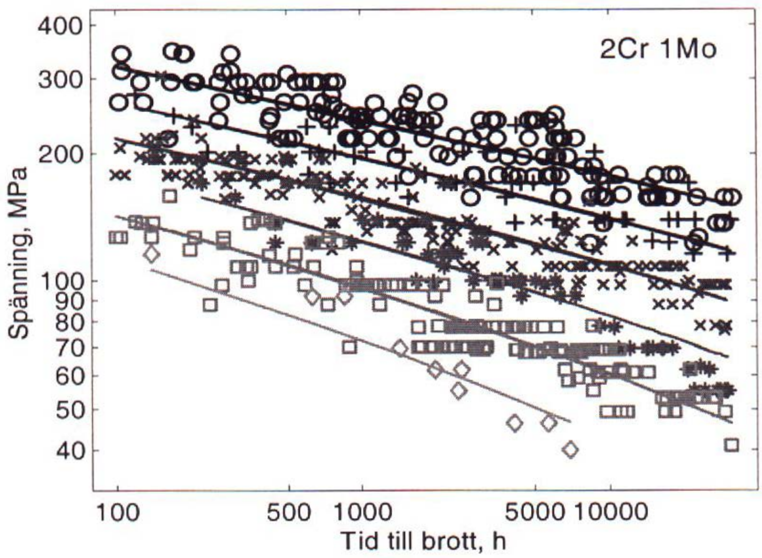

Figure 4. Stress as a function of time to rupture during creep in 2.25%Cr 1%Mo-steel at 500 to 625 °C. The experimental points are specified using the same symbols as in Figure 3. The solid lines have been derived from the master curve in Figure 3.

6.5.3 Extrapolation to Long Time-periods

The accuracy of extrapolated values increases with the quantity of available data and with times and temperatures covered. According to the applicable ISO standard, it is not recommended that extrapolation should be performed by more than a factor of 3 in time [1]. Unfortunately, this recommendation entails very strict requirements on the availability of creep data. Furthermore, the methods that are traditionally used are hardly suitable for long-term extrapolation. Modern high temperature facilities are typically designed for a lifetime of between 200,000 and 300,000 h (23–35 years). If only a factor of 3 in time is allowed for extrapolation, data up to between 70,000 to

100,000 h must be available. Unfortunately, such data are only available for a few of the most common pressure vessel steels and, when available, only for a limited number of temperatures.

Therefore, it must be possible to possible to perform the extrapolation for longer time intervals than a factor of 3. This is a deciding factor in the case of materials used in containers for spent nuclear fuel. For several of the properties under consideration, laboratory data are available for only up to about one year of testing time. It must then be possible to extrapolate these data to 200-500 years, which is the "active part" of the canister lifetime.

One method which allows for long-term extrapolation has recently been developed [3]. A short description of the principles of this procedure is presented below.

6.5.4 Lower and Upper Bound for Error

If a linear curve is fitted in a log-log diagram to creep rupture data with the least-squares method, the spread of the data results in a statistical uncertainty in the results [12], [3].

$$\log\varepsilon_{stat} = \frac{\log f\left[2\log t_{ext} - (\log t_{\max} + \log t_{\min})\right]}{0.6\sqrt{n-2}(\log t_{\max} - \log t_{\min})} \qquad (t_{ext} > t_{max}) \qquad (4)$$

where t_{min} and t_{max} is the shortest and the longest creep rupture time in the database, t_{ext} is the extrapolation time, f is a measure of spread, for example, the variance in stress in the data and n is the number of experimental data points. The uncertainty in the result ε_{stat} is given with the same measure of spread which is used for f. The error in the extrapolation can never be less than ε_{stat} which represents a lower bound.
If the fitted curve is within the uncertainty band for data $\pm$ log$f$, a corresponding upper bound can be derived. Assuming that the extrapolation follows certain principles and that the creep

rupture curve is not too non-linear, the following upper bound ε_{max} is obtained for the error [2].

$$\log \varepsilon_{\max} = \frac{\log f\left[2\log t_{ext} - (\log t_{\max} + \log t_{\min})\right]}{(\log t_{\max} - \log t_{\min})} \quad (t_{ext} > t_{max}) \qquad (5)$$

Expressions (4) and (5) have essentially the same form.

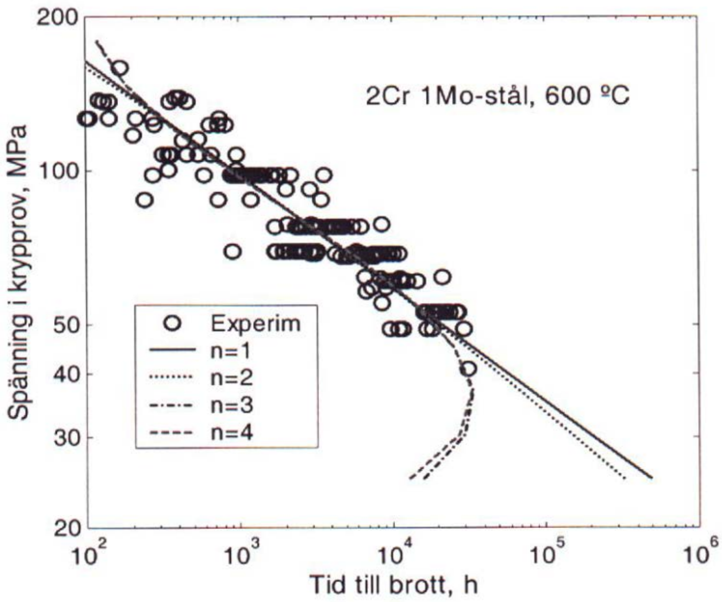

Figure 5. The same as Figure 1, but where the polynomial in log stress instead of log time has been adapted to the data. Fitting the polynomial in log stress results in a lower stability in the extrapolated data. In the figure, this is illustrated by the fact that two of the curves (for 3rd and 4th degrees) show an entirely unphysical behaviour since they reverse.

6.5.5 Stability

The method used must be stable, namely, if a limited quantity of experimental data points are added or removed from the analysis, the result should only be marginally affected. An important condition is that the fitted curve should have the correct physical form, namely, negative first and second derivatives. The stability is then verified using the criterion mentioned above. Predicted values for $t_{ext} = 3\ t_{max}$ may not be changed by more than 10% if the longest experimental times ($> t_{max}/3$) are excluded from the analysis.

6.5.6 Choice of Fitting Parameter

Traditionally, a polynomial in log stress is adapted to experimental data to represent the materials data. It has recently been shown that a smaller margin of error is obtained if the polynomial is used instead in log time [2]. This is illustrated in Figure 5.

Figure 5 is exactly the same as Figure 1, besides the fact that a polynomial in log stress instead of log time has been used, which has led to a dramatic deterioration in the fit.

6.6 Pure copper as canister material

6.6.1 Requirements on the Canister Material

It is planned to store spent nuclear fuel in Sweden in double-walled containers made of cast iron and pure copper. The containers will be deposited at a depth of about 500 m in the bedrock. The cast iron insert will be designed to bear the external hydrostatic pressure and will have a minimum wall thickness of 50 mm. The main function of the outer copper canister is to provide corrosion resistance [4].

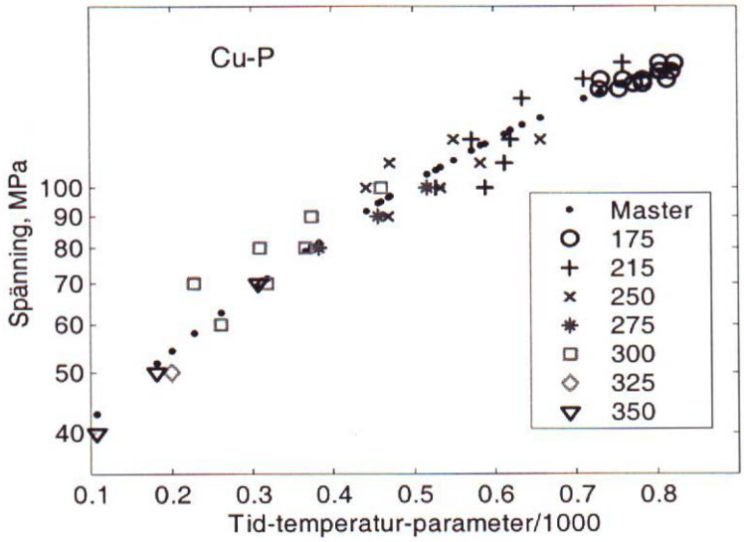

Figure 6. Stress as a function of the time-temperature parameter in equation (3) for phosphorus-alloyed copper at 175 to 350 °C. This master curve represents data from all of the test temperatures. Using the master curve, stresses for long time-periods can be derived from the test data at higher temperatures

The wall thickness of this component is 30–50 mm. The copper canister will have an outer diameter of 1 050 mm and a height of 5 000 mm. The gap between the inner and the outer container is planned to be 1–2 mm. When the canister is exposed to a slow increase in external pressure from the groundwater, the gap will be reduced as a result of copper creep.

The temperature of the canisters due to the heat from the nuclear reactions in the spent fuel is estimated at about 80–90 °C during the first 100–200 years of final disposal. The creep deformation of the copper (about 4%) is expected to occur during this time. Consequently, it is important that the pure copper should have an adequate creep ductility. Subsequently, the temperature will gradually be reduced during the first thousand years until the temperature in the surrounding rock, 15 °C, is reached.

Currently, only pure copper with about 50 ppm of phosphorus is planned to be used. Copper without phosphorus has been shown to lead to low creep ductility. The reason is that the material is sensitive to contamination from sulphur which results in pitting and cracking in the grain boundaries during creep [13]. A large grain size emphasizes the problem. Therefore, during the manufacturing of copper canisters, it must be ensured that the grain size will remain within the specified limits.

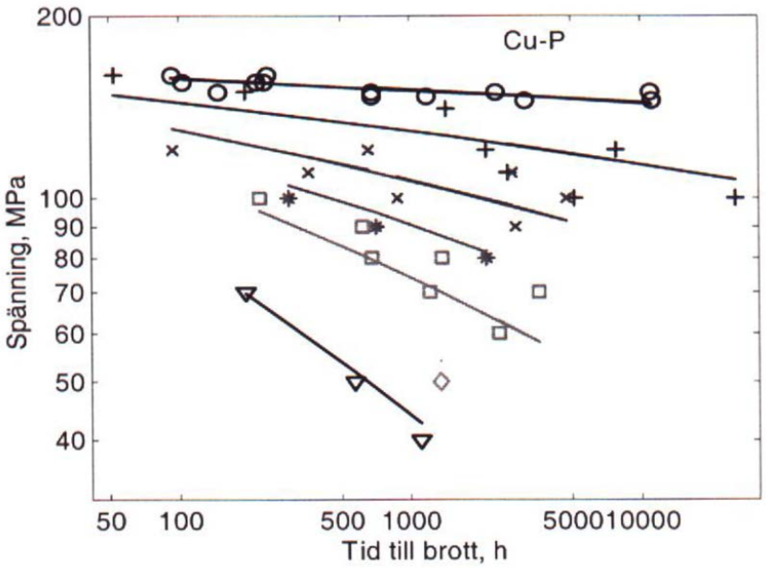

Figure 7. Stress as a function of time to rupture during creep in phosphorus-alloyed copper at 175 to 325 °C. The experimental points are specified using the same symbols as in Figure 6. The solid line model values are derived from the master curve in Figure 7.

6.7 Extrapolation for the canister material

6.7.1 Creep Rupture Strength

The method described in the previous chapter for extrapolation to long time-periods has been applied to pure copper containing phosphorus. Creep data for this type of material can be found in [14], [15], [16], [17]. Data for seven temperatures from 175 to 350 °C have been included in the analysis. Figures 6 and 7 show the experimental data together with the master curve and the predicted values. The analysis meets the criteria specified above, with respect to stability, for example.

The materials strength values for pure copper are not design basis requirements since the canister design is such that the cast iron container will absorb loads. On the other hand, the materials strength properties are necessary to predict canister behaviour in the repository. For example, it is important to know how and for how long canister deformation will occur.

6.7.2 Creep Ductility

The mechanical property that is most decisive for canister integrity is creep ductility. As mentioned above, the copper material will be deformed by up to 4%. The area around the lid weld, where tensile strains will occur is particularly critical. Therefore, the creep ductility of the material must be at least 4% to avoid cracking. However, the stress state of the canister must also be taken into account. In the case of conventional creep testing, the stress state is uniaxial, namely, stresses only occur longitudinally in the test rod. In the copper canister, more complex multi-axial stress states will occur. Under these multi-axial stresses, the creep ductility will be lower, at least by a factor of 3 [18]. The exact factor is not known. The requirement on uniaxial creep ductility will therefore be at least 15% with a certain safety factor.

Creep ductility has been determined for phosphorus-alloyed copper for temperatures between 175 and 300 °C in the same experiments as for creep rupture strength. The results are shown as a function of the grain size in Figure 8. The creep ductility is approximately independent of grain sizes up to 0.4 mm. However, it decreases for higher grain sizes. One consequence of this is that the grain size should be less than 0.4 mm, with a certain margin. Furthermore, since non-destructive testing of a small grain size is easier, SKB has established the requirement that the grain size should not exceed 0.25 mm.

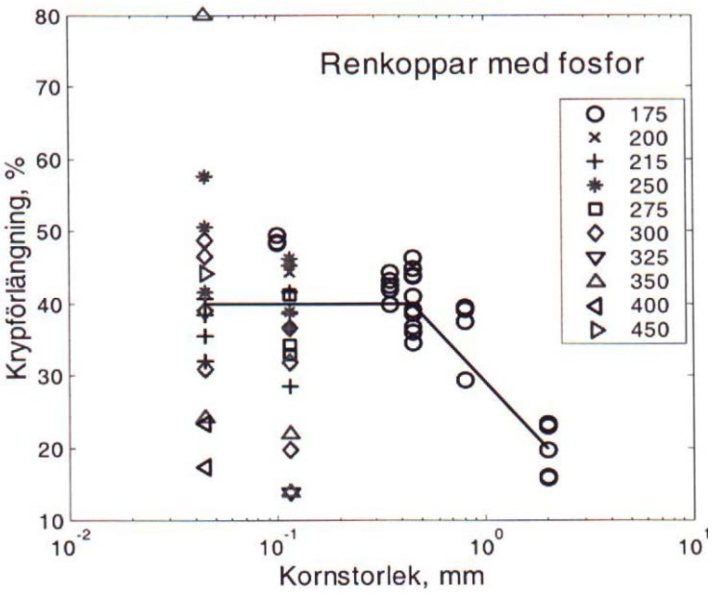

Figure 8. *Creep ductility expressed as creep elongation as a function of grain size for phosphorus-copper alloy at 175 to 450 °C.*

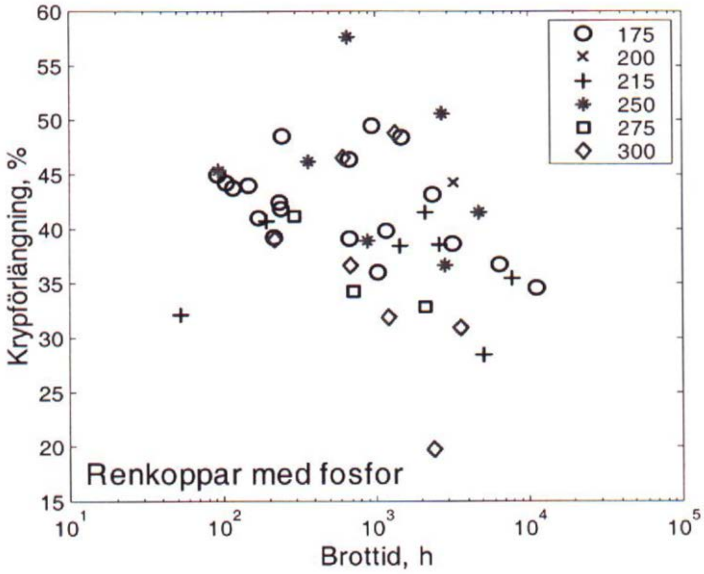

Figure 9. *Creep ductility expressed as creep elongation as a function of the rupture time for phosphorus-copper alloy at 175 to 300 °C. The creep ductility generally decreases with an increase in temperature.*

Creep ductility also decreases with an increase in rupture time, see Figure 9. A certain temperature dependence can also be observed. With an increase in temperature, the ductility typically decreases, even if there is a considerable spread.

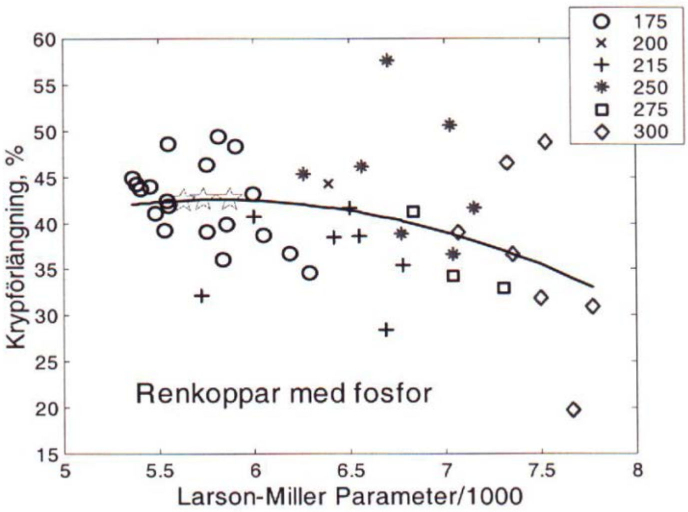

Figure 10. Creep ductility expressed as creep extension as a function of the rupture time for phosphorus-pure copper alloy at 175 to 300 °C. The creep ductility generally decreases with an increase in temperature.

To be able to extrapolate ductility to long periods of time, the assumption is made that the decrease in ductility after long periods of time can be represented by values at higher temperatures with shorter fracture times. With this assumption, a time-temperature parameter can be used. The result is shown in Figure 10. The Larson-Miller parameter in equation (2) has been used. The same constants have been used as for the extrapolation of creep rupture strength. Ductility decreases slowly with an increasing time-temperature parameter. As mentioned above, the copper canister will be exposed to creep for 100–200 years at a temperature of about 80 °C. These conditions are marked in the figure as five-pointed stars. To cover a wide range of times, 500 years at 80 °C is also included.

These three points are located close to each other so that the exact assumption regarding time is not critical. As shown in Figure 10, the conditions in the repository result in a ductility of 42%. If the uncertainty band is also taken into account, the creep ductility is expected to be between 35 and 50 %. This means that the requirements have therefore been met.

If it is not possible to represent the ductility data using a time-temperature parameter, the lowest value within the temperature interval that can be considered to be relevant would have to be selected. This is a very conservative approach. In this case, the creep ductility would be 20%.

Further studies on the creep ductility of copper are essential. It is particularly important for properties of the weld joint to be investigated as well as the impact of multi-axial loads.

6.7.3 Uniform Corrosion

Extrapolation of corrosion data will only be briefly dealt with and limited to the two most important corrosion types – uniform (general) corrosion and localized corrosion. There is relatively little information in the scientific literature on corrosion after long time periods and on how data from laboratory experiments can be extrapolated to these time periods. Most of the information seems to concern implants in the human body which have to function for many years. Unfortunately, this information is of limited value for the corrosion of canister material. The handling of data for uniform corrosion and localized corrosion occurs in different ways and will therefore be handled separately.

Uniform corrosion occurs though an even corrosive attack on the material surface. In the case of the pure copper in the canister, this type of corrosion will continue until the oxygen in the near field has been consumed, which is expected to take a few hundred years [19], [20]. Due to the thermodynamic stability of the copper in groundwater under reducing conditions, continued corrosion only occurs through dissolved sulphides. Since even the access to these are limited, the thickness of the canister can easily be chosen so that the corrosion does not affect the integrity of the canister. In this case, the extrapolation is based on the fact that only oxygen and sulphides can result in corrosion due to the chemical stability of the material in groundwater and the fact that the maximum mass transport of these dissolved substances can be relatively simply estimated.

6.7.4 Localized Corrosion

Localized corrosion involves local corrosion attack due to an enrichment of aggressive substances. If pitting is too deep, there will be a risk of penetration. Even if penetration does not automatically lead to radionuclides leaking into the surrounding rock, the safety assessment will be essentially facilitated if penetration is avoided. Pure copper is characterized by the fact that corrosion pitting is not deeper than a factor times the uniform corrosion [19]. This quantity is known as the localized corrosion factor and is typically less than 5. If uniform corrosion is limited, the results can be extrapolated to long time-periods if a conservative localized corrosion factor is used. Localized corrosion assumes the presence of oxygen in the groundwater which, according to the above, only exists for a few hundred years at most. Localized corrosion attack will therefore gradually be reduced in the repository.

In the same way as for uniform corrosion, extrapolation is based on a maximum mass transport of oxygen. The uncertainty factor is the size of the localized corrosion factor for long time-periods. However, archaeological objects also show moderate localized corrosion factors, which further reinforces the view that laboratory data can be used.

6.8 Discussion and conclusions

The use of materials data for long time-periods and, especially, for the time-periods involved in a repository for spent nuclear fuel, requires the application of systematic methods for extrapolation. One condition for extrapolation to be possible, is that some form of accelerated testing has been conducted. The testing may be conducted at an increased temperature or stress, or an increased concentration of aggressive substances, etc. Furthermore, the aim must be to show that the damage mechanisms that are expected in the repository are the same as those in the accelerated testing. Testing must also be conducted until the state of the material corresponds to the state resulting from the repository conditions.

Three different approaches to extrapolation have been described above. The first procedure is based on the use of a transformation parameter (time-temperature parameter) and the use of laboratory data from a higher temperature in the calculation of long-term properties at a higher temperature. This method was used for creep rupture strength and creep ductility.

In accordance with the second procedure, the result is based on the lowest or highest value observed through relevant laboratory experiments, depending on which is most unfavourable. The laboratory experiments must, in some accelerated sense, be said to represent the conditions in the repository. This approach is used for creep ductility and localized corrosion.

The third procedure is based on the limitation of the extent of the damage mechanism by a process, such as mass transport. By estimating the maximum transport, an upper limit can be obtained for the damage mechanism. Uniform corrosion and localized corrosion (partially) can be analyzed in this way.

The three above-mentioned approaches can normally not be substituted for each other. Where the methods can be substituted, a transformation parameter, such as a time-temperature parameter provides the most accurate estimate.

6.9 References

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7 Calculation models for the transfer of radioactive substances from a deep repository for spent nuclear fuel to the biosphere[1]

7.1 Introduction

This chapter summarizes the state of knowledge about models which describe how radioactive substances from a geological repository can be transferred through ecosystems to flora, fauna and mankind. Radionuclide migration to the biosphere and to man primarily occurs via groundwater flows passing through the repository. For such migration to occur, the integrity of the repository barrier system must be breached, for example, due to manufacturing defects in the copper canisters containing the spent nuclear fuel or as a result of long-term corrosion in the canister or due to other canister defects. The groundwater is connected to various water systems such as wells, marshes and wetlands, lakes, water courses as well as coastal and sea waters. The groundwater can also contaminate cultivated land via groundwater transport to the cultivation zone and via irrigation. The dilution volumes in the various water systems of the biosphere largely determine the consequences from leakage from a deep repository for spent nuclear fuel in the form of radiation doses to man and animals. Another exposure pathway from a deep repository to man is via bottom sediment in seas and lakes

[1] This chapter was written by Dr. Christopher Rääf, Department of Radiation Physics, Malmö, Lund University, Malmö University Hospital and Prof. Sören Mattsson (member of KASAM).

which, after future land rise, may dry up and later be cultivated to produce foodstuffs.

7.2 Mathematical description of the biosphere for the safety assessment of a deep repository

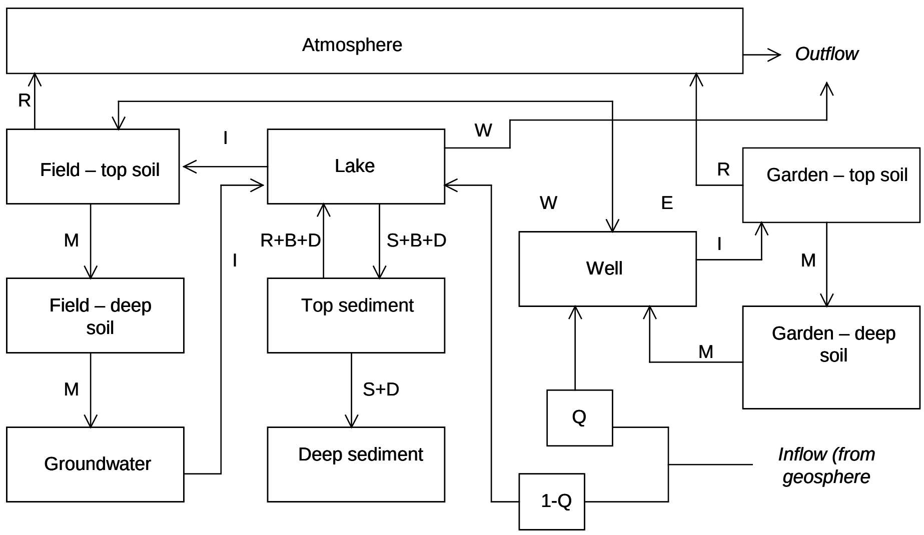
Biosphere models within environmental radiology and radiation protection are used with the aim of describing radionuclide transport through different types of ecosystems in order to estimate contamination levels and radiation doses to man, fauna and flora. Safety assessments for the final disposal of high-level spent nuclear fuel must be based on theoretical models and calculations since the leakage of radioactive substances from these facilities is not expected to occur until far into the future and far beyond the possibility of our generation or civilization to conduct measurements. The most common method is to mathematically describe the biosphere using a compartment model where each compartment or reservoir represents a unified system, such as water in a lake, a certain soil or sediment layer. Large water systems (watercourses, lakes, coastal waters and wells) are usually considered as mathematical compartments when describing an area where a deep repository could be theoretically located. When developing the models for leakage of radioactive material from the deep repository, it is assumed that the radionuclides will be transported via the groundwater flow up to the water system in the biosphere and then transferred to plants, animals and man. In schematic descriptions, transport processes are represented in the form of arrows between the different compartments. Essential parameters are the size of the compartments (such as the water volume expressed in m^3 in a lake) and the radionuclide transfer rate between the systems represented by the compartments.

Most of the compartment models are based on *first order kinetics.* This means that it is assumed that the transport rate or the radionuclide flow from one compartment to another is proportional to the quantity of the substance in the first compartment. For example, it is assumed that the sedimentation of the radionuclide ^{135}Cs, which is commonly found in spent nuclear fuel, in a typical watercourse is proportional to the concentration of this substance in the water. In certain compartment models, the possibility that the transfer rate between different compartments can vary in time is taken into account. The models used are then referred to as *dynamic (varying in time).* A more detailed review of the mathematical description of the models is provided in Appendix 1 of this chapter.

Another important characteristic of these compartment models is that it is assumed that a *complete and immediate* mixing of the radioactive substance occurs when it reaches a compartment, such as a watercourse or a well. Many models also use seasonal or average annual values of water levels, groundwater flows, irrigation, precipitation etc. Such an approximation may be very crude and not useful for biosphere models that describe radionuclide transport in the event of acute leakage to the biosphere. However, the case for radionuclide leakage from a geological deep repository is different since the leakages occur over a very long period of time (tens of thousands of years). Annual variations are therefore not as significant for the consequences in a long-term perspective.

An example of a basic model for the description of radionuclide transport is provided in Figure 1. However, the values for the transfer constants (the *parameter values* of the model) are specific for each individual element. The behaviour of these substances varies depending on their chemical and physical properties. One example is ^{237}Np, which has a high tendency to adhere onto particles along the transport path through the groundwater via soil layers and bottom sediment. This property delays the radiation exposure of the population. Another example is ^{129}I, which often occurs in a volatile form and has a low tendency to sorb onto particles. This property allows the substance to reach the biosphere and expose the population more rapidly. The model may be *general* or *site-specific* to varying degrees. Most often, biosphere model developers use a general basic model. They have then adapted the model to specific local conditions, using geological, physical and chemical properties in the ecosystem at a specific site (which does not necessarily have to be the actual repository site). The model has thereby become more site-specific (see also History – KBS-3).

Figure 1
Example of a compartment model for radionuclide turnover in the biosphere (BIOPATH – Studsvik, Sweden). The structure of the model describes radionuclide transport from ground waters into lake water and wells.



The following processes are included in the compartment model:
Q = the fraction of radionuclides in the geosphere groundwater that flows into the well
(1-Q) = the remaining fraction that flows to the lake water
W = waterborne radionuclide transport
S = mass radionuclide transport (see "Facts")
D = radionuclide diffusion (see "Facts")
B = bioturbation (see "Facts")
E = secretion of radionuclides by contaminated cattle
I = irrigation by contaminated water
R = radionuclide resuspension (see "Facts")
M = migration (see "Facts")

Most often, the ecosystems are highly complex and there are considerable variations within a small geographical area as well as over relatively short time-periods. For example, it is difficult to determine a representative value for volume and water turnover in a well during the course of a year. The same applies to the water volume in a watercourse, annual precipitation in a long valley, the fish population in a lake etc. The problem is assigning representative values to all of the transfer parameters describing different processes in the ecosystem. The biosphere model can be constructed using parameter values, each of which only has a single value and which are, as far as possible, representative of the ecosystems to be described. This type of model is a *deterministic* model (see "Facts"). A *stochastic model* can also be used (see "Facts"), where the parameter values are described by probability distributions characterized by a median value and a given statistical distribution (see Appendix 1).

These biosphere models are necessary in order to perform an overall safety assessment of the repository. In parallel with the biosphere models, models of radionuclide transport in the geosphere, of radionuclide interaction with rock pores, with particles in the groundwater and with gas bubbles and colloids are also developed. A particular problem is merging the geosphere model with the biosphere model used in the interface between deep groundwater and near surface groundwater. In general, a retardation on the order of 100 to 1000 years of the radionuclide transport can be expected in the actual interface before a balance occurs between the radionuclide inflow from deep groundwater and the outflow of these substances via groundwater transport to the biosphere (1).

The solution to the final disposal of spent nuclear fuel discussed by the nuclear industry involves depositing the material deep underground in a stable geological formation. In turn, the repository must have a number of barrier systems, such as the canister and surrounding clay buffer (of natural bentonite) and must be located in an area with a relatively low groundwater flow. Chapter 8 of this report provides a description of different countries' plans for spent nuclear fuel disposal.

A selection of biosphere models used internationally in order to model leakage from a deep repository as well as from an acute leakage from a nuclear installation is provided below (Table 1).

Table 1

Overview of biosphere models being used or which have been used for consequence analysis in connection with radioactive releases. Descriptions of some of the computer codes below are provided by the OECD's Nuclear Energy Agency (NEA) [(2)].

Model	Origin - User	Application
AIRDOSE/EPA	USA, DOE - Department of Energy[(3)]	Airborne activity
BIOPATH (BIO42 – package)*	Sweden, SKB[(4)]	Underground leakage (BIO42) + Atmospheric fallout (BIOPATH)
BIOSPHERE-CBS*	Belgium, SCK.CEN[(5)]	Underground leakage
BIOTRAC-CBS*	Canada, AECL - Atomic Energy of Canada[(6)]	Underground leakage
CRRIS	International, OECD, NEA – Nuclear Agency[(2)]	Atmospheric fallout
DETRA	Finland, VTT[(7)]	Atmospheric fallout + Underground leakage
ECOSYS	Germany	Atmospheric releases
MELODIE*	France, ANDRA[(8)] and ISPN[(9)]	ABRICOT describes biosphere – included in the MELODIE code package
NRPB MiniBIOS-CSB*	UK, NRPB[(10)]	Underground leakage
SACO + AMBER-CSB*	Spain, CIEMAT[(11)]	Underground leakage (Deterministic and stochastic model)
SS57-ORNL (NUTRAN - package)	USA, ORNL - Oak Ridge National Laboratory[(12)]	Atmospheric releases + Underground leakage
TERFOC	Japan, Tokaimaru Atomic Research Institute	Atmospheric releases

*These models are included in a comparative study from the mid-nineties – the BIOMOVS study, Phase II. A list of contacts and a detailed description of these models is included in BIOMOVS II[(13)].

Almost all of these models are compartment models with a varying number of compartments and a varying share of dynamic compartments. A separate set of parameters must be determined for each radionuclide due to the different properties of the different elements. Many biosphere models used for consequence analysis of leakage from a deep repository have similar characteristics to models of acute atmospheric releases in the short term, for example, the transfer from pastureland via cattle to man. However, the radionuclides that are relevant to a deep repository and which may potentially migrate to the biosphere, have such long physical half-times that models for acute releases are not always useful. The Canadian biosphere model (BIOTRAC-CBS) is entirely based on a stochastic description of all processes, whereas such calculations can be conducted with other models, although they are primarily intended for deterministic calculations.

7.2.1 Scenarios – accident types, dispersion processes and climate periods

The model calculations are based on a base scenario in order to be able to compare the consequences for different accident scenarios (Table 2). The base scenario most often assumes a slow deterioration in canister and barrier system performance. This means that after some ten thousand to hundred thousand years the groundwater starts to leach through the spent nuclear fuel. Such a base scenario can then be combined with a scenario where the site is covered with a continental ice-sheet for several ten thousand years, which affects the direction and rate of the groundwater flows and, particularly the biosphere. This type of scenario is now commonplace in the safety assessments in Sweden, Finland and Great Britain where major climate changes can be expected to occur with a certain regularity during the period in question. However, in order to get a continuous radionuclide leakage, as described above, to occur over a period of 10,000 years, at least one of the thousands of fuel canisters in the repository must have a manufacturing defect.

One essential simplification that is often made in these models is that the biosphere is assumed to remain unchanged during a long period of time (about 10,000 years in the Swedish calculations). Bearing in mind the major climate changes that occur, even over periods of time that are very brief from the geological standpoint, the biosphere models should take these variations into account in the calculation. In practice, it has been found to be easier to instead assume a constant biosphere under the stated calculation period and to compare the calculated concentration and dose values with values obtained from other biosphere models that represent climate types that can probably exist in the area, depending on the phase in the climate change that is of interest. In Great Britain, a comparison has been made of a biosphere model with a Nordic climate and a model with a typical Mediterranean climate. These have then been separated as two different climate scenarios. It seems that too wide ranges in the values for the groundwater flow, non-glacial periods, growth seasons etc. exist to be incorporated into a dynamic or probabilistic compartment model in a meaningful way.

Table 2
Examples of different scenarios that can be combined in a safety assessment for the repository. Different source terms, flow and climate scenarios can be combined into a specific scenario.

Source term (repository state)	Description /characteristics of flow parameters*	Climate scenario
Base scenario (slow canister degradation)	Reasonable	Unchanged biosphere
	Pessimistic	Changed biosphere
Initial canister defect	Reasonable	Unchanged biosphere
	Pessimistic	Changed biosphere
Accidental intrusion	Reasonable	Unchanged biosphere
	Pessimistic	Changed biosphere
Earthquake, volcano eruption etc.	Reasonable	Unchanged biosphere
	Pessimistic	Changed biosphere

*These parameters determine the radionuclide transport rate through the bedrock up to the surface and the extent to which the radionuclides reach man in the biosphere.

7.3 Model development

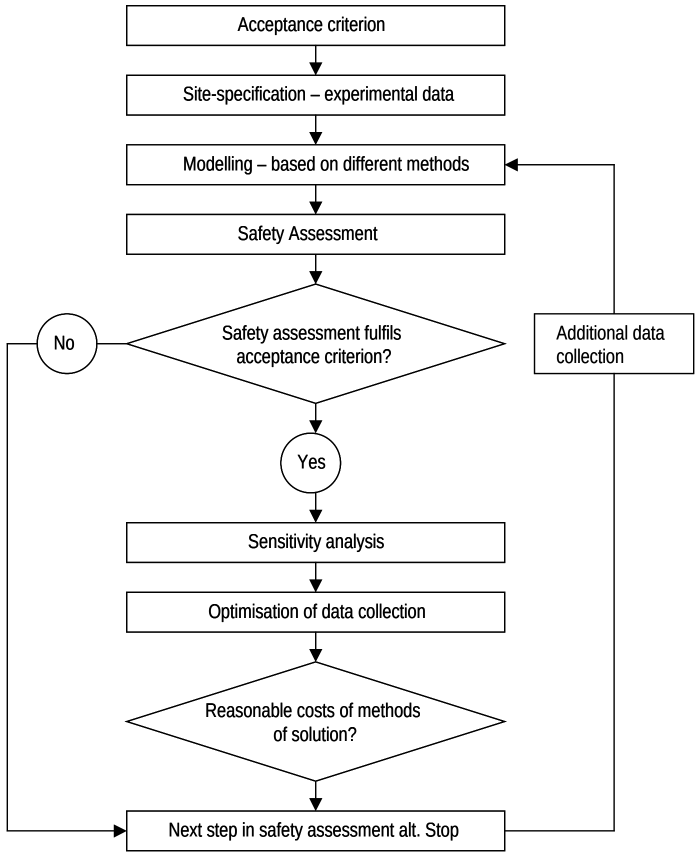
7.3.1 Strategy for model development

At first, an overall description of biosphere models for the consequence analysis and safety assessment of a deep repository for spent nuclear fuel is made, see Figure 2, which is taken from the US decision support system, SEDSS(14). Here, the strategy for model construction that can provide decision support and consequence analysis for a final disposal method is shown. This strategy is similar to that provided by the Finnish, VTT(7,15) and the British Nirex's(16) description of the development of the model. The starting point in the flow chart is an acceptance criterion in the form of, for example, a highest tolerable risk or effect to an individual (individual dose within a critical group – see Section 7.3.3) or to a whole population (collective dose – see "Facts") that can result from a radioactive release from a deep repository.

The next phase of model development is the collection of site-specific data that can serve as a basis for the next stage – a conceptual model of how the transport of radioactive substances occurs through the ecosystem. After an initial consequence analysis, a decision must be made concerning whether the model has to be further developed or whether the result obtained from the consequence analysis is sufficiently reliable to provide a basis for decision-making.

Both Nirex(16) and SKB show, in their descriptions of biosphere model development, a common pattern in accordance with the strategy described in Figure 2. The starting point is simple static compartment models where approximate simplifications are made based on the principle of not attempting to underestimate or overlook an important pathway – known as a conservative calculation. This is an initial step in the development, in order to arrive at an approximate consequence analysis of leakage. The models have then been refined by including dynamic processes. Another important line of development is that both in the case of Nirex's(16) and SKB, a distinction was made between reasonable and conservative assumptions. These assumptions concern the values that are assigned to the different parameters that describe the transport process. At a relatively early stage, Studsvik AB began, on behalf of SKB, to develop a model for the interface between the geosphere and the biosphere, since the retarding effect of the groundwater catchment on radionuclide transport to the biosphere had not previously been taken into account(38). At a later stage, SKB extensively refined its models as it obtained an adequate database for the quantification of many of its model parameters. For example, the erosion of agricultural land and transport processes in the soil have been taken into account. On the other hand, a model for a forest ecosystem has not yet been developed since experimental data are not available for radionuclide dispersion and uptake by plants and animals in a forest environment (see also Section 7.10).

Figure 2
An example of a possible safety assessment strategy. The figure is taken from Sandia Environmental Decision Support System[14]. A further feedback in addition to cost-related aspects exists if the safety criterion is made more stringent due to pressure from public opinion etc. This means that the model developer at the first stage may have to review the model and collect additional field data.



7.3.2 Collection of field data

Both in Sweden and in Great Britain, general models have been combined with site-specific data for the biosphere taken from investigations at a specific pre-investigation site or from other sites in the country in order to represent, as adequately as possible, the local conditions at possible deep repository sites. In Sweden, work was based on the BIODOSE model with parameter values for conditions in the USA. A number of investigations were conducted at study sites (including Äspö in Småland, Finnsjön in Northern Uppland, Gideå in Västernorrland). The investigations included bedrock sampling and the determination of radionuclide turnover in local water ecosystems. As the extensive field studies at the different sites have started, the original values for the USA were replaced by local, site-specific data. Where data was unavailable, the IAEA's guidance values were used(17).

One problem that was previously mentioned was that most of the experimental data on radionuclide transfer from soil to plants etc. are based on studies of atmospheric fallout where the radioactive substances come from the air instead of from the groundwater, which would be the case for a leaking deep repository. Alternative exposure pathways, such as irrigation from wells and water sources, are also possible.

Nirex(16) points out that many values for transport processes from soil to plants are based on lysimeter data, where the quantity of water and other substances taken up by plants under varying climate conditions has been studied in soil columns. It is important to remember that the conditions may be different for plants in arable land where groundwater transport freely occurs.

7.3.3 Criteria for consequences

The key parameters in the consequence analysis comprise the effective dose of radiation to an individual within the critical group, defined in accordance with the International Commission on Radiological Protection (ICRP)[37]. However, SKB has, in its most recent safety assessment, SR 97[18], used the radiation dose accumulated over 50 years to the most exposed individual (including assumptions concerning lifestyle) to describe the consequence of a release to an individual. However, Nirex[16] and VTT[7] use the annual effective dose to the most exposed individual. Another measure that the Finnish VTT[7] has used is the collective dose on the regional and global level. SKB has also calculated collective doses to different groups in the first phases of the safety assessment, KBS1-3.

The effective dose can be converted into a *risk.* In connection with low levels of ionizing radiation, only the concept of late effects is used. These effects can occur within a typical time interval of 5 to 50 years. Late effects primarily refer to cancer induced by radiation. The relationship between the risk receiving lethal or non-lethal radiation-induced cancer or/and genetic effects, and the effective radiation dose is largely based on extensive investigations of survivals from the atom bomb attacks in Japan in 1945. Mathematically, this relationship can be described in simplified form as follows:

Risk = the sum of probable radiation doses (in sievert) that an individual can receive x 0.073 (1.)

The figure of 0.073 can be interpreted in the following way: For low doses and low dose rates an individual runs a 5% per Sv risk to die from radiation-induced cancer. Estimations of the detriment due to non-fatal cancer and severe hereditary effects give 1.0% and 1.3% per Sv respectively. Altogether this means

7.3% per Sv (which explains the figure 0.073 for risk per Sv in equation 1). In comparison, it can be mentioned that a Swede, on average, receives an annual effective dose of 1 mSv as a result of natural background radiation (in the form of cosmic radiation, radiation from the ground and building material as well as potassium in the body; the contribution from daughter nuclides of radon, mainly to the lungs and respiratory passages is not included in this figure), which thus corresponds to an annual risk of around 0.01% or 10^{-4}.

In its regulatory code from 1998 (FS 1998:1[19]), the Swedish Radiation Protection Institute (SSI) stated that the annual risk for late effects after the closure of the repository must be no higher than 10^{-6} for a representative individual in the group exposed to the highest risk, also known as the *critical group.* The acceptance level is therefore more than 100 times lower than the contribution from the above-mentioned natural radiation components. In the commentary to the regulations, SSI states (as does the ICRP in its documentation on the critical group) that a tolerance interval must be defined around the acceptance value, where the ICRP proposes the factor of 10 or 3 for the whole interval, depending on the conditions. SSI has specified a tolerance level of a factor of 100 as a reasonable value for a more regional interval, that is from a risk of 10^{-5} to 10^{-7}, which means that an individual in the critical group could run the risk of 10^{-5} (1 per 100,000) per year. In SR 97, SKB correctly interpreted SSI FS 1998:1, but did not present any reasons to justify (in cases where 10^{-5} was used) that this value is still within SSI's acceptance criterion.

According to the ICRP's definition, the critical group[37] is a relatively homogeneous group with respect to age, diet and lifestyle and which is representative for the most exposed individuals. These individuals may be a hypothetical group of people who are assumed to live under conditions that are most susceptible to contamination by radionuclides. Such a lifestyle includes living of locally produced foodstuffs, cultivating crops without using artificial fertilizers and having game, berries, mushrooms and fresh water fish as an important part of their diet. The fresh water and water for irrigation are assumed to come from local wells or watercourses. Both SKB and Nirex[16] have included these exposure pathways to a possible human settlements near to the repository. Nirex[16] points out that the definition of a critical group must be based on general observations of societal structures existing today (but with the reservation that either a Mediterranean or Nordic climate could occur in the region) and not on extreme conditions.

The environment in general is also dealt with in SSI's 1998:1 regulatory code. The ICRP is currently reviewing its existing recommendations, including those within the environmental area, with the aim of presenting new recommendations at some point between 2004 and 2005. In SR 97, SKB did not deal with any environmental issues. However, in its material for SAFE (the new safety report for SFR, the repository for low and intermediate-level waste beneath the seabed outside Forsmark), SKB presents new methods with the aim of taking environmental considerations into account in general. SSI also leads a project within the EU's fifth framework programme under the acronym, FASSET – Framework for ASSessment of Environmental impact[20]. The aim is to develop a system for handling the consequences in the entire environment. SKB is another Swedish participant in this programme (see also Section 7.10.3).

7.3.4 Time frame for risk estimates

Nirex[16] emphasizes that estimates of consequences beyond 10,000 years are of limited value and the British Radiation Protection Board (NRPB) states that predictions extending further than 1,000,000 into the future are not meaningful. In SKB's case, the conditions are somewhat different, since the probability of a glaciation of the earth's surface above the repository sites is significantly greater than in the British case. Therefore, climate scenarios involving recurring glaciations, have also been included in one of the scenarios used in the safety assessment and the time frame for meaningful risk estimates has been increased to at least 100,000 years. In SKB's view, glaciations will lead to less severe consequences compared with a scenario without glaciations. A glaciation means that the site will be uninhabited for long periods of time and prevent radionuclide transport through aquatic ecosystems.

7.4 History – Sweden

7.4.1 KBS-1 and KBS-2 (1977)

In April 1977, the Riksdag (Swedish parliament) passed a resolution that the Swedish nuclear power producers should present a completely safe solution to the deep disposal problem for spent nuclear fuel, before any new licences could be issued for new reactor construction. Two safety assessments were initiated the same year (KBS-1, 1977 and KBS-2, 1978). The fact that no reprocessing and re-use of Swedish nuclear fuel was allowed, was also taken into account in the KBS-2 assessment. This changed the conditions for the source term for the repository in terms of size and composition. The long-lived high-level nuclear fuel would be deposited underground in stable bedrock at a depth of about 500 m.

In the KBS-1 assessment, three different base scenarios were included: *i.)* radionuclide leakage from the groundwater to a well in a long valley, *ii.)* leakage to a lake and *iii.)* radionuclide leakage to the Baltic Sea. Like other consequence analyses of leakage from deep repositories, SKB was aware that the decisive factor for the radiation dose to the exposed population was the dilution volumes of the different water systems to which the groundwater was connected. The estimated dose values concerned the effective dose to the most exposed individuals (critical group), living in an area near to wells, lakes or the coast of the Baltic Sea. A compartment model was used for the calculation, comprising 17 compartments[(21)], and the nuclide-specific dose values for 16 radionuclides were estimated using the BIOPATH code[(22)] developed in the mid-seventies. The results of the analyses indicated that the most important exposure pathway was drinking water from the contaminated well. The next was through the consumption of fish. Radionuclide transport in the interface between the geosphere and biosphere was not discussed at this time. One conclusion was that the maximum individual dose would not exceed 4 mSv over a 30-year period and that releases from the repository should, thereby, not lead to any significant radiation doses to man.

Apart from the fact that the source term in KBS-2 comprised spent nuclear fuel, the difference between KBS-1 and KBS-2 was that the model was expanded to also include the global turnover of carbon and iodine as well as the calculation of dose values for 22 radionuclides.

The central scenario was that the release would start after 100,000 years and continue for 50,000 years. For the first time, the input parameters for the calculation models were also divided into two categories: *i.)* reasonable values and *ii.)* conservative values for groundwater flow rates and retardation in the bedrock which is assumed to be porous. These two categories of parameters were combined with the three different base scenarios that were used in KBS-1 and six different scenarios were thereby available as a basis for calculation and comparison. In general, the conclusions drawn were that drinking water via wells was the most important exposure pathway for radionuclides to mankind as well as the fact that fish consumption was the most important pathway for the abundant radionuclide ^{135}Cs. In the “reasonable” base scenario, the annual dose to the critical group was estimated not to exceed 0.11 mSv, while the doserate to the critical group via the well water was calculated at 0.70 mSv/year. The cumulative collective dose over 500 years was estimated to be no greater than 85 manSv. In the KBS-2 assessment, a more sophisticated analysis was also conducted of the impact of the decay chains on the dose values, where the ^{234}U – ^{230}Th – ^{226}Ra chain was considered to be of particular interest.

7.4.2 KBS-3 (1983)

In 1983, a further safety assessment, KBS-3[(23)], was conducted where the sediment in the sea ecosystem was now represented by an additional compartment. As before, the biosphere was divided into a local, a regional and a global zone for the calculation of collective doses. A description of the equilibrium relationship between the concentration of the radionuclide dissolved in the solid material (Bq/kg) and in the water solution (Bq/l), K_d was also used. The same year, Studsvik presented[(24)] a method to determine the uncertainties in the parameter values determining the transfer and concentration in the BIOPATH model (see Section 7.7). The retardation in the interface between the groundwater and biosphere was a further uncertainty that the report did not deal with, but which was evaluated in the late eighties[(1)] in connection with an international model validation study. In the case of ^{129}I, it was estimated that it would take between 100 and 1,000 years before equilibrium for the concentration in the root-zone was reached as a result of a continuous release into the groundwater.

The British National Radiological Protection Board (NRPB) was given the task of reviewing SKB's safety assessment, KBS-3. On the whole, SKB was commended for the quality of its work and the NRPB considered that SKB's description of the safety assessment was a model of coherence that other nuclear power-producing countries should emulate. On the other hand, SKB was criticized for *i.)* not taking into consideration a dynamic biosphere *ii.)* not including a scenario with human intrusion into the deep repository and *iii.)* not performing a sensitivity analysis for the parameter values of the biosphere models. SSI adopted a more distanced approach to SKB's biosphere work than the NRPB.

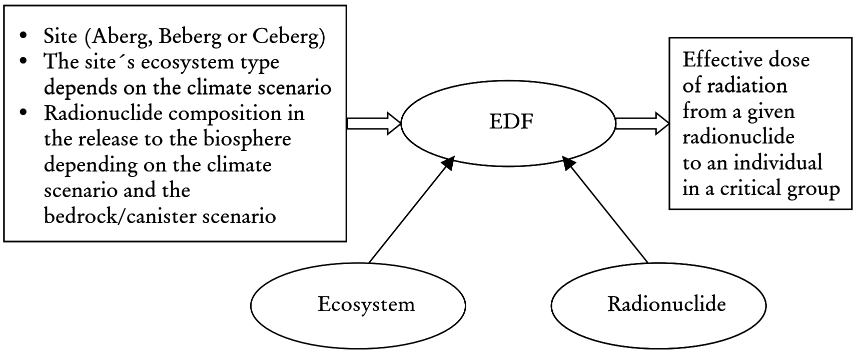
7.4.3 SKB-91 (1991)

The main purpose of the biosphere model development in this stage of the safety assessment was to determine the dose factors (=effective dose per unit of radioactivity released to the biosphere). Work was conducted on three different scenarios for radionuclide transport from the geosphere via the groundwater to the biosphere: *i.)* groundwater outflow where 99% goes into a lake and 1% goes into a well (see Figure 1), *ii.)* 100% of all of the groundwater flow containing radionuclildes goes into the well, *iii.)* the groundwater goes into the Baltic Sea. Several important assumptions were made, for example, constant living conditions for humans and no retardation between the groundwater transport in the interface between the geosphere and biosphere. The concept of the ecosystem-specific dose factor (EDF) was introduced. The EDF is the effective dose that an individual in a critical group receives over a period of 50 years from a given radionuclide and is calculated on the basis of a simulated release of 1 Bq/year for 500 years with an unchanged biosphere. For every radionuclide that occurs in connection with leakage from a repository, there is an EDF value for a given ecosystem (this concept was used for the first time in 1987 in a safety assessment of a repository for low and intermediate level waste, SFR-1 (1987)). The BIOPATH code was also used in this case. The dilution volume of the wells was also calculated in this case. Once again it was found that drinking water from wells was the most important pathway to man. The uncertainty analysis was conducted using the computer code, PRISM[18,25] and the conclusion was that the uncertainty in the dose to man was more dependent on the radionuclide behaviour in the biosphere in general than on the variation in different pathways to man.

7.4.4 SR 97 (1999)

In the latest safety assessment conducted by SKB, SR 97[(18)], the biosphere models were collected into a single code package, BIO42. BIO42 is included in a three-stage code for the calculation of exposure pathways for radionuclides from the source term in the repository via the geosphere (far field) out into the biosphere. The radionuclide composition of the leakage to the different ecosystems is calculated by models of ground-water flow in the far field (FARF31). An intrusion scenario was also included and the consequences calculated to the individuals and their families who unintentionally drill into the rock through the repository. As a matter of principle, intentional intrusion or terrorist actions are not included in these safety assessments, since that risk is considered to be generally applicable to many different kinds of industrial activities (see discussion in SR 97[(18)]). Extensive work on mapping the local ecosystems at three sites was conducted and this enabled the most advanced scenario descriptions and detailed consequence analyses so far to be conducted. Each site has been divided into quadratic zones that are 250x250 m^2, known as modules, and depending on the type of vegetation and terrain dominating the module, one of several possible ecosystem types have been assigned to the site[(26)]. The following ecosystem types were identified: *i.)* lake, *ii.)* well, *iii.)* watercourse, *iv.)* coastal strip/water, *v.)* agricultural land and *vi.)* marsh and wetlands (peat bogs). Certain additional details were introduced into the compartment models, based on the original BIOPATH models, such as erosion and the biological transport of radionuclides in soil. EDFs were determined for all five types of ecosystems and 44 different radionuclides. In this case, the EDFs were calculated on the basis of a simulated continuous unit release (1 Bq/year) for a period of 10,000 instead of 500 years in SKB-91. More advanced climate scenarios were also included with periodical glaciations and dynamic radionuclide transfer between different compartments. The result of the safety assessments are published in SR 97 – Post-closure Safety (Main Report in two volumes and Summary).

Figure 3
Principles for how consequence analysis for a given location, ecosystem type and radionuclide are estimated according to SR 97. EDF means Ecosystem-specific Dose Factor.

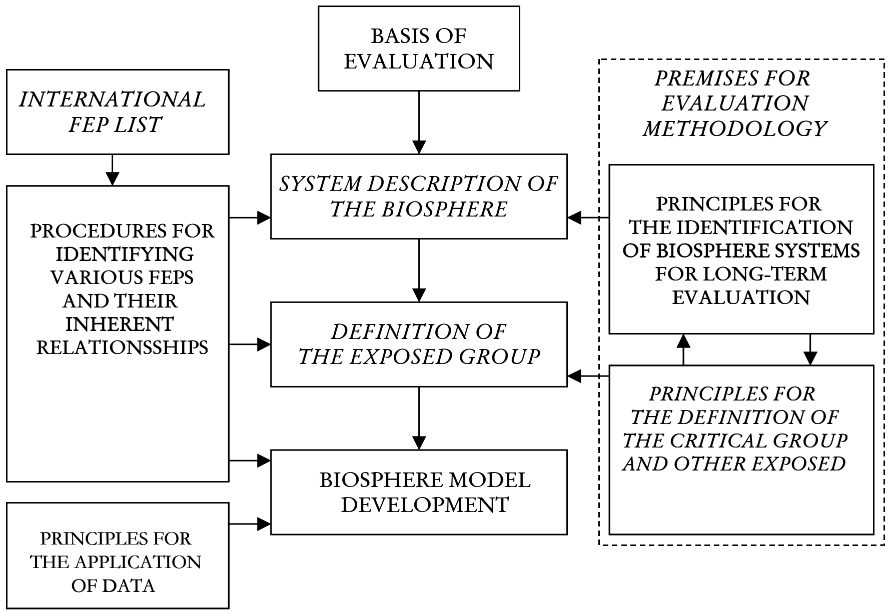


7.5 International model validation

In 1985, an extensive project was started in co-operation with about 20 other countries with the aim of validating existing biosphere models and of identifying the agreements and areas where continued research was needed. The project was called BIOMOVS (BIOspheric MOdel Validation Study)[(27)]. The models concerned dispersion processes of both long-term releases from the deep repository and acute releases from nuclear installations. The study showed that the results of different model calculations for relatively well-documented ecosystems could differ by up to a factor of 5–6.

BIOMOVS was considered to be a success and the project continued in the form of BIOMOVS II[28]. The aim of BIOMOVS II was to test the accuracy of various biosphere models and to explain the deviations between the different models detected in the first project. The aim was also to recommend priorities for future research to improve the accuracy of the predictions of the effective dose to man. A set of reference biospheres for long-lived waste that would make it easier to compare the result of different models was developed. One conclusion from BIOMOVS II[28] was that the models for radionuclide transport through the biosphere should be seen as an illustration of a possible consequence and only one step in the safety assessment process. Another important conclusion concerning long-lived radioactive waste is that dispersion models cannot be validated in the same way as, for example, models for the atmospheric dispersion of radioactive substances from the Chernobyl accident. The latter case involves severe releases over a relatively short time-period that can be used during our lifetime as a basis for validating dispersion models. The predictions concerning leakage from a deep repository must, on the other hand, apply over a time-period of several thousand years. The lessons from the atmospheric global fallout from nuclear weapons testing and from the Chernobyl accident provided us with important knowledge regarding which ecosystems are more important than others with respect to the transfer of radionuclides via the ecosystems to man (UNSCEAR 1977, 1982, 1993[29],[30],[31]). However, only limited parts of this knowledge can be applied to the transfer of groundwater-borne releases to flora (and subsequently to fauna and man). BIOMOVS II[28] therefore emphasizes that models of radionuclide transport from a deep repository for spent nuclear fuel must be thoroughly and systematically evaluated.

Figure 4
Figure of how the model development and evaluation strategy is created using features, events and processes (FEPs) taken from the BIOMASS document(32). The diagram provides a rough illustration of the relationships between the design and the definition of the critical group, procedures for the identification and selection of FEPs and the development of a reference biosphere.



In 1996, the IAEA started an additional international project, Programme on BIOsphere Modelling and ASSessment – BIOMASS[32], with the purpose of developing reference biospheres that would be applied in connection with safety assessments for the disposal of spent nuclear fuel. With reference biospheres, it would be possible to prepare international guidelines for long-term deep repository safety. It was also emphasized that there is a need to issue guidelines for detailed and site-specific processes in the biosphere modelling. An illustration of the methodology for the development of reference biospheres has been taken from a working document and is specified in Figure 4. BIOMASS is a continuation of BIOMOVS II, used to establish and develop a logical method of creating biosphere models using FEP lists (see "Facts"). FEPs are geological, chemical, ecological and physical Features, Events and Processes. The BIOMASS project mainly aims at developing practical examples of reference biospheres constructed using the FEP methodology. It is partly through this work that criticism of and weaknesses in SKB's safety assessment (see Section 7.9) have been made.

7.6 Comparison with model development by other countries

7.6.1 Great Britain

In 1991, an area outside Sellafield, Great Britain, was selected for closer investigations into a deep repository for low and intermediate level radioactive waste. A safety assessment was conducted and the principles used were described by Nirex[34]. An overall description of British knowledge of waste disposal, particularly with respect to safety assessments is also provided by Nirex[34]. According to the description, the British have conducted the same stepwise model development as SKB, where climate scenarios, dynamic transfer coefficients and consideration of retardation in the geo/biospheric interface were only used during the final phases of the safety assessment.

7.6.2 Finland

In Finland, work on safety assessments started in the early eighties, partly based on SKB's proposal for the deep disposal solution(15). Finland also used a reference site (Olkiluoto – located on the coast) that was then considered to be representative for possible future candidate sites within the country. Three scenarios were used for repository evolution: *i.)* a base scenario with continuous canister degradation with subsequent leakage that starts after one million years. The comparative scenarios were *ii.)* initial defect in one of the canisters with leakage starting after 100 years after deposition (the latter value is a very conservative calculation in accordance with SR 97's work(35), *iii.)* the same as the base scenario but with an oxidizing environment in the geosphere. In all of the scenarios, the transport time through the geosphere is estimated at about 5,000 years. In this case, the conclusion was also reached that drinking water from wells is the main exposure pathway to man. In scenarios *i.)* and *iii.)*, it was also taken into account that extensive climatic changes would occur in the distant future with subsequent changes in the groundwater flow and biosphere.

The latest safety assessment for a repository for spent nuclear fuel is called TILA-99(36) and has been conducted by POSIVA (the Finnish counterpart to SKB). POSIVA has worked with four different sites (one of which is at Olkiluoto). The radionuclide transport models are described as "non-complex". However, POSIVA points out that throughout the work the company has been conservative in its assumptions and used conservative models and calculations. The biosphere models have been deterministic and, as for the models of repository leakage and radionuclide transport through the geosphere, the FEP list methodology has been used to develop relevant scenarios and reference biospheres that can then be compared with international studies. Based on reasonable assumptions, the prediction for the base scenario is that the repository will not leak. In the two scenarios that involve leakage it is assumed that there is *i.)* an initial canister defect (one or more) as well as *ii.)* canisters that "disappear" after 10,000 years of disposal (i.e., that the entire inner barrier of the repository, in the form of the copper canister, ceases to exist, a mathematical operation for taking into account the worst case leakage situation).

As in Sweden, dose rate factors have been used to describe the consequence in terms of an effective dose from an individual radionuclide to a critical group if the nuclide reaches the biosphere from a leaking repository. The factors apply to a reference biosphere, WELL-97, where the main exposure pathway is drinking water via deep-drilled wells. This model has then been compared with the Swedish biosphere model that was developed by SKB 91[(25)] for an agricultural ecosystem (combined leakage to the well and lake). A sensitivity analysis was also conducted and extremely conservative assumptions were applied to determine the consequences in the form of dose values to the critical group. In no case was the acceptance criterion exceeded. The acceptance criterion in Finland is 0.1 mSv/year and individual, which is seven times higher than that applied by SSI (0.015 mSv/year and individual), and 3 times lower than the level recommended by the ICRP (ICRP 81[(37)]. One important detail is that, in the report[(36)], the radioactive inventory in the repository is compared with the levels of naturally occurring radionuclides in the typical Finnish bedrock. It is stated that the occurrence of ^{238}U and its daughters (such as ^{226}Ra) in a waste package corresponds to the quantity of uranium that occurs naturally in a 72x72x50 m^3 rock block.

In a new report from the Finnish Radiation Protection Institute[(38)], the consequence to mankind of the contamination of different simulated wells in one of the reference areas is calculated. The scenario assumes an initial defect in a single canister in the repository and calculates the dose to an individual who drinks 2 litres of well water a day (not including water for irrigation and cooking). In the report, it is also calculated that the dose rate cannot exceed $2x10^{-9}$ mSv/year. If all of the containers were damaged, this value must be multiplied by 860 (= the number of canisters in the deep repository). Also in this case, the annual dose is less than the Finnish acceptance criterion. It must be mentioned that the Finnish investigation used considerably larger dilution volumes for well water than those used in SR 97 (90,000 m^3/year, compared with 2,600 m3/year in SR 97). The report bases its assumed values on recent simulations that are also described in the work, where the conclusion is dilution volumes in the range of between 30,000 to 460,000 m^3/year.

7.6.3 USA

In the USA, deep disposal of low and intermediate level waste of military origin has already started (1999). The repository is located in New Mexico in a stable, 225 million year-old salt formation. The repository is near to a laboratory where geological investigations have been conducted(39). Even if this repository only is intended for operational waste (including transuranic elements) and nuclear fuel is not mentioned, research is conducted concerning processes controlling radionuclide transport through the geosphere (solubility, gas formation, water flow, colloid formation etc.). The research is conducted by Sandia National Laboratory(40) which also has an overall research programme for several types of risk waste – including non-radioactive waste. Safety assessments have been performed for radionuclide migration to the surface as well as for unintentional intrusion. The quantities that can leak out into the groundwater in a 10,000-year perspective have been taken into account and it was noted that the levels were not close to the values that would justify further assessments of the level of detail performed by SKB(41). The acceptance criterion for the operation of a repository in the US has been established at

0.15 mSv per year and individual. However, only a provisional model for radionuclide transport in the biosphere with conservative calculations was used as an assurance that the criterion was met. No specific detailed biosphere modelling for the area has been performed for the safety assessment.

For spent nuclear fuel, a similar safety assessment has not been conducted so far since many difficult legal problems have been encountered regarding how such work should start. The US Department of Energy is responsible for resolving the final disposal issue. Yucca Mountain in Nevada has been identified as a possible site for final disposal and investigations are in progress to ascertain the suitability of the site.

7.6.4 France

In France, ANDRA(8) is conducting research into underground deep disposal. As in Sweden, geology, geomechanics, hydrology and chemistry in the bedrock are being studied. Intensive research is being conducted in these areas. Scenario descriptions are conducted for geology at the repository and the surrounding environment. Even if ANDRA and the French expert group on radiation issues, ISPN(9), have biosphere models for radionuclide migration from a repository, it would seem that the French have not yet compiled their results into a basis for an evaluation of the total risk(42).

7.6.5 Japan

In 2000, Japan's counterpart to SKB, (Japan Nuclear Fuel Limited – JNC) presented an extensive safety assessment of a repository for high-level nuclear waste (in this case waste from nuclear fuel reprocessing) in the H12 report(43). It is emphasized that this safety assessment was conducted in the absence of any directly applicable legislation for such an activity in Japan. However, work was based on the recommendations and guidelines issued by the Japanese energy commission. A compilation in English is provided of a number of different details in the proposed solution to which not only JNC[(43)] has contributed with research and expertise. The aim is to demonstrate to the highly anti-nuclear Japanese public that, in spite of the very unstable tectonic and seismological conditions in Japan, a safe solution to final disposal can be found, similar to those found by other nuclear power-producing countries, involving geological disposal in multi-barrier systems. A strategy for developing scenarios in accordance with recommendations from NEA/OECD[(2)] has been applied and the scenarios divided into three classes: *i.)* a base scenario with an unchanged biosphere and absence of human impact, *ii.)* a disturbed environment – here a number of different scenarios involving well drilling and repository leakage interact to provide an exposure pathway to mankind as well as *iii.)* scenarios containing disturbed conditions in the proximity of the repository which involves volcanic activity and human intrusion.

In the scenario description, the Japanese have, like SKB, divided the transport models into different segments. In the Japanese report, the biosphere is represented by a topography specification (hill, flatlands etc.) as well as a specification of different geospheric/biospheric transitions (surface water, wells). The reference biosphere concept (cf the BIOMOVS Project[(13)]) was also used and the latest findings from the BIOMASS Project were taken into account[(32)]. In an earlier safety assessment, only groundwater concentrations of different radionuclides released were calculated. These values were compared with recommended safety levels for water. Scenarios with changed human activity or radically changed climate conditions were not calculated. The groundwater flows are taken from experimental data from a specific location in Japan. The hypothetical group of exposed individuals is assumed to conform to present-day Japanese living conditions with respect to fishing, agricultural activities etc. The strategy was to identify the most effective exposure pathways before defining the critical group. The reference case was selected as a leakage of radionuclides to a watercourse in a coastal flatland (80% of the Japanese population lives on the coastal flatlands), where most of the drinking water is taken from watercourses and dams. Interesting differences between the Japanese and Swedish biosphere model include the fact that, in Japan, retardation in the interface between the groundwater and surface water was not taken into account. Also, a very detailed, if somewhat difficult to grasp, illustration in the form of an interaction matrix was provided of the different relationships assumed between all of the processes and compartments included. The mathematical model in the form of the AMBER calculation code was used with first-order kinetic compartment models. In spite of efforts, it was necessary to use non-Japanese data for most of the parameter values apart from consumption data. For example, the same uptake values from the soil to rice plants as from the soil to traditional, western-type crops were assigned. The effective dose per unit of release has been calculated and is similar to the ecosystem-specific factors that SKB has used for thirty relevant radionuclides. The exposure pathways to man that are dominant are intake via drinking water and agricultural products. When these tables are combined with the actual source term from the leaking repository, the effective dose values are obtained. As is the case for the Swedish conditions, the dominant nuclides in the release are expected to be ^{135}Cs, ^{63}Nb, ^{79}Se and ^{237}Np. It is also interesting that an investigation has been carried out of how the presence of stable isotopes of the same elements as the radionuclide affect the dose calculation, for example, in the case for ^{79}Se. Dose calculations for the critical group are presented and the dose levels are about 0.01 mSv/year at the most.

7.6.6 Canada

In 1978, the Canadian Government started a programme to solve the nuclear waste issue, the Canadian Nuclear Fuel Waste Management Program (NFWMP). AECL[6] was then given the task of finding a technical solution to the final disposal of nuclear fuel. AECL has developed a very detailed description of its biosphere model that is a part of the overall safety assessment for the repository[44]. The model is called BIOTRAC, and was originally conceived in 1978. However, since then, it has been successively developed. The intention is to deposit the nuclear fuel in a geological repository 500 meters underground, which is similar to the solution that SKB is working on. BIOTRAC has therefore been specially developed for modelling radionuclide transport from the deep repository.

BIOTRAC comprises four sub-models that describe the soil/ground, surface water, atmosphere as well as the food chain and dose calculation to man. Wetlands of a temporary and permanent nature have both been considered to be a sub-group under the sub-model for surface water and for soil models. In turn, BIOTRAC is included in an integrated model for radionuclide transport that comprises the repository, geosphere and biosphere (also in this case there is a similarity with SKB and Finnish VTT's[7] division). The development of BIOTRAC was also given an impetus by a research committee which collected experimental data and presented annual reports of updates. In this way, it has been possible to improve the model and focus on and identify critical exposure pathways. In 1993, a report was presented[44] with a detailed description of the model's characteristics and how it had been developed. An interesting aspect is that, in this case as in Japan, an attempt was made to integrate the radiation aspect with the chemical toxicity of the different radioactive substances in the safety assessment. For the calculation of the dose to man, the recommendations of ICRP 26 were followed, which also means that a detailed safety assessment has not yet been conducted for other organisms besides man. However, as was the case with the Japanese JNC[(43)], an attempt was made with BIOTRAC to calculate the concentrations of radioactive substances with the levels that occur naturally in order to understand the consequences that leakage could have on the environment.

Although the details are somewhat unspecified, the characteristics of the biosphere models, with respect to local geographical conditions have been taken from the high-plateau outside Ontario, where a hypothetical repository is located. As with SKB, in the previous safety assessments, collective dose calculations were conducted. However, since, in accordance with the ICRP's guidelines, the acceptance criterion must be primarily determined on the basis of a critical group, work concentrated instead on the definition of such a group and dose calculations were conducted in accordance with the models used by the other countries. An acceptance criterion of the same level as in Sweden was used, namely a risk corresponding to 10^{-6} per year (see Section 7.3.3). In this case, the strategy recommended in the BIOMASS Project was also chosen[(32)]. This strategy entailed preparing a complete list of various Features, Events and Processes (FEPs) and, on the basis of this, working on a central scenario which involves a slow degradation of the repository and its barriers. The central scenario was formulated when it was determined that the FEP list was as complete as possible.

Stochastic compartment models were used in BIOTRAC, where each parameter was assigned a probability distribution. The distribution of these probability distributions does not only reflect the parameter uncertainty (as for example in the case of the fish population of a lake) but also includes temporary as well as local variations in the biosphere conditions (for example, the variation in the fish concentration in different parts of the lake as well as its seasonal variation). It is clearly stated that these distributions must take into account both the uncertainty of the parameter values and their temporal and spatial variation. Since

Canada has not specified a site for the deep repository, there were difficulties, in selecting representative parameters in the model. The groundwater flows, in particular, are very specific for a given geographical location and, therefore, a compromise had to be made through the use of hydrological and geological field data from a specific site where research is in progress (Whiteshell Research Area, Pinawa, Manitoba). Local topographical conditions were also used as a basis of the biosphere modelling in BIOTRAC in connection with the safety assessment. In spite of this, the objective was to obtain a "general" biosphere model that could be used in the safety assessment prior to the decision on repository siting. The composition of the critical group in BIOTRAC was defined in accordance with the ICRP guidelines[37] where present-day human living habits and behavioural patterns were used and where the size of farms and land areas were specified.

According to Canadian guidelines, the time frame for the predictions was set at 10,000 years. However, a prognosis for 100,000 years into the future was also made, although this did not take into account ice-ages. The report also describes the concept of conservatism, namely, the level of uncertainty with which radionuclide flows to man should be calculated. With respect to the definition of the critical group, model construction and values of input parameter values, conservatism has been applied. The report also states that, if the calculations are too conservative, there will be a risk of rejecting a solution that is actually better than that ultimately chosen.

In summary, BIOTRAC covers a number of different conditions, in spite of the fact that only a central scenario was used. This has to do with the fact that the biosphere characteristics could not be specified to any great extent.

7.7 Data uncertainty and probabilistic analysis

7.7.1 Model variance and sensitivity analysis

SKB distinguishes between three types of model calculations: two types where each individual parameter included in the model is assigned an unambiguous value and a third where a probability distribution is instead assigned for each parameter and where the middle point of the distribution corresponds to the estimated median value for the parameter. Furthermore, its width corresponds to the variance (stochastic model, see “Facts”). The former calculations are based on a set of reasonable assumptions and a set of conservative assumptions for parameter values. A comparison between the conservative and the reasonable scenario gives an idea of the margins included in the safety assessment in order to comply with the acceptance level. By using instead a set of probability distributed parameter values, a probabilistic analysis of the consequence can be conducted. There are two advantages of such an analysis of the biosphere model calculations. One advantage is that the safety assessment results – the consequence that is to be quantified in the form of the dose to the most exposed individuals – take the form of a median value and a probability distribution. In turn, this allows a clearer view to be obtained of the size of the uncertainty interval that applies to the final consequences on the basis of a given scenario. Another advantage is that, through a statistical analysis (covariance analysis and regression analysis), covariances or inherent relationships between different processes described by the model can be traced. This also means that the processes whose uncertainties will make the largest contribution to the total uncertainty of the radiation dose to the critical group can be identified. Processes or events in the radionuclide transport that are characterized by other processes can also be traced. In this case, it means that certain processes within the radionuclide transport through the ecosystems can counteract or stimulate each other so that the total transfer to mankind is affected. A good illustration of how such covariances affect the sensitivity of the biosphere model was done by Studsvik in the early eighties[24].

SR 97[18] shows the extent of the variations that can exist in the size of the ultimate dose to the critical group. The variance of the ecosystem-specific dose rate factor results in a dose difference of at least a factor of ten (depending on both the parameter values and assumed repository site and scenarios). The uncertainty in the dose contribution of each radionuclide to man has been classified into three categories: Uncertainties derived from *i.)* biological parameters, *ii.)* parameters that describe human behaviour (consumption of water, fish, meat etc.), *iii.)* physical-chemical parameters. These have been calculated for each ecosystem type. Through this process, one has also illustrated how the variation in local conditions affect the risk estimates. From the start, the Canadian BIOTRAC model integrated uncertainties in transfer constants with variations in different local conditions and variations in time for a given site, since, in contrast to SKB, one has not been able to work with the pre-investigation area and with more detailed ecosystem conditions.

British Nirex mentions that probabilistic calculations have also been an objective for them[34]. However, there is a difficulty associated with this type of probability calculation. SKB clearly states in SR 97[18] that a complete probabilistic assessment of the model is not possible if the sole aim is to base the uncertainties of each parameter on empirical data, since the experimental data base is not adequate. Consequently, a series of probabilities must be based on hypotheses and on general standard distributions, for example those that have been recommended by the IAEA[17, 25].

7.7.2 The Variance between different models

The summary of BIOMOVS[27] (1985-90) presented in 1993 specified a number of causes of uncertainties and inaccuracies in the biosphere models.

- Validity of the structure of the model: For example, can the sedimentation rate of the radionuclides in the water phase be described as a first order differential equation?
- Accuracy of the mathematical description of the process: How correct are the parameter values that specify how rapidly a radionuclide migrates from one compartment to the next, for example, for the accumulation of ^{135}Cs in lake and seawater fish?
- Human factors – incorrect programming (namely, incorrect input values, incorrect calculation routines etc.), evaluation errors and incorrect interpretation by the operators.

It was also stated that increased complexity in the model does not guarantee that the final result in the form of the calculated dose to the critical group will be more accurate. This is due to the fact that greater complexity in the model must be based on more experimental and theoretical assumptions (in many cases, the databases are very limited). The study also stated that the variance (see "Facts") in the calculated ultimate dose to man between different models amounted to almost a factor of 100,000 for long-lived nuclides (that can leak from a deep repository). It was remarkable that such well-documented processes as transfer via the pastureland-cow-milk pathway could yield such different final results depending on the calculation model used. Since this, major improvements have been made to model arable land and pastureland. However, the mapping of radionuclide migration through natural ecosystems is relatively deficient.

7.8 Natural analogues

In Sweden, in the early eighties, experimental studies of naturally occurring radium, thorium and uranium in lakes and coastal water systems were conducted. The distribution between the concentration of ^{226}Ra in different "compartments" such as lake water and bottom sediment and the corresponding compartments for coastal water were determined.

In the Nirex report[34], the value of comparing the dispersion model with the processes that determine the turnover of naturally-occurring radionuclides in the groundwater and water catchments is mentioned. The repository will contain long-lived transuranic elements and uranium isotopes with the same physical half-lives as the naturally-occurring isotopes of uranium and their daughters. A database of the occurrence of natural thorium and radium in wells, water courses, sea and lake water in the region around Sellafield has been compiled and has been used to calculate the corresponding transfer processes for the short-lived artificial nuclides.

In the international model validation study, BIOMOVS II[28], it was stated that few groups prioritized studies of natural analogues. The focus was primarily on the turnover of artificial nuclides from acute releases.

7.9 Review of SKB's safety assessment

7.9.1 SSI and SKI's review of SR 97

In their joint review[49], SSI and SKI considered that SKB should clarify the arguments for the selection of important data and models and why certain unfavourable relationships are not treated in the risk assessment. As far as biosphere models are concerned, SSI considers that SKB should more clearly have specified the system for the development of biosphere models in a systematic way, based on different Features, Events and

Properties (FEPs), which more completely describe a situation that involves risk (see Section 7.2.1). Scenario selection must be connected to the risk assessment. This means that there must be a system for the selection of FEPs for the most important scenarios and SSI points out that this has not been adequately achieved in SR 97.

Furthermore, it was emphasized, as in the review by the NEA's International Review Team[(45)], that the evaluation of the total environmental protection is deficient. SKB is criticized for only taking into account the effects of radiation in the form of a risk of cancer and for not considering the overall detrimental effect. A task group within the ICRP is currently working on developing a philosophical framework for how the impact of radiation on the environment and nature should be integrated into the overall risk and consequence assessment of the type conducted by SKB in SR 97.

7.9.2 The OECD/NEA review of SR 97

On the whole, SKB is commended by the NEA for the quality of its work in SR 97[(45)]. On the other hand, proposals are presented for improvements that should be made within the framework of SKB's safety assessment. The NEA points out that a more traceable and clear sensitivity analysis and investigation of model uncertainties is necessary. SKB is criticized for its classification into conservative and reasonable parameter values in the model presented in the probabilistic assessment. For some of the assumptions, the NEA review considers that SKB is unclear or lacking a basis. An improvement is necessary in order to be able to do a definitive site selection in the near future. Furthermore, the NEA, like SSI and SKI, would like to see an overall consequence analysis for nature and the environment. This also indicates further pressure on SKB to expand the concept of risk to include other forms of life apart from mankind. Furthermore, the NEA does not consider that it is sufficient to perform conservative calculations in order to be on the safe side in its risk assessments. More realistic models must be used (see 7.6.6). The NEA also stated that SKB needed to improve or clarify the strategy behind the selection of the scenarios that it works with, which is also in line with the objectives of the BIOMASS Project[37]. Models cannot be validated but they can be "invalidated", which is something that the NEA also points out.

7.10 The need for future research

7.10.1 General

In general, the biosphere modelling in SKB's safety assessment is probably the most complex and detailed analysis so far, also from the international standpoint, in spite of the criticism directed to the fact that SKB did not have a clearly defined strategy in its scenario descriptions (see Section 7.9). SKB has the largest number of ecosystems in its biosphere modelling and, furthermore, has site-specific data from three different locations in Sweden. After more than 20 years of refinement, SKB's models indicate that the most significant exposure pathways to man comprise fresh water uptake via contaminated wells and the cultivation of marshlands and peat bogs with a high radionuclide content and high root uptake. In the light of this, the value can be questioned of further refining models for less significant exposure pathways with respect to radiation dose to man, such as via the saltwater ecosystem, where it can be certainly assumed that dilution will mitigate the consequences of a release via the groundwater or, for example, via highly cultivated arable land, where root uptake is low compared with peat bogs. Furthermore, it must not be forgotten that the natural occurrence of radium and thorium constantly result in annual dose contributions to man via exposed water catchments, with radiation levels and corresponding risks that significantly exceed the regulatory acceptance criterion for the final disposal of nuclear waste. The natural dose contribution via drinking water is otherwise considered to be a significant radiation protection problem (SSI). One deficiency that SKB is also aware of is that it has not managed to incorporate forest ecosystems in its biosphere models. The type of natural conservation associated with the definition of the critical group includes a large number of forest products, such as mushrooms, berries, game etc. Based on the experience from the Chernobyl accident, it is known that these products are significant stages in the transfer chain of radionuclides to man, that is, large quantities of certain radionuclides tend to end up in these products for a given concentration in the soil. However, it is once again difficult to draw excessively large parallels with the Chernobyl accident, since future leakage from the repository will largely originate from deeper soil layers and be transported upwards via root uptake and capillary movements. To a certain extent, SKB considers that it compensates for the absence of forest ecosystems in the biosphere model by very conservative calculations for the total radionuclide transfer from wetlands and peat bogs.

A research project is currently being conducted which aims at validating a model for the turnover of radioactive caesium from a geological formation in forest ecosystems(46). SKB is attempting to link the behaviour of caesium to the turnover of carbon in nature since, in a long-term perspective, the food and energy balance determines the transfer of a substance from different trophic levels to man. It would be desirable to be able to subsequently extend such a model to include other radionuclides besides caesium and, in this way, include the forest ecosystem in the safety assessment for geological deep disposal. The release contributions should be put in proportion by also calculating the doses from the water as a result of the possibility that the bedrock may contain considerable radon.

A detailed review of the different modules in SKB's BIO42 package shows that it is obvious that SKB has very conservatively calculated or selected levels, for example, the annual water consumptions, the annual milk consumption, etc. If the acceptance criterion for a risk that is less than 10^{-6} have been met in accordance with the calculation model, in spite of all of the conservative parameter levels, then the value of further refining models for future populations should also be questioned. There is naturally considerable knowledge to be gained with research work concerning the radionuclide transfer chain from different ecosystems to man. However, this should be applied to present-day living conditions and for situations that are acute in nature, for example, in connection with extensive, uncontrolled releases such as those from the Chernobyl accident where the relatively rapid dispersion process in the lower atmosphere can result in high local concentrations of fallout.

7.10.2 Forest ecosystem

The forest ecosystem should be described in a model as soon as possible.

7.10.3 Environmental protection

SSI and SKB are both included among the contractual parties in the European Union's Fifth Framework Programme, FASSET[(20)], which aims at developing a framework for the radiation protection of the whole environment, focusing on flora and fauna. The intention is to identify target organisms in different European ecosystems and to develop corresponding dosimetric models for man. The intention is also to study the biological effects and damage that radiation can give to non-human tissue, known as end-points. Previously, such studies have been conducted on man or on laboratory animals (mainly mice). However, the aim of such studies was to calculate human response to radiation. The objective of this project is that safety assessment and consequence analysis for the overall environment should be made possible and not only limited to man. In connection with the 1992 Declaration of Rio (UNCED[47]) a number of general environmental protection principles were determined. On the basis of these, the intention of FASSET was to develop guidelines for consequence analysis. Similar projects are also in progress within the ICRP where the intention is to expand and develop the guidelines that were conducted in 1990 for man. Some discussions on these problems have also been conducted in UNSCEAR[31] and IAEA[48].

Appendix 1 – Compartment models

Relationship between two compartments

Figure 5 illustrates a simple system of two compartments. The transport of radionuclide D between the two compartments is described mathematically using a first order differential equation (static or dynamic) as follows (Eq. 2.):

N_i = activity concentration of radionuclide D in compartment i (for example a water course) [Bq m^{-3}]
N_j = activity concentration of radionuclide D in compartment j (for example the bottom sediment of a water course) [Bq m^{-3}]
M_i = the activity concentration of radionuclide D's mother nuclide in compartment i
$F_{i,j} = f_{i,j} \times N_i$ = transfer rate of radionuclide D from compartment i to compartment j (for example the sedimentation of radionuclides in the water course to the bottom sediment) [year^{-1}] (2.)
$F_{j,i} = f_{j,i} \times N_j$ = transfer rate in the opposite direction from compartment j to compartment i (for example, resuspension of sedimented radionuclides to the water course) [year^{-1}]

The transfer rate from one compartment to another is the product between the transfer coefficient, $f_{i,j}$, and the concentration of radionuclide D in the compartment from which a given process transfers the substance. In this case, for compartment j the change in the concentration of radionuclide D per time unit is as follows (Eq. 3.):

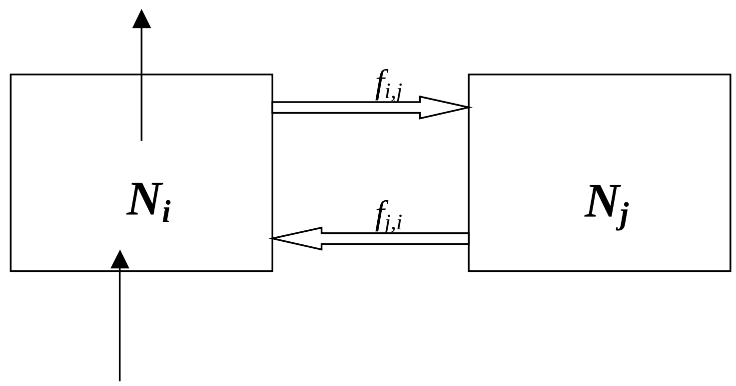
$$(\mathrm{d}N_j/\mathrm{d}t) = F_{i,j} - F_{j,i} = f_{i,j} \times N_i - f_{j,i} \times N_j \qquad (3.)$$

If radionuclide D's physical half-life is of the same order of magnitude as its residence time in the compartment/system or shorter, losses via decay must also be taken into consideration. It may also be the case that the radionuclide is continuously generated through the physical decay of its mother nuclide, M. If we include these two factors in the calculation, we obtain the following (Eq. 4.):

$$(\mathrm{d}N_j/\mathrm{d}t) = F_{i,j} - F_{j,i} = f_{i,j} \times N_i - f_{j,i} \times N_j - \lambda_D \times N_i + \lambda_M \times NM_i \qquad (4.)$$

where λ is the decay constant for each radionuclide. The decay constant states the probability of the size of the population of radionuclides that decays per unit of time. NM_i corresponds to the radionuclide concentration of the mother nuclide M in compartment i.

Figure 5
Diagram showing the relationship between the radionuclide concentration and transfer between two different compartments in a biosphere model. $f_{i,j}$ and $f_{j,i}$ are transfer coefficients that describe the rate with which a substance is transferred from one compartment to another. The values of f are parameter values that the model developer assigns on the basis of experimental data or assumptions. The vertical arrows describe the net transfer flow to compartment i from other compartments/systems.



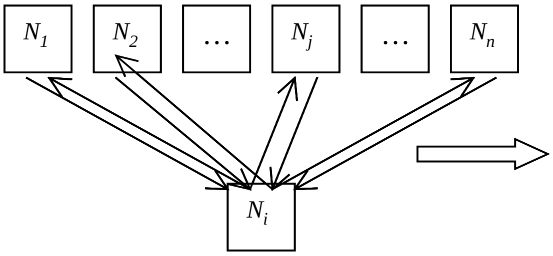
Relationship between a system of n compartments

In a complete system of *n* compartments, relationships between the concentration of radionuclides *D* in compartment *i* and all of the other compartments can be generalized using the following mathematical expression (Eq. 5.):

$$\frac{dN_i}{dt} = \left[\sum_{j \neq i}^{n} f_{j,i} N_j + \lambda_M NM_i + S_i(t) \right] - \left[\sum_{j \neq i}^{n} f_{i,j} N_i + \lambda_D N_i \right] \quad (5.)$$

If it is the concentration of radionuclide *D* in compartment *i* that is of interest, all of the transfer coefficients ($f_{i,j}$) with the *n*-1 other compartments must be described. Contributions from all compartments apart from compartment *i* itself are aggregated (index *j* varies from 1 to *n*). In many cases, there is no direct link between certain compartments, for example, the radionuclide concentration in the sediment layer of a water course does not depend on the radionuclide concentration in grass in an adjacent pasture. The transfer coefficients between them would thereby be 0 in the matrix in Figure 6.

Figure 6
Diagram of the relationship between a system of n different compartments, showing how these compartments, including their inherent transfers, can be represented in a matrix of different transfer coefficients.



j \ i	1	2	…	n
1	–	$f_{1,2}$	$f_{1,j}$	$f_{1,n}$
2	$f_{2,1}$	–	$f_{2,j}$	$f_{2,n}$
…	$f_{i,1}$	$f_{i,2}$	–	$f_{i,n}$
n	$f_{n,1}$	$f_{n,2}$	$f_{n,j}$	–

On the other hand, an indirect connection can exist if the grass in the pasture is irrigated with water from the water course. It is partly up to the model developer to classify and determine which relationships are considered to be of sufficient importance to be included in the compartment model. An additional aspect is the compartmentalization and the number of compartments (model structure). However, for calculation-related reasons, an unlimited number of compartments cannot be used to completely describe an ecosystem or a biosphere and certain simplifications are, thereby, unavoidable.

The effect of the physical decay of the substance during transport and residence in different compartments (or reservoirs, such as water courses and lakes) is also included in the mathematical expression shown above. The continuous occurrence of the radionuclide from the physical decay of the mother nuclide M may also be of the order of magnitude that this must be included in the calculations. The constants λ_M are in reverse proportion to the physical half-life of the mother nuclide.

The term $S_i(t)$ in (5.) includes the effect of the possibility that the observed radionuclide D can be transferred to one of the compartments in the model from a "source" or a system that does not have a compartment represented in the model. For reasons mentioned before, an indefinite number of "compartments" cannot be included in order to achieve a complete description of an ecosystem or a biosphere. Consequently, this source term is a way of compensating for this.

Dynamic compartment models

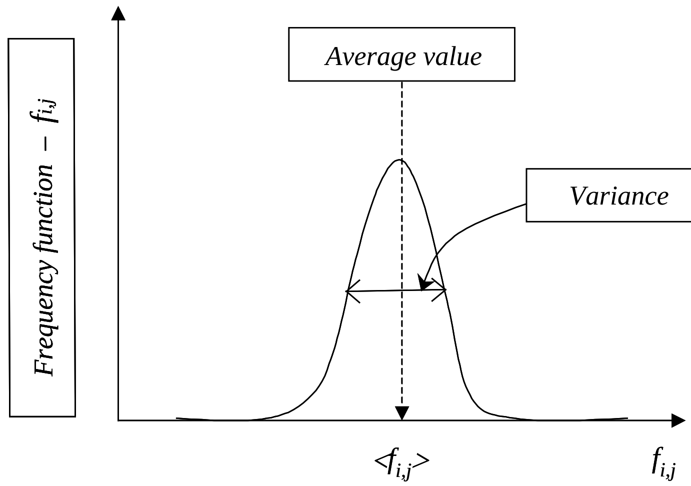
If a dynamic process, that is, a process that varies in time, is to be described with the transfer coefficient $f_{i,j}$, the coefficients f must also vary in time, namely $f_{i,j} = f_{i,j}(t)$. In other words, a *dynamic* compartment model must be used. It is also up to the model developer to determine the cases where it may be advantageous to use a time-varying relationship between different compartments. If we take the relationship between the transfer rate in a certain process that leads from compartment i to j: $F_{i,j} = f_{i,j} \text{ x } N_i$ and instead write $F_{i,j} = f_{i,j}(t) \text{ x } N_i$, we will obtain a description of a system where the transfer coefficients will be time-dependent. This will mean that the solution to the system of differential equations that describe the transfer processes must be resolved using other methods besides those that apply in the stationary case and the computer code will require longer calculation times.

Stochastic compartment model

In stochastic compartment models, it is assumed that each value for the transfer coefficients can vary between different values, in accordance with the pattern in Figure 7. The variation pattern is called a probability distribution and takes into account the local and temporal uncertainty and variance for the transfer coefficient concerned. The probability distribution is characterized by a median value and a variance which is a measure of the spread or uncertainty in the value of $f_{i,j}$. Using a computer code, these uncertainties can be included in the calculation and an idea of the size of the variation in the final calculated dose to the critical group can be obtained.

Figure 7

Example of a typical probability distribution for a parameter – transfer coefficient that can be used in stochastic compartment models. The frequency function is a measure of the probability that the parameter f_{ij} will assume a given value.



Facts

bioturbation	Process associated with the transport of substances resulting from the movement of worms and insects in the soil.
collective dose	The sum of all individual doses within a group. The size of the group can vary from all of the personnel employed at a nuclear facility to the entire population of a nation, or all of the people in the world. When reference is made to the collective dose to a defined group the aim is to obtain an estimate of the consequences of the total late radiation effects to the group in question, given that each radiation dose leads to a certain but low risk for late effects (cancer). The total of the varying individual doses is therefore proportional to the statistically expected number of cancer cases.
covariance	Describes how a parameter/variable is dependent on another parameter. For example: the water level of a lake is regularly measured and its variation over time (over a year, from year to year) is determined. This allows a description to be made of the variance of the water level. At the same time, the precipitation in the region is measured and the annual variation tracked. It may be found that the variation in the water level of the lake has the same pattern as the precipitation variation, with a certain time delay. If this is so, it can be said that the water level and precipitation are covariant parameters (inherently related).

deterministic	Deterministic – predictable. Deterministic radiation effects refer to the effects that can be predicted to occur if the person is exposed to a certain dose level. If a person receives a radiation dose to the lens of the eye of more than about 2 gray (see below), the lens will become permanently opaque. The damage is therefore pre-determined or deterministic.
diffusion	Random process determined by thermal and other physical driving forces that results in an attempt by the substances to distribute themselves evenly over as large an area as possible.
FEPs, FEP list	Features, Events and Processes, FEPs, in a given system which have a certain probability of occurrence. Each individual feature, event or process, for example the displacement in the bedrock that occurs following a glacial retreat over a land mass, can be combined with other FEPs to make a scenario. For example, it can be assumed that this displacement occurs at the deep repository site, combined with unfavourable flow conditions in the bedrock which result in rapid radionuclide transfer and migration from the repository to the biosphere. Many safety assessments conducted abroad have used this method to establish lists of FEPs which, after assessment and screening on the basis of low probability, are combined into different scenarios. The consequence to the critical group in the form of an effective dose is then calculated on the basis of the scenarios.

gray (Gy)	Basic unit to describe the radiation dose in joule per kg to a medium. Most often the radiation dose to individual organs in the body is expressed in milligray (0.001 gray).
mass transfer	Transfer of radionuclides that are clustered to large particles which, in turn, move due to gravitation and pressure, for example in a sediment layer.
migration	Directed radionuclide transport due to rain-water/precipitation that penetrates into the soil and results in an active radionuclide migration downwards.
probabilistic assessment	A collective name for the types of compartment models where the parameters are allowed to be stochastic, namely, the computer code can simulate different probable values for transfer coefficients on the basis of postulated probability distributions. A probabilistic assessment also takes into account the probability of the assumed scenario and different scenarios may then be weighed together into a weighted average value for the consequence in the form of an effective dose to the critical group.
resuspension	Transfer of substances from a solid phase deposit to another medium which is in a gaseous or liquid form, for example, from land and vegetation to air, or from bottom sediment to seawater.
sievert (Sv)	Unit for the *effective* dose of radiation. It is the effective dose that is assumed to be proportional to the risk for late effects, in accordance with Eq. 1 (Section 7.3.3).

stochastic	Stochastic – random; a stochastic process is a process where the outcome cannot be predicted in an individual case, but where chance determines the outcome to a certain extent. **Radiation doses and risk** It is not possible to say with 100% certainty that a given radiation dose will result in late effects in the form of cancer of e.g. the large intestine 30 years after exposure. On the other hand, it can be said that there is a certain probability that this will occur. In the case of whole-body irradiation, the probability of receiving some radiation related lethal cancer is about 10% per received Sv for high doses and dose rates and 5% per Sv for low doses and dose rates. Estimations of the detriment due to non-fatal cancer and severe hereditary effects for low dose and low dose rate exposure has given 1.0% and 1.3% per Sv respectively, altogether 7.3% per Sv (which explains the factor 0.073 in equation 1 for risk). **Parameters in compartment models** Stochastic distributed parameter values for a biosphere model concern the randomness that exists when a parameter is determined physically/chemically/biologically. For example, the fish population in a lake cannot be exactly determined since the measurement method is associated with measurement uncertainties which mean that we obtain a result with a certain probability. If we know the probability distribution that determines

	the uncertainties in the measurement process (which is seldom known, but which can often be estimated), and if we know the average value on which this distribution is centred, the computer code can simulate various probable values for the fish population in the lake. By simulating a series of stochastically-distributed parameter values in the compartment model, an idea can be obtained, in turn, of the size of the random variations that each parameter finally results in (radiation dose to the critical group).
variance	Concerns the numerical value of the magnitude/variation of the parameter. Different types of variances exist: variances over a geographical area or over time, as well as the variance associated with the measurement uncertainty.

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8 Nuclear waste management in some countries[1]

8.1 Introduction

This chapter presents a brief overview of how the waste management issue is handled in a number of selected countries outside Sweden. The emphasis is on high level waste and spent fuel, but some information on low level wastes (LLW) and short-lived intermediate level wastes (ILW) has also been included, as many of the issues associated with siting etc. tend to be common to all types of wastes.

In addition, some information is also included outlining waste management-related activities in some of the major international organisations (IAEA, OECD/NEA, EU).

In order to give a useful breadth to the information, the chapter includes countries with very different nuclear policies and very different waste management programmes. Therefore, a number of European countries, together with Canada, Japan and USA, are discussed. Some of these countries (e.g. France and Japan) have a strong and ongoing nuclear power programme, whereas most of the others have a more static or even decreasing nuclear programme, as in Sweden.

The information is based on "National Profiles", a set of information sheets which are produced by Phil Richardson of EnvirosQuantiSci (U.K.) for a large number of countries all over the world and which are updated on a regular basis. Many other

[1] This chapter has been prepared by Phil Richardson. EnvirosQuantiSci, U.K. and Tor Leif Andersson, Secretary of KASAM.

colleagues abroad have also kindly helped us with material for this chapter.

8.2 Canada

8.2.1 Nuclear power programme

At the end of 1999 there were 22 licensed reactors in Canada, of which 14 are currently operating, mainly in Ontario, with one each in Quebec and New Brunswick. They are owned and operated by the provincial government utilities, namely Ontario Power Generation Inc. (OPG), Hydro Quebec and New Brunswick Power, although some are to be operated by British Energy. Eight other reactors are currently shut down.

8.2.2 Relevant institutions

Nuclear energy in Canada is regulated by the Canadian Nuclear Safety Commission (CNSC) which replaced the Atomic Energy Control Board (AECB) in June 2000 under a new Nuclear Safety and Control Act. CNSC is an agency of the federal government, and licenses sites for storage of radioactive wastes, and publishes guidelines for disposal. Atomic Energy of Canada Ltd, (AECL) a federal Crown Corporation, has the mandate to develop and promote nuclear power, and sell the CANDU reactor overseas, which is a unique Canadian design, based on natural uranium fuel, moderated and cooled by heavy water.

The primary responsibility for management of radioactive wastes in Canada rests with the producers of the wastes. Provincial governments are responsible for long-term security of uranium mining and milling wastes, whilst the federal government provides funding for the development of a technology for permanent disposal of nuclear fuel. Canada does not propose to reprocess any of its commercial spent fuel, although some HLW will be generated from reprocessing of research fuels etc. For the past 25 years, commercial spent fuel has been stored in ponds at the generating sites, and dry storage of spent fuel, in concrete canisters at each of the NPP sites, has been in use since 1988.

8.2.3 Management of nuclear waste

8.2.3.1 LLW and short-lived ILW

Canada makes a distinction between so-called 'current ' arisings and 'historic ' wastes from past uranium milling activities, most of which are located around Lake Ontario.

Options studies for a final disposal site for 'current ' LLW are being performed by OPG with operation planned to begin by 2015. There is an operational storage facility adjacent to an existing reactor site.

In order to identify an acceptable disposal site for the 'historic' wastes, the 'Co-operative Siting Process' was established in 1986. A Task Force undertook extensive public consultation, and invited interested communities to volunteer for site development. Two communities were finally identified in 1994, although one withdrew shortly after. Following a positive referendum vote in 1995, the remaining community council signed an Agreement in Principle to allow work to continue, but this lapsed by the end of 1996 when the federal government refused to accept the terms of the Agreement. There are currently suggestions of a commercial venture involving the local council.

8.2.3.2 Spent fuel and/or HLW

The final disposal concept proposed for Canada, by AECL, involved placing spent fuel at a depth of some 500–1 000 metres in the crystalline rocks of the Canadian Shield. The repository was originally planned to be in service by 2025 and take some 40 years to fill, before it was to be sealed and abandoned. However, no siting-related work was permitted before concept approval. As part of an Environmental Impact Assessment process the disposal concept was examined in a series of public hearings, before a federally nominated expert panel, in 1996/7. In March 1998 the panel recommended that although the technical aspects of the concept appeared satisfactory, there was insufficient public acceptance to allow siting to begin (see also chapter 2 of this report).

Amongst its recommendations the panel said that a number of steps were necessary from government to achieve broad public support. These were:

- Issuing a policy statement on waste management;
- Initiating an Aboriginal participation process;
- Creating a new nuclear fuel waste management agency (NFWMA);
- Carrying out a public review of AECB's (now CNSC) regulatory documents;
- Developing a comprehensive public participation plan;
- Developing an ethical and social assessment framework;
- Developing and comparing other options for managing nuclear fuel wastes.

It also said that the NFWMA should be established as soon as possible, be funded by the waste producers only, and have a board of directors representing all key stakeholders. In addition there should be a strong and active advisory council representing all interested parties.

Until these recommendations have been implemented and a broader public acceptance of the management concept has been achieved, the panel concluded that the search for a specific repository site should not proceed.

The Ministry of Natural Resources (NRCan) issued its response statement to the panel's report on 3 December 1998.

Whilst agreeing to the establishment of a semi-independent agency (i.e. an agency formally attached to a government department but with great freedom to act autonomously on most matters) to carry out future work on waste management and disposal, NRCan rejected the suggestion that siting work for a repository should be postponed. It also gave overall responsibility for establishing the new agency to the waste producers and owners, who will have total control over the makeup of its board of directors.

The statement set out three policy objectives to be followed;

- The establishment of a dedicated fund for spent fuel management by the producers and owners;
- The new agency must report regularly to the federal government on its activities;
- A federal review and oversight mechanism must be established to provide oversight and access to funds.

The statement also proposed an unspecified period of public consultation to establish options as to how these objectives should be met, and required the agency to make its recommendations within 12 months of being set up, including a description of how it proposed to carry out public participation and foster involvement of the public, especially that of Aboriginal People. It is also to develop an amended AECL-type concept and to review other management methods, including extended storage at either reactor sites or at some form of centralised facility, either above or below ground.

To date (November 2000), no final details have emerged of the new waste management organisation. As regards a timetable for its establishment, this will probably become clearer following the recent federal elections. Most observers envisage some action around 2001–2002. An operational repository by 2035 is still the intention.

8.3 Finland

8.3.1 Nuclear power programme

There are two commercial reactor sites in Finland, each currently with two reactors, one at Loviisa operated by Fortum (former IVO), with Russian-built VVER 440's and one at Olkiluoto operated by TVO, with two Swedish-built boiling water reactors. An application for a ”Decision in principle” has been made to the Government concerning a fifth reactor, to be built at either of the two reactor sites.

8.3.2 Relevant institutions

The two utilities are responsible for the safe management of wastes and for the necessary research and development as well as for covering the costs of the whole operation. The objectives and schedules of waste management are set out in a government policy from 1983, with the regulatory basis set out in the 1988 Nuclear Energy Act and Decree. The Ministry of Trade and Industry (KTM) supervises waste management activities and the R&D work. It also finances research in order to maintain independent expertise. The Finnish Centre for Radiation and Nuclear Safety (STUK) is responsible for the regulation and supervision of the safety of nuclear facilities and review and assessment of waste management plans and activities. Facilities must be licensed by government. The KTM decides each year the fees that the utilities must pay into a government controlled Nuclear Waste Fund, designed to cover the future costs of waste management.

In the past both utilities followed different spent fuel management strategies. Fuel from Loviisa was shipped back to Russia for storage and reprocessing, whereas it was stored on site at Olkiluoto, in a water pool storage facility. However, Russia is not accepting the spent fuel any more and new solutions became necessary. An amendment to the Nuclear Energy Act was passed in late 1994, such that no spent fuel was to be exported after 1996. IVO and TVO formed a joint company, Posiva, to carry out all disposal related work.

8.3.3 Management of nuclear waste

Waste classification in Finland distinguishes between low and intermediate level wastes and spent nuclear fuel, which is not to be reprocessed.

8.3.3.1 LLW and short-lived ILW

Both nuclear utilities have developed rock cavity repositories adjacent to their existing reactor sites, using vertical silos and/or horizontal caverns. These began operation in 1992 and 1998 respectively.

8.3.3.2 Spent fuel and long-lived ILW

Following a KTM decision in 1991, which followed on from a Decision in Principle of the Council of State in 1983, deep disposal is the chosen route for spent fuel. This was formally reiterated in the Nuclear Energy Act and Decree passed in 1988.

The latest proposal in the so-called 'TILA-99' Safety Assessment, released in 1999, calls for a repository using a disposal concept similar to KBS-3 in Sweden, to be built at a depth of 400–700 metres, depending on final site conditions.

Posiva proposes that the final design at the actual repository site would be produced on a 'design-as-you-go' basis, allowing flexibility dependent on actual geological conditions found during development, whilst realistically accepting that some deposition holes may actually be placed in poor conditions.

The estimated cost of the final disposal for spent nuclear fuel is around 5000 million Finnish marks (about 800 million $US).

Five potential repository locations were identified for future work by TVO, from an original shortlist of 85 potential areas, drawn up between 1983–1985. Surface-based work took place during 1987–1992 at all five sites, with five deep (500–1 000 metres) boreholes at each, together with geophysical work. The government decided that the objectives of the whole investigation phase should be to complete preliminary site investigations by the end of 1992, select two or three sites for further study and to update the technical plans by the end of 1992. Consequently, three sites were selected for further study in December 1992, at Eurajoki, near Olkiluoto (close to the NPP), Romuvaara in Kuhmo and Kivetty in Äänenkoski.

In addition to these sites, Posiva also undertook detailed investigations near the NPP at Loviisa in 1997, on the island of Hästholmen. Four boreholes, each 1000m deep, were drilled, and wide ranging hydrogeological testing carried out.

In January 1998 Posiva submitted an *'Environmental Impact Assessment Programme'* to the KTM, and at the same time it was published for general review and comment. It was also circulated to the Swedish, Estonian and Russian authorities, as required under the Espoo Convention.

Following a series of public hearings in Spring 1998, KTM commented to Posiva, in late-June 1998, and requested that more work should be done on assessing the potential impact as regards the radiological safety of a 'zero option'. In addition it said that retrievability should also be examined, as well as a number of alternative disposal concepts, such as horizontal emplacement, use of a hydraulic cage and deep boreholes. Posiva published the final version of the Environmental Impact Statement on 26th May 1999 and thereafter submitted an application to the government for a Decision in Principle.

An international panel was assembled by STUK to examine the safety case of Posiva's application for a Decision in Principle and this reported in autumn 1999. The panel recommended, amongst other things, that STUK should conduct a number of additional reviews in the future, following government granting of the Decision in Principle. These should include:

- Review of Posiva's site characterisation plans before shaft construction begins;
- Review of Posiva's proposed underground experimental programme;
- Regular (every 3–4 years) review of the content and achievements of Posiva's R&D Programme (as occurs in Sweden);
- Review of Posiva's interim safety assessments;
- Implementation of other key elements of an independent assessment capability to increase public confidence.

STUK issued its own report based on the Panel's assessment in January 2000, and this supported Posiva's request to continue with its proposals at Olkiluoto. Approval of a repository by the municipality is required by law. Therefore the municipality council held a vote in January 2000, and this too was in favour of the facility, by a margin of 20 to 7.

All the review material and the ministry's summary were made available to the public in spring 2000.

The Decision in Principle was delayed by several months because of a judicial appeal by two citizens from the community. In April 2000 they each applied for a court hearing to question the legality of the process by which the area was selected as potentially suitable. The applications were rejected in early June, as were appeals to the Supreme Court in November. The Decision in Principle was made by the government in late December 2000, and it is now estimated that the Parliament will make its decision during the spring 2001.

8.4 France

8.4.1 Nuclear power programme

By end-1999 there were 58 PWR units in operation at 29 sites in France, together with a commercial reprocessing plant on the northern coast at Cap de la Hague.

8.4.2 Relevant institutions

Under legislation passed in 1975, any producer of waste must arrange for its disposal, at its own cost, by a body approved by the public authorities. For this purpose the government set up a specialised agency, ANDRA (National Agency for Radioactive Waste Management) in 1979, within the Atomic Energy Commission (CEA). This is responsible for designing, constructing and operating, long-term disposal facilities as well as undertaking all necessary studies to this end, and for promoting the application of technical specifications for waste treatment to be carried out by producers prior to storage.

ANDRA is financed by the waste producers, in particular Electricité de France, CEA and fuel-cycle firms such as COGEMA, operator of the reprocessing facility. Its activities are supervised by the safety authorities reporting to the Ministries of Industry, Health and Prevention of Technological and Major Natural Disasters, particularly the Central Service for the Safety of Nuclear Installations (CSNI). ANDRA is not currently responsible for managing all radioactive wastes in France, in particular not for those at reprocessing facilities or defence-related material, although in 1999 a report by a member of a parliamentary advisory group recommended that such authority be given to it as soon as possible.

CSNI receives technical backing from a special group of experts as well as from the Institute for Nuclear Safety and Protection (IPSN). Ministers submit regular reports on waste management studies to the Higher Council for Nuclear Safety and Information, which comprises leading scientists, members of industry, trade unions, spokesmen for environmental protection movements and journalists. It is planned to establish a more independent regulatory body, but the required legislation has not yet been submitted to parliament.

8.4.3 Management of nuclear waste

Radioactive waste in France is divided into 2 categories, short-lived (A-wastes) and long-lived, depending on the length of time it remains a hazard. Long-lived wastes are also referred to as B-wastes (equivalent to long-lived ILW in other countries), C-wastes (equivalent to HLW) and spent fuel, most of which is reprocessed.

8.4.3.1 LLW and short-lived ILW (A-wastes)

These wastes are disposed of in an operational near-surface facility in north-eastern France.

8.4.3.2 Spent fuel and/or HLW (B and C-wastes)

Currently, it is intended that the spent fuel will be reprocessed, and the resulting long-lived low and medium level wastes ('B-wastes'), high level vitrified fission products ('C-wastes'), and any spent fuel not reprocessed are all to be disposed of in a deep repository, following on-site storage. However, in 1998, an unpublished report recommended to government that any future strategy must acknowledge the fact that as much as a third of future spent fuel in France is unlikely to be reprocessed as originally intended. It also suggested that France should immediately seek to repatriate some of its stockpile of plutonium recovered from reprocessing foreign fuel.

As far as long-term interim storage of spent fuel is concerned, the site for an ICE (Installation Central d'Entreposage) has yet to be selected, although it is likely to be a modular pool storage facility.

For development of a deep repository, four areas were originally selected for examination in the mid-1980's, in clay, granite, schist and salt. Four years of investigation were originally planned between 1987–1990. One site was then to be chosen, an underground laboratory developed and further work carried out between 1991–1995 with construction planned to begin in 1997. However, there was public opposition at all of the proposed sites, which stopped all work from proceeding. An amended Waste Law was passed in December 1991, under which ANDRA became a public service company regulated by the Ministries of Environment, Industry and Research and was removed from organisational contact with the Atomic Energy Commission. This was to demonstrate independence and increase transparency and openness.

The Law, No. 91-1381, laid out in clear terms major areas of research which were to be carried out by ANDRA. These were:

- Partition and Transmutation;
- Waste packaging and the effects of long-term surface storage;
- Development of at least two underground laboratories, in different geological media, but only after close local consultation and public involvement.

The Law specified that initial designation of a site as a candidate for a laboratory would require a public inquiry and authorisation from the Government. It would then be impossible for that site to be proposed as a potential repository site for at least 15 years after the date of the Law, without an inquiry and enabling legislation. It was stipulated that ministers would present regular reports on progress to parliament, and that ANDRA would submit a final status report in 2005, to be followed by recommendations on siting a repository in 2006.

A further amendment to the Law was announced in early 1998 to allow funding for a 4th research strand, concerned with the examination of long-term storage possibilities.

In order to monitor the progress of the research work in these areas, and to report to parliament, the Law established the National Evaluation Commission (CNE), which holds regular hearings on the main topics, and where ANDRA and others make presentations as requested. Reports are submitted to government on an annual basis, and reviewed by the Parliamentary Commission on the Assessment of Scientific and Technological Choices (OPECST). CNE is also responsible for the organisation and submission of the overall repository project report due in 2005.

CNE consists of 12 people: Six suitably qualified experts, nominated by OPECST, at least two of whom are from abroad (currently from Sweden and Spain), two experts nominated by government and four experts nominated by the French Academy of Sciences

CNE issued a report in 1998 that recommended major changes to the French disposal strategy, including separation of the large volumes of non heat-generating 'B ' wastes from HLW. They would be placed in a deep repository, as they are regarded as wastes with no likelihood of ever being of future use. The HLW, on the other hand, following around 50 years of surface storage, possibly in liquid, not vitrified, form, during which time they would cool down, could go into 'subsurface storage' with continuous monitoring and assured retrievability. Both these periods of storage are intended to allow for the possibility of new technologies being developed in the future which might enable their re-use. Spent fuel could be stored in retrievable form in either surface or near-surface facilities.

CEA began work on the so-called 'Very Long-Term Interim Storage' Project (ETLD), in 1998. The intention is to examine a number of different concepts, from near-surface, as at CLAB in

Sweden, to deeper, drift-accessed facilities as proposed in Switzerland. Even dry storage on the surface at existing NPP's will not initially be precluded.

As many as 10 concepts will be subjected to detailed technical and economic studies between 2002-2006, to enable a full comparison between them and the original deep disposal concept to be made on the same timetable as laid down in the 1991 Law. CEA is also to research the feasibility of developing a storage facility in subsurface caverns in marl in the Gard region, although this area was excluded from development of a deep laboratory, as described below.

The 1991 Law established the new post of Mediator, to facilitate site selection and development for the underground laboratories. Christian Bataille, Member of the Parliament, was appointed to this position in 1992, and empowered to offer financial compensation – in accordance with what was offered communities accepting industrial plants –, up to about FF60 million per year, to communities which offered themselves for further investigations. He was to consult with elected officials, local populations and environmental protection organisations, and in December 1993 he submitted a report recommending four areas for further investigation, three in sedimentary formations and one in crystalline. Subsequently, in 1994, ANDRA announced that a number of potential sites had been identified, one of which straddled the border between two of the four recommended areas, and detailed site investigations began that year. 15 boreholes were drilled in total, up to 1 100 m deep, at three sites.

Following the drilling, public inquiries were held between February and May 1997. In December 1998 the government authorised ANDRA to develop an underground laboratory in clay beneath a site at Bure in northeast France, but rejected another clay (marl) site near Marcoule in the Gard and a granite site in the Vienne as being geologically unsuitable. A decree licensing construction and operation at Bure was issued in August 1999, valid until 2006. At the same time, the government instructed ANDRA to find more potential sites in granite by the end of 2002. Despite examining approximately 20 granite areas in Brittany and the Massif Central, the project was actually suspended in June 2000, not least because of local opposition at all of the proposed sites.

Excavation of the first shaft at Bure began in early September 2000. A number of geotechnical, hydrogeological and other boreholes have been drilled and instrumented so as to allow study of the impact on the rock of the shaft sinking process. Various geophysical surveys will be carried out as work proceeds and be correlated with surface measurements taken during late-1999. A number of investigation niches will be established at various levels in the overlying rocks as the shaft is developed, as well as in the host Callovian-Oxfordian age clay at potential repository depth.

8.5 Germany

8.5.1 Nuclear power programme

As of April 2000, there were 19 reactors in operation, none of which are in the former German Democratic Republic, following the closure in 1990 of the Rheinsberg nuclear power plant, of four operating reactors at Greifswald and a fifth unit still being in the commissioning phase.

The Social Democrats/Green Coalition Agreement of October 1998 proposed a phase-out of the use of nuclear energy for electricity production in Germany. After lengthy discussions, the government and the utilities finally initialled an agreement on nuclear policy on 14^{th} June 2000, which stipulates that all existing reactors will be phased out at the end of their operating lives. Each will be allocated a maximum amount of electricity which can be generated, thus allowing capacity to be added to newer, more efficient reactors, thereby extending their operation, and allowing closure of the less efficient. The amount of electric power agreed on, roughly corresponds to an operational lifetime of 32 years. No new reprocessing contracts will be allowed, and after the 1st July 2005 all spent fuel will follow the direct disposal route. Only reprocessing contracts running up to that time will be honoured.

8.5.2 Relevant institutions

When the Federal Office for Radiation Protection (BfS) was established in 1989, it took over the responsibility for the safe disposal of all types of radioactive waste from the Federal Institute for Science and Technology (PTB). The German Company for the Construction and Operation of Waste Repositories (DBE) was set up as ”Third Party” (contractor) to carry out the tasks assigned to it by the BfS. The BfS is under the jurisdiction of the Federal Ministry of the Environment, Nature Protection and Nuclear Safety (BMU). Fundamental research work is conducted in conjunction with the Federal Ministry for Research and Technology (BMFT) and the Federal Ministry of Economics (BMWi).

A new Radiation Protection Ordinance (StrlSchV) has been drafted by BMU and is currently being examined. This is designed to bring the German regulations in line with EU Directives, and lowers the permitted worker dose rates from 50 mSv/year to 20 mSv/year, with one or two exceptions.

According to the Atomic Energy Act, the State government carried out licensing. Originally, all spent fuel was to be reprocessed; this was amended in 1994 to allow direct disposal of spent fuel to take place as well. Some utilities have already cancelled reprocessing options after 2000.

8.5.3 Management of nuclear waste

As all types of waste are planned to be disposed of in a deep repository, wastes are basically separated into two categories; heat-generating and non heat-generating. According to the 1998 Coalition Agreement, a single repository in deep geological formations is sufficient for the disposal of all types of radioactive waste, in a rock type yet to be decided and at a site yet to be formally identified. This will naturally have serious implications for the repository development programmes already underway.

8.5.3.1 LLW and ILW (non heat-generating)

Until recently non heat-generating wastes with alpha emitter concentrations up to 4.0 x 10^8 Bq/m^3 were disposed of in the ERAM facility (Endlager für Radioaktive Abfälle Morsleben) which began operation at the former Bartensleben salt mine in 1971, and was last licensed in 1986, by the authorities in the former GDR. According to a further amendment of the Atomic Energy Act following the German unification, the licence was due to have expired on 30 June 2000. According to the September 25, 1998 order of the Superior Administrative Court of the state of Saxony-Anhalt , BfS had to immediately stop further radioactive waste disposal in the so-called eastern emplacement field of the Morsleben repository. Due to the results of the BfS examination of the court order (preliminary decision), it was decided to stop waste emplacement in this repository. At present (December 2000), BfS activities are concentrated on the licensing procedure for the closure of ERAM, with focus in particular on the further development of the back-filling and sealing concept.

The licence application for a deep repository for non heat-generating LLW and ILW at the former Schacht Konrad iron-ore mine near Salzgitter, in Lower Saxony, was submitted as long ago as 1982. After the longest Public Inquiry in German history between September 1992 and March 1993, the Lower Saxony Government (headed at the time by the present Federal Chancellor) continued to refuse to grant a licence for the facility, against the wishes of the Federal Authorities. Following the June 2000 Agreement, the responsible authorities shall conclude the licensing procedure for Schacht Konrad according to the legal provisions. BfS withdrew the application for the immediate enforcement of the licence in order to allow a court examination on the merits of the main proceedings.

8.5.3.2 Spent fuel and/or HLW (heat-generating)

Previous to the 1994 amendment of the Atomic Energy Act, the only route for spent fuel was reprocessing, which took place either in France or the UK. Plans to develop a German reprocessing plant at Wackersdorf were abandoned in 1989, because of intense, often violent, opposition.

Repatriation of existing vitrified HLW began in May 1996, following the licensing of the interim storage facility at Gorleben in Lower Saxony in early June 1995. A second shipment, in early March 1997, involved thousands of police in a massive security exercise. In addition, an interim storage facility for spent fuel was built at Ahaus, near the Dutch border. Even shipments to that from German reactors were met with violent demonstrations. Under the 1998 Coalition Agreement and the June 2000 Agreement, shipments will be reduced drastically according to the new on- or near-site interim storage facilities.

Up until recently, it was always proposed that for HLW (and possibly spent fuel also), the plan in Germany would be to develop a deep repository in a suitable salt dome. However, under the June 2000 Agreement, the whole disposal policy is open to re-examination. Although deep disposal remains the preferred method, other rock types are to be examined before any decision on siting is made.

Originally, the Gorleben salt dome was selected in 1979 as the sole candidate site for the disposal of all types of radioactive waste. Investigations were scheduled to be completed by 1997, but, as delays have occurred, were not expected to be finished before 2003. Under the June 2000 Agreement, all underground investigations at Gorleben ceased at the end of September 2000, although the emplaced monitoring systems will continue to function and be serviced. No new work will begin for at least two to three years, up to a maximum of 10 years.

As it became clear that examination of other potential repository sites in other rock types would be required, BMU established the AKEND Committee in February 1999 to develop a new siting procedure. The committee is working to a schedule designed to report its findings and recommendations to the government by 2002. It envisages a 3-phase programme for the implementation of a new site selection procedure. This includes development of a site selection procedure and corresponding criteria, the political and legal establishment of the procedure and its implementation.

Initially, work is being carried out to agree and apply a series of siting criteria and draw up a list of potentially suitable areas. It will move on to nomination of sites over the subsequent 12–18 months. Public participation has been identified as being absolutely crucial to the success of the AKEND Committee, and a specific sub-committee has been established to address the issue. Details of the committee's activities will be made available to the public via a dedicated website, and through a series of annual workshops, the first of which was held in Kassel, in mid-September 2000.

8.6 Japan

8.6.1 Nuclear power programme

A total of 51 commercial reactors are currently in operation, run by the Japan Atomic Power Company and 9 other independent electricity companies. The prototype fast breeder reactor, called Monju, is currently shut down, after an accident involving loss of the sodium coolant in December 1995. Four new reactors are presently (November 2000) under construction, one of them at a greenfield site. Another four reactors are being planned to construction.

8.6.2 Relevant institutions

The Atomic Energy Commission (AEC) and the Nuclear Safety Commission (NSC) determine basic guidelines on radioactive waste management. The AEC is responsible for the planning and determination of basic policy, whilst the NSC is responsible for safety criteria and regulations.

The Ministry of Economy, Trade and Industry (METI) and the Ministry of Education, Culture, Sports, Science and Technology (MEXT) implement licences for waste management and burial based on the ”Law for Regulation of Nuclear Source Material, Nuclear Fuel Material and Reactors”, although a new ”Special Radioactive Waste Final Disposal Act” to deal with HLW disposal was approved by the Cabinet in March 2000, and passed in the Japanese parliament (the Diet) in May 2000. It includes a requirement for the establishment of long-term disposal plan every five years, with a complete re-evaluation after ten years. *(As a result of a major reorganisation in January 2001, the Ministry of International Trade and Industry (MITI) and the Science and Technology Agency (STA) were reorganised to METI and MEXT, respectively.)*

The "Japan Nuclear Cycle Development Institution" (JNC) is responsible for work on advanced reactor designs, fuel cycle technology and R&D associated with HLW disposal. This organisation replaced the larger Power Reactor and Nuclear Fuel Development Corporation (PNC) in 1998, which was restructured after a number of incidents at various of its sites.

8.6.3 Management of nuclear waste

Current Japanese policy includes reprocessing of spent fuel and utilisation of the plutonium and enriched uranium, including the development of a MOX (Mixed-Oxide) fuel fabrication capability. In the past, spent fuel has been reprocessed abroad, although an experimental reprocessing facility was in operation at PNC's Tokai Works until March 1997, when there was an explosion and fire. It was restarted in November 2000.

A commercial-scale reprocessing facility has been under construction since 1993 at Rokkasho, in Aomori Prefecture, which is also the site of an operational LLW repository, an enrichment plant and a storage facility for returned HLW. Japan Nuclear Fuel Service Co. Ltd (JNFL) manages both of these facilities.

8.6.3.1 LLW and short-lived ILW

These wastes are disposed of in a near-surface repository at Rokkasho in Aomori Prefecture, which began operations in December 1992. The repository was co-located with the planned spent fuel reprocessing facility, currently scheduled to begin operation in 2005.

8.6.3.2 Spent fuel and/or HLW

The tentative target for commissioning of a disposal site in Japan, given in the 1994 ”Long-Term Programme for Development and Utilisation of Nuclear Energy” produced by AEC, was sometime in the 2030's or by around 2045 at the latest, taking into account the need for a facility in terms of development of the Japanese nuclear program by that time. This overall timeframe is repeated in a recent generic safety assessment made by JNC, and in the draft of a new Long Term Programme, which is currently under discussion.

The 1994 Long Term Programme restated an earlier intention to establish a ”disposal entity” around the year 2000, which would carry out the actual disposal program. In accordance with this, and as allowed for in the new Waste Act, the Federation of Electric Power Companies applied to government in early October 2000 for permission to establish this body. Government approval was immediately forthcoming, and the Nuclear Waste Management Organisation (NUMO) came into being in October 2000, based in Tokyo.

It is understood that a number of sites will be investigated, beginning in 2001, that preliminary survey sites will be selected in 2004, and that a few detailed survey sites will be selected around 2010. The disposal site is expected to be determined around 2025.

In August 1989 it was decided to build the ”Deep Experimental Facility” at the disused Kamaishi iron/copper mine, in Iwate Prefecture, despite strong local opposition which delayed the beginning of the project. Phase 2 of the project ran until 1998, intended to develop an understanding of groundwater movement in the area, as well as to allow study of the effects of earthquakes on an underground facility. Operations were terminated in March 1998 when the agreement with the local Municipality expired.

An experimental shaft, some 150 m deep, has also been in operation since 1986 in the Tono area in Gifu Prefecture, central

Japan, in an undeveloped uranium deposit in sandstone overlying crystalline granitic rocks.

A new underground facility was approved for development in the same area, at Mizunami, in December 1995. Surface-based investigations began in late 1997 and are scheduled to last for up to five to six years. The construction of the underground laboratory, including shaft sinking, will begin by the end of 2002, and take at least eight years. Experimental levels will be developed in granite at around 1 000 m depth, and the facility is intended to be operational for up to 20 years. It will take over from Kamaishi as the main crystalline research site, and, like it, has been designated as a ”research-only” facility.

After many years of discussions between JNC, Hokkaido Prefecture and Horonobe town, the three parties finally, in November 2000, concluded an agreement on the underground research at Honorobe under the condition that radioactive material should not be used. Detailed research programme is being prepared and the boring investigation will begin soon. Honorobe underground research laboratory is envisaged to be a centre of sedimentary rock research, while Mizunami is for granite.

8.7 Russia

8.7.1 Nuclear power programme

As of end-1999, there were nine nuclear power plants in operation in Russia, with a total of 29 reactors, of which 15 are of RBMK type, 13 are VVER's, and one is a fast breeder. Four reactors have been decommissioned. As many as 75 research reactors have also been in operation, some of which are closed down. In addition to these NPP’s, there are also numerous facilities involved in uranium mining, fuel fabrication, reprocessing, isotope production etc., as well as large defence complexes associated with plutonium production and nuclearpropulsion operations of the Northern Fleet in the Kola Peninsula and the Pacific Fleet around Vladivostok. There is a commercial reprocessing plant at Chelyabinsk (now referred to as Ozersk). Another was under construction at Krasnoyarsk (now referred to as Zheleznogorsk), but work has now been abandoned.

8.7.2 Relevant institutions

In the past, responsibility for radioactive waste was split between four different ministries:

- The Ministry of Atomic Power (Minatom) had the responsibility for wastes from civilian nuclear power and from the production of nuclear weapons. It was founded in 1992. There are approximately 150 enterprises associated with Minatom, including 15 so-called ”closed cities”, where a total of 13 plutonium producing reactors have been operated, three of which are still running. Rosenergoatom is responsible to Minatom for operation of all nuclear power plants, and management of the associated wastes;
- The Ministry of Defence had the responsibility for wastes from nuclear-powered naval ships;
- The Ministry of Marine Transports was responsible for wastes from nuclear-powered icebreakers;
- The Ministry of Construction and Housing Policies which is managing the special facilities at ”Radon” (for treatment and disposal of low and intermediate radioactive waste) was responsible for the management of radioactive waste generated in industry, medicine, research, etc.

In 1993, a presidential decree gave Minatom the role of coordinating waste management activities in all four ministries, and in May 1998 Minatom announced that it would also be taking over responsibility for management and treatment of decommissioned submarines and associated naval wastes. Plans were announced in November 1998 to form a new company to promote nuclear energy and to improve revenue from it. It is to be known as ”Atomprom”, and will control all financial and operational aspects of all nuclear reactors in Russia. These plans were repeated in July 2000.

The regulatory body in Russia is the Federal Nuclear and Radiation Safety Authority of Russia (Gosatomnadzor of Russia). It became responsible for licensing and inspection of all nuclear facilities, including military, under the law *”On the Use of Nuclear Energy”*, adopted in November 1995. Following on from this law, those organisations involved in nuclear waste production and management were required to submit new operating licence applications, some of which are still under consideration.

8.7.3 Management of nuclear waste

8.7.3.1 LLW and short-lived ILW

It has been proposed to develop a repository for military LLW in permafrost in northern Russia and a deep repository for industrial (non-power reactor) wastes near to Moscow, in salt or clay. There are currently no sites being sought for disposal of reactor LLW and ILW, which is stored on site at the present time.

8.7.3.2 Spent fuel and/or HLW

Russia originally planned to reprocess only spent fuel from certain reactor types, namely the VVER-440, VVER-1000, BN-350 and BN-600. There are no plans to reprocess RBMK fuel. VVER-440 fuel is reprocessed in the RT-1 plant, operated by the Mayak combine, at Ozersk, in the Southern Urals, which began operations in 1948 on military fuels, and was modified in 1976 to reprocess civilian fuel. The construction of the RT-2 plant at Zheleznogorsk to reprocess VVER-1000 fuel was suspended in 1989, and finally abandoned in 1998, reportedly for both financial and technical reasons. It has now been proposed to reconstruct the head-end facilities at Ozersk to accommodate VVER-1000 fuel. VVER fuel from NPP's had already started to be sent to Zheleznogorsk in the early 1990's, where it is stored in pools, whose capacity is expected to be exhausted by 2005.

RBMK fuel is stored for 3–5 years in pools in the reactor halls at the NPP's, and then in interim pools also at the NPP site. Such on-site interim stores currently exist at only the Leningrad and Kursk NPP's, but are under construction at Smolensk.

Other liquid wastes, including HLW, from various activities, have been injected into deep boreholes at Ozersk, Zheleznogorsk, Dimitrovgrad and Seversk for many years.

Development of a strategy for spent fuel and HLW management and disposal is now the responsibility of the Russian Institute of Geology of Ore Deposits, Petrography, Mineralogy and Geochemistry (IGEM). Responsibility for developing a more acceptable management system for the wastes from reprocessing at Zheleznogorsk (should the RT-2 plant have begun operation), was given to the Khlopin Radium Institute in St. Petersburg.

Several different deep disposal concepts are currently under investigation. As the authorities do not consider retrievability to be desirable, both mined cavity and deep borehole emplacement are thought possible.

In keeping with the centralised thinking of the past, and the wish to locate facilities near to the source of the waste involved, attention has been focused on the Zheleznogorsk and Ozersk sites themselves.

The Khlopin Radium Institute in St. Petersburg has studied sites around Zheleznogorsk, and work has also been carried out by other institutes looking at basalts and granite rocks of the Baltic Shield. Originally eight sites were under consideration, but latest results from work in 1996 have led to the identification of two candidates, of which one has been selected for further investigation (subject to funding). This work has been supported by the IAEA Expert Contact Group, and financed by PNC of Japan, the DOE in the US and the authorities in Finland.

Work at Ozersk has been funded by the former USSR Academy of Sciences. A site within the boundaries of the Complex was originally identified and four deep boreholes drilled to depths of at least 900 m. It is intended to develop an underground laboratory to conduct experiments and in-situ characterisation, although recent research suggests that it could prove difficult to locate a suitable final repository in the area, because of uncertainties regarding the tectonic stress conditions. Work on this project is being carried out as part of an EU-supported PHARE programme, and includes input from several western organisations. So far IGEM have identified three possible repository zones and cast doubt on the suitability of the original site.

The management and disposal of spent fuel and other wastes from the defence-related industry, in particular the large accumulations from nuclear submarines, has also become a major, pressing problem. Much of this waste, in the form of spent fuel and various liquids, is stored in unsatisfactory conditions at either the Northern Fleet bases on the Kola Peninsula around Murmansk and Archangelsk, or in the Pacific Fleet bases around Vladivostok. Up to 48,000 spent fuel assemblies are thought to be in storage in the Northern Fleet bases, in facilities seen to be leaking and in poor condition.

Many nuclear submarines are laid up, as many as nine with intact reactor cores. Although removal of this only restarted in summer 1996, after a two year break, funded from the State budget, this was insufficient, and waste is beginning to accumulate again, some of it in the open. It is estimated that at least 100 other submarines will require decommissioning in the near future.

An IAEA working group, the Contact Expert Group, reported in February 1998 that the NW region of Russia should be targeted as a priority area for global co-operative projects, so poor is the state of waste management there.

Three options have been examined: a new wet store, a new dry storage facility or renovation of the existing wet stores. For a dry store, estimated costs of $50 million would be borne by Sweden, Norway, France and Russia, according to an agreement signed in February 1998, in addition to EU support, which was confirmed as forthcoming in May 1998.

The US announced in July 1998 that it would fund shipments of spent fuel from Vladivostok to Ozersk, because of fears over the safety of the existing storage facilities, and following a visit to Murmansk in 1999 by the UK Defence Secretary, £5 million was made available immediately to help with the most dangerous problems in the area. Following the accident with the Kursk nuclear submarine this has been increased to around £80 million.

The Kola Mining Institute has been involved in numerous studies to develop underground storage facilities for the Northern Fleet HLW, with a proposal being made as long ago as 1994 describing a four year programme for a deep repository in Kola, along conventional lines, in hard crystalline rocks. It would be preceded by an experimental facility. Little work actually appears to have been carried out to date, however.

It emerged in April 1999 that a US company, ”Non-Proliferation Trust Inc.” (NPT) has been established to pursue development of an international spent fuel storage facility at Zheleznogorsk. This would have a capacity of around 6 000 metric tonnes of uranium, and a lifetime of at least 40 years. The money generated would be used to help cleanup at Russia’s defence facilities, in the safe management of up to 50 tonnes of plutonium and to support the repository projects currently in progress. However, for this project to take place, Russian law would have to be amended to allow for the import of foreign waste. A set of three laws, which in future will allow for the import of foreign spent fuel, passed the first hearing in the State Duma on 21 December 2000:

- On the introduction of changes to the Federal Law on the Environmental Protection (Article 50);
- On the introduction of changes to the Federal Law on the Use of Nuclear Energy (Article 64);
- On Special Ecological Programmes for Rehabilitation (remediation) of the Radioactively Contaminated Regions of the Russian Federation funded by earnings from the foreign-trade operations with irradiated nuclear fuel.

The constitutional majority of the Deputies voted for these drafts.

8.8 Switzerland

8.8.1 Nuclear power programme

There are currently four commercial reactor sites in operation in Switzerland, with a total of five reactors, as well as six research reactors. There is presently a moratorium on new reactors ending in 2000, although this may change when a revised Atomic Law is passed sometime in 2003.

8.8.2 Relevant institutions

In Switzerland, the producers of nuclear waste are responsible for waste management. The power supply companies and the Swiss Confederation – responsible for wastes from medicine, industry and research – joined together in 1972 to form NAGRA (the National Co-operative for the Disposal of Radioactive Waste). NAGRA is responsible for research and preparatory work for final disposal. For the implementation of particular projects, site-based construction and operation companies are established, e.g. ZWILAG Würenlingen for interim storage, and GNW (Co-operative for Nuclear Waste Management Wellenberg) for the LLW/ILW repository. The responsibility for spent fuel reprocessing and transport and for waste conditioning and interim storage at the reactor sites remains with the utilities.

The Federal Government is supported in its decisions on waste management by the Federal Interagency Working Group on Nuclear Waste Management (AGNEB), by the Federal Commission for Safety in Nuclear Installations (KSA) and by the Federal Commission on Nuclear Waste Management (KNE), itself a sub-committee of the Federal Geology Commission (EGK).

The regulatory body in Switzerland is the Swiss Federal Nuclear Safety Inspectorate (HSK) of the Federal Office of Energy (BFE), itself part of the Federal Department of the Environment, Transport, Energy and Communication (UVEK).

Because of the various delays in obtaining public acceptance of repository siting activities – especially with regard to the Wellenberg repository – several working groups have been established by the Federal Government over the last five years. Besides discussions concerning the continuation of work at the Wellenberg site, these groups have investigated different waste management concepts. With regard to Wellenberg, there was a consensus – all groups recommended continuing the work with an exploratory drift. However, the dilemma between ”indefinitely monitored retrievable storage” and ”passively safe geological disposal” failed to reach a satisfactory solution. In view of this, the Federal Government decided to set up the Expert Group an Disposal Concepts for Radioactive Waste (EKRA) in June 1999. The group developed the concept of ”monitored retrievable long-term storage”.

EKRA came to the conclusion that geological disposal is the only method for isolating radioactive waste which fulfils the requirement for long-term safety. However, social demands oriented towards the principle of reversibility must be taken into account. Hence, a stepwise approach is envisaged, with a phase of monitoring and facilitated waste retrieval prior to closure of the geological repository. In addition to the main waste emplacement facility (the actual repository), the concept foresees a so-called ”pilot facility”, in which part of the waste is emplaced in a small but representative ”copy” (also in terms of waste inventory) of the main repository that can be instrumented for long-term observation. The architecture of the main facility foresees the possibility of retrieving the waste from a closed repository if the findings in the pilot facility would so require. Of course, this idea of ”monitored long-term geological disposal” must be adapted to the site geology and the waste types under consideration for a particular repository.

8.8.3 Management of nuclear waste

Prior to the construction of a final repository, all types of waste will be stored in the ZWILAG facility at Würenlingen, Canton Aargau, Northern Switzerland, which opened in April 2000.

8.8.3.1 LLW and short-lived ILW

Because of the high population density in Switzerland, no shallow land burial is envisaged, even for LLW or short-lived ILW. It is currently planned to dispose of these wastes in caverns with access through a horizontal tunnel, in a suitable rock formation, at a depth of several hundred metres. After an extended site selection procedure, NAGRA in 1993 identified Wellenberg in Canton Nidwalden, Central Switzerland, as a potential site. In 1994 the site-based company GNW was established. The local community accepted the project in two votes in 1994 by 63 and 70 % yes. Despite this, a cantonal referendum did not approve the mining concession, as required by the Law of Canton Nidwalden. The two most important objections to the project were

- that a mining license for the exploratory drift and the subsequent construction of the repository had been applied for at the same time (”license in principle”) and
- that a ”classical” geological repository had been envisaged without special provisions for retrievability and post closure in-situ monitoring.

Since then, the geological suitability of the site has been re-evaluated and confirmed by the Federal Safety Inspectorate and GNW has decided to restrict its licence application in the first step and to modify the repository concept to include a prolonged phase of monitoring and a step-wise approach towards the closure of the repository. In view of this, the Federal Government entered into a discussion with the Cantonal Government yielding an agreement in June 2000 along the following lines:

- GNW will modify its repository concept to meet recommendations of EKRA;
- GNW will clearly state that no HLW, spent fuel or transuranic waste will be disposed of in Wellenberg;
- GNW must accept pre-defined exclusion criteria for the geological suitability of the site.

A Cantonal Expert Group (KFW) has been established to prepare and later supervise the project.

The KFW Group started its work in July 2000. After several consultations with GNW, with NAGRA (acting as the scientific and technological competence centre for GNW) and with the Safety Inspectorate (HSK), the necessary project modifications have been agreed upon and documented in a report by GNW, which was submitted in November 2000. In December 2000, the KFW judged the report to be sufficient and the Cantonal Government declared to be willing to accept an application from GNW for mining licence, restricted to the exploratory drift. The application will be submitted to the Government by the end of

January 2001. If the review will be positive, the application will be submitted to popular vote in autumn of 2001.

8.8.3.2 Spent fuel and/or HLW

For about one-third of their spent fuel, the Swiss utilities have reprocessing contracts with France and the U.K.; for the remaining two-thirds the utilities wish to keep their options open. However, in its current version, the draft of the new Atomic Law, which is expected to enter into force after a public referendum around 2003, does not allow for further reprocessing beyond the existing contracts. Vitrified HLW will be returned to Switzerland for interim storage in the ZWILAG and ZWIBEZ (storage facility at the site of the Beznau reactor site). The first shipment from France is expected to take place in 2001. Spent fuel will also be stored at the aforementioned facilities prior to disposal.

Swiss law requires the permanent safe disposal of radioactive wastes in geological repositories. As a prerequisite to the continued operation of existing nuclear power plants or construction of any new plants, a Government ruling of 1979 called for a project demonstrating the feasibility of safe disposal of all wastes arising in Switzerland to be submitted by 1985. This project – ”Project Gewähr” – was submitted to the Federal Government by NAGRA in January 1985.

In June 1988, the project, based on crystalline host rock, was approved by the Government. Although the safety case and the technical feasibility of repository construction were fully accepted by the safety authorities, the question of the existence of a sufficiently extensive body of host rock with the required properties for making the safety case was not answered to their complete satisfaction. As Project Gewähr was based only on a crystalline host rock option, the authorities requested that future work be extended to cover sedimentary options.

NAGRA follows a three-phase repository siting strategy. Phase 1 covers regional studies based on a series of deep boreholes and associated regional geophysical surveys. Phase 2 involves detailed characterisation of smaller areas from the surface and, at a later stage, phase 3 would include underground exploration.

Crystalline basement option

The regional field work (phase 1) on the crystalline basement finished in 1989 and the corresponding synthesis reports were completed in 1994. The main reports include a comprehensive geological synthesis, a performance assessment study and a summary report. These documents are still under review by the safety authorities, who are expected to submit their evaluation by mid 2001.

At the end of 1994, NAGRA applied for federal permits for two site investigation programmes, one for the Opalinus Clay in the Zürcher Weinland and one for crystalline basement in the Böttstein/Leuggern area. The programmes were reviewed by the federal authorities and their experts. For the crystalline basement, subsequent discussions in a working group including representatives of the federal authorities, their experts and NAGRA led to a modified programme. It was recommended to focus on a different area west of Leuggern, namely the Mettau Valley, for continuing exploration of the crystalline basement. Accordingly, a local two dimensional seismic curve (as part of a phase 2 programme) was carried out in the new area during winter 1996/97. The survey marks the end of field work in the crystalline programme for the time being.

The underground laboratory in the crystalline rock of Central Switzerland – the Grimsel Test Site – has been in operation since 1983. To construct the laboratory, a horizontal tunnel system was excavated from an existing access tunnel to a hydro-electric facility on the Grimsel Pass. Since 1984, an extensive test programme, covering hydrogeology, geophysics, rock mechanics, etc. has been underway, with wide international participation.

Opalinus Clay option

The Opalinus Clay (OPA) had been considered as a potential host formation prior to Project Gewähr, in 1979. Desk studies carried out in 1986/7 had also evaluated six other potential sedimentary formations and the options were narrowed down to two final candidates, namely the OPA and the Lower Freshwater Molasse (USM). The latter can reach a thickness of up to 4 km and contains units of high clay content and low permeability.

Two preferred study areas were identified for OPA, which, like the crystalline areas, are located in the north of Switzerland. As part of the Phase 1 programme, a regional two dimensional seismic study, extending over 230 km, was conducted in 1991/92. The seismic data were calibrated on the basis of the existing NAGRA boreholes and oil exploration wells. This work confirmed a regional thickness of the clay of between 100 and 120 metres. Further studies were also carried out for the USM. Based on these investigations, an interim evaluation of the sedimentary options was undertaken together with the authorities in 1994. The USM was assigned second priority and is regarded as a reserve option. First priority was assigned to the eastern area for the OPA. After a further selection process in this area, the region of the Zürcher Weinland (Canton Zürich) was identified for further investigations.

These further investigations (Phase 2) consisted of a three dimensional seismic survey over an area of around 50 km^2 and a deep borehole at the Benken site. In the Zürcher Weinland, the sedimentary rocks are almost horizontally bedded and the Opalinus Clay has a sufficient thickness of 100–120 metres at an appropriate design depth of 400 to 900 metres below ground surface. Since the deposition of the sediment, the region has experienced almost no tectonic movement or disruption of the original sedimentary layers, making it an ideal candidate site.

Another important source of information on the in situ properties of the Opalinus Clay, and clay formations in general, is the work being carried out at the Mont Terri Rock Laboratory (Canton Jura) within the framework of an international project under the patronage of the Swiss National Hydrological and Geological Survey. The facilities are located near a motorway reconnaissance tunnel traversing the Opalinus Clay formation at a depth of around 300 metres.

Next milestone in the Swiss HLW programme

The next milestone in the Swiss HLW programme will be finalisation of the demonstration of feasibility of HLW disposal in Switzerland (Project ”Entsorgungsnachweis”). This will include demonstration of the existence of a sufficiently extensive body of host rock with the required properties for making the safety case (as required by the authorities in the evaluation of Project Gewähr, see above), together with demonstrations of safety and engineering feasibility. Because of its excellent explorability from the surface and the positive results of the investigations to date, this will be done only for the Opalinus Clay. This does not mean, however, that, for an eventual construction of a Swiss HLW repository, the crystalline option will be abandoned.

The key reports and supporting documentation for Project ”Entsorgungsnachweis” will be submitted for evaluation by the safety authorities at the end of 2002. A decision by the authorities on how to proceed further is not expected before 2005.

8.9 United Kingdom

8.9.1 Nuclear power programme

The UK currently operates 19 Magnox Reactors, 14 AGR's and one PWR. The AGR and PWR reactors are operated by British Energy Generation, comprising the previously State-owned Nuclear Electric and Scottish Nuclear Corporation, which merged in January 1999. The Magnox stations are still State-owned, and are operated by Magnox Electric, itself taken over by British Nuclear Fuels Ltd (BNFL) in 1998. It was announced in July 1999 that BNFL is itself to be partially privatised sometime in the future.

8.9.2 Relevant institutions

The regulatory authority in the UK is the Nuclear Installations Inspectorate (NII), assisted by the Environment Agency (EA) and the Ministry of Agriculture, Fisheries and Food. Since July 1997 NII has also had responsibility for regulating wastes held on sites operated by the Ministry of Defence. In Scotland, the EA's responsibilities are discharged by the Scottish Environmental Protection Agency (SEPA).

The government is advised on waste management issues by the Radioactive Waste Management Advisory Committee (RWMAC), whose members are appointed by a minister. These come from the nuclear industry, academia, public bodies (health authorities etc.), and, more recently, a number of independent experts have been appointed.

There is a large commercial reprocessing facility at Sellafield, in Cumbria, operated by BNFL. A smaller facility exists at Dounreay, in northern Scotland (also the site of the Experimental Fast Breeder Reactor, now shut down), operated by the United Kingdom Atomic Energy Authority (UKAEA), built to reprocess specialist fuels and Highly Enriched Uranium from research reactors.

AGR and Magnox spent fuel is currently stored on-site in pools at the NPP's, for a cooling-off period, as will be the fuel from the PWR at Sizewell, before being shipped to Sellafield for long-term storage and eventual reprocessing. Dry storage of Magnox fuel has been undertaken at only one NPP, but construction problems there resulted in corrosion of the fuel canisters.

In 1994, the Thermal Oxide Reprocessing Plant (THORP) began operation, intended to reprocess around 7000 tonnes of spent oxide fuel (from AGR's, PWR's, LWR's etc.) by 2005. The problems encountered by BNFL in Japan, caused by poor quality control in the manufacture of mixed-oxide fuel (MOX), have however cast doubt on some future reprocessing contracts.

8.9.3 Management of nuclear waste

8.9.3.1 LLW and ILW

Whilst short-lived LLW is the responsibility of the waste generator, as is currently the case for HLW, the responsibility for disposal of long-lived ILW and future LLW and short-lived ILW arisings has lain with the Nuclear Industry Radioactive Waste Management Executive, now known as UK Nirex, since 1982. This was formed by all the members of the nuclear industry in 1981, each with a seat on the board. HLW has never been part of Nirex's responsibility.

A commercial shallow land disposal site for LLW and short-lived ILW has been operated by BNFL at Drigg, in Cumbria, adjacent to its Sellafield site, since the 1960's and it was originally proposed by Nirex to continue shallow disposal for these wastes, when this was full, at a replacement site and to carry out deep disposal of long-lived ILW in a disused anhydrite mine. However, due to local opposition, the mine site was abandoned in 1985.

When in 1986 three further shallow LLW sites were added to the original single candidate, there was again local opposition involving widespread civil disobedience. These sites were abandoned in May 1987, just prior to a General Election after which deep disposal then became the preferred option for all LLW/ILW. This proposal was also soon amended, reverting to the currently preferred option; deep disposal for long-lived ILW, with LLW and short-lived ILW to go to Drigg.

Following a two-year nation-wide survey, two sites were chosen in 1991 for further investigation, both at existing nuclear complexes, at Sellafield and Dounreay. A further ten sites were also short-listed but not made public.

Sellafield became 'favourite ' in 1993, and over £250 million was spent in surface-based characterisation. The concept evolved from a simple repository, and in 1992 Nirex announced its intention to develop a Rock Characterisation Facility (RCF), in which to carry out limited experimentation and development, prior to construction of a full-scale repository. Nirex applied for permission to begin development of the RCF in 1994, but this was refused after an inquiry in 1995. In his report the Inspector said that Nirex had failed to convince him of the validity of their geological interpretation, and that the design was poor and ill conceived. Nirex immediately announced that they were withdrawing from Sellafield, although they reserve the right to return in the future.

In November 1997 the UK House of Lords Select Committee on Science and Technology (HoL) announced an independent wide-ranging inquiry into all nuclear waste management-related issues, including the future role of Nirex. Verbal evidence began in February 1998 and the Inquiry report was finally published in March 1999. It concentrated on the development of a so-called 'phased approach' to the management of wastes, especially of LLW/ILW, culminating in the development of at least one deep repository for long-lived ILW. The report also acknowledged the need for a surface facility as replacement for Drigg, sometime in the next 15–25 years.

The inquiry recommended the setting up of an independent 'Nuclear Waste Management Commission', which would develop a disposal strategy for all types of waste, involving extensive public consultation. It would then establish a new and transparent siting process, likely to involve the use of volunteerism, coupled with the introduction of community incentive packages. Once one or more sites had been identified, it would instruct a newly created 'Radioactive Waste Disposal Company' (replacing Nirex), to carry out the facility design, followed by site characterisation, repository construction and ultimately operation. All wastes should be emplaced in a retrievable form.

The Commission, which would subsume the role of RWMAC, would be appointed by government, and have its own professional staff and funding.

The report stressed the need for a rapid response by government to its proposals, so as to begin the long process of developing public acceptance, and the Department of Environment, Transport and the Regions (DETR) finally issued one on 25th October 1999. This welcomed the House of Lords report, and acknowledged the need expressed in it to engage in a wide-ranging consultation process prior to the adoption of a long-term strategy for radioactive waste management in the UK.

Following publication of a Green Paper (i.e. a proposed text of a new Bill, open for consultation), the DETR plans to consult widely on issues such as the need for a deep repository for long-lived LLW/ILW, the period of which monitoring and retrievability of emplaced wastes might be necessary, and the whole issue of repository siting. Other issues, such as a possible reclassification as waste of some of the plutonium currently in storage at Sellafield, and the options for a possible successor to the Drigg LLW disposal site, will also be examined. The consultation will also allow for discussion of issues outside existing policy, such as the management of defence-related wastes. The Green Paper, which will begin the consultation process, has yet to appear (Decemer 2000).

8.9.3.2 Spent fuel and/or HLW

Current plans envisage cooling of UK-derived HLW at Sellafield for between 50–100 years, at which time the government will make decisions concerning their disposal. In the past, the only certainty as regards disposal was that it would involve deep disposal, in a rock type yet to determined, at a site yet to be determined.

Up until 1981 a Disposal Programme was underway, involving boreholes and other research. Some limited exploration work was carried out in crystalline and sedimentary rocks in the late 1970's, with detailed studies at a site near Dounreay. This was abandoned following widespread public opposition, and only generic research is now carried out. Deep disposal as a concept was restated in a government White Paper on waste management in 1995, but no specific programme was discussed. A schedule for repository development was produced for government in 1999 but no specific work has yet been carried out.

8.10 United States

8.10.1 Nuclear power programme

The US currently operates some 103 reactors at over 80 different sites.

Because commercial reprocessing of spent fuel was halted in 1977, HLW from non-defence sources comprises only a small fraction of that currently awaiting a management solution. More than 95 % by volume originated in defence-related reprocessing activities under the jurisdiction of the DOE and is in tank storage at various DOE sites, awaiting vitrification. Two plants began operation in 1996, in South Carolina and New York State.

8.10.2 Relevant institutions

In the United States, disposal of nuclear waste is paid for by the producers. The disposal of commercial spent nuclear fuel and HLW is the responsibility of the Office of Civilian Radioactive Waste Management (OCRWM) within the U.S. Department of Energy (DOE). Under the provisions of contracts entered into with the utilities following the passage of the Nuclear Waste Policy Act (NWPA) of 1982, OCRWM was supposed to take title to the utilities waste for the purpose of disposing it in January 1998.

The Nuclear Regulatory Commission (NRC) has primary authority for the regulation of the disposal of HLW. The NRC shares regulatory authority for the transportation of HLW with the Department of Transportation (DOT). The Environmental Protection Agency (EPA) has an important regulatory role in setting the standard for the disposal of HLW.

8.10.3 Management of nuclear waste

In the United States, waste contaminated with plutonium and other long-lived radionuclides is referred to as Transuranic or TRU-waste. Waste must have more than 100 nanocuries (3 700 Bq) per gram of transuranic nuclides with half-lives longer than 20 years to be classified as TRU waste. All other wastes, including spent fuel, are classified as either low or high-level wastes.

8.10.3.1 LLW

LLW management in the US is the responsibility of the waste producers and the host states are responsible for disposal. A series of groupings, or Compacts, of individual States has been formed, and attempts are being made in many to locate suitable sites. So far no facilities have been sited to replace the two which are presently operational, despite the expenditure of many hundreds of millions of dollars. In the latest development, a disposal site in Utah was recently licensed. Envirocaire, Utah accepts only naturally-occurring and Class A low level waste. An application for permission to accept Class B&C wastes has not yet been approved (December 2000).

8.10.3.2 TRU wastes

DOE sited and developed the Waste Isolation Pilot Plant (WIPP) in New Mexico in the 1980's, where it has recently begun to dispose of transuranic or 'TRU-Wastes', from weapons production, at a depth of about 650 metres (2 150 feet) in a bedded salt formation. It is mandated to dispose of these wastes in a deep repository by US law.

After nearly a decade of delay, the final Environmental Impact Statement (EIS) was submitted to the Environmental Protection Agency in 1996 and a Final Draft Ruling was issued in May 1998, allowing DOE to commence limited disposal operations. However, because of the presence in much of the waste of unknown amounts of various toxic materials and metals, a separate licence was required from the State of New Mexico according to the Resource Conservation and Recovery Act (RCRA). A draft licence was issued by the State, also in May 1998, but because of legal challenges, no action actually took place until a federal judge ruled that 36 drums of Los Alamos waste could be shipped to WIPP without a full licence. The first shipment arrived on

26th March 1999, followed during April by wastes from the Idaho National Laboratory (INEL).

In early May 1999 the State of New Mexico withdrew a number of its legal challenges, and the RCRA permit was finally issued on 26th November 1999, although DOE must complete detailed waste audits at all the sites which will be sending wastes to the facility.

8.10.3.3 Spent fuel and/or HLW

Spent fuel from civilian power reactors is currently stored on-site, although the existing pool storage space is insufficient for the volumes likely to be generated over the lifetime of all operational and planned reactors (estimated at approximately 87 000 tonnes). Assuming that there is not a repository in operation, approximately 80 000 tonnes of additional storage will be necessary by the 2030's. An estimated 40 000 tonnes is currently in storage at the various NPP's, increasing by approx. 2 000 tonnes per annum.

In 1993, when states and utilities realised that goals established in their contracts with the DOE were not going to be met, a series of lawsuits were filed by some states and utilities. These suits were aimed at forcing the DOE to acknowledge its responsibility to begin accepting spent fuel for disposal in 1998 and to establish avenues for utilities to seek monetary damages in the event DOE did not meet its obligation.

In *Indiana Michigan Power Co. v. U.S., 88 f.3d 1272 (D.C. Cir.1996)*, the court ruled that the DOE does have a statutory obligation to begin accepting spent nuclear fuel (SNF) for disposal no later than January 31, 1998. In response to the lawsuit, the DOE informed contract holders on December 17, 1996 that it anticipated a delay in the beginning to dispose of SNF. DOE then requested comment on how those affected by the delay might be accommodated, given that the delay was unavoidable.

Following the DOE's announcement, utilities and states filed petitions seeking enforcement of the NWPA and of the ruling and remand order in Indiana Michigan Power Co. In *Northern States Power Co. v. U.S.D.O.E., 128 F.3d 754 (D.C. Cir. 1997)*, the DOE was ordered to proceed with contractual remedies in a manner consistent with the ruling in Indiana Michigan. It was precluded from concluding that it has not yet prepared a permanent repository, or that it had no authority to provide storage. In other words, DOE was prohibited from deciding that the delay is "unavoidable". An appeal by the DOE to the Supreme Court was turned down.

Subsequent cases *(Atomic Yankee Electric Company 1998 and Northern States Power 1999)* were filed to determine whether utilities have to follow the administrative procedure for resolving disputes set forth in their contracts or go directly to court to press their claims for damages. Following conflicting decisions, the two cases were appealed. In *Atomic Yankee Electric Company (2000)* and Northern States Power (2000), the utilities were permitted to pursue directly their cases in court. Thus, unless utilities settle with DOE, these rulings suggest that DOE will have to litigate several score of individual cases to determine the damages it owes. Recent estimates of the damages potentially owed by the Government if a repository never opens range from 39 to 61 billion US dollars.

Throughout 1998, 1999 and the early part of 2000, attempts have been made to introduce new legislation into the Senate which would amend the original 1982 NWPA. Different versions have proposed that some form of interim storage facility is built, to where spent fuel from the reactors could be shipped, thereby reducing the pressure on the facilities to build stores on-site at the reactors. The bills also would have removed the upper limit of 70 000 tonnes for the proposed repository capacity.

To date, none of the bills have become law, because the President of the U.S. either vetoed or threatened to veto them. There are a number of reasons President Clinton gave for opposing the bills, a major one being his opposition that no interim storage facility should be built at Yucca Mountain, Nevada until a decision is made on whether or not to proceed with building a repository there. All efforts to override a Presidential veto, which requires a 2/3 majority vote in both the U.S. House and Senate, have failed to date. The sponsor of the current bill, however, has promised to re-open debate on this issue.

The only amendment of the NWPA which has been successful took place in 1987, and removed the instruction on repository site selection which called for a number of sites to be investigated before a final candidate was identified. This resulted in the selection by DOE of the Yucca Mountain site in Nevada, adjacent to the DOE Nevada Test Site, as the sole candidate.

The 1987 Amendment (the NWPAA) also established the Office of the Waste Negotiator, mandated to find a volunteer site for an interim store for spent fuel, the so-called 'Monitored Retrievable Storage facility' (MRS) and instituted the Nuclear Waste Technical Review Board (NWTRB) to formally review the scientific and technical work of undertaken by the DOE to disposal of spent nuclear fuel and HLW, including transportation and packaging of the waste.

The latest conceptual layout of the subsurface emplacement area at Yucca Mountain includes one primary area that is crossed by parallel emplacement drifts that will be used for final disposal. The target horizon for the emplacement area is in a rock formation comprised of lithophysal welded tuff, some 300 metres above the water table.

Most major structural features were identified and marginally characterised from surface investigations. Extensive research is currently underway in the Exploratory Studies Facility (ESF), a looped tunnel construction completed in 1997. One main project currently underway is the Drift-Scale Heater Test, in which rock temperatures of up to 200°C are planned. It is not expected to be completed until at least 2004.

Other work is focussed on the tests, analyses, models and designs that are needed to support the site recommendation. As a first step towards site recommendation, the DOE plans to issue a Site Recommendation Consideration Report (SRCR) sometime in 2001. Once this document is issued, the DOE plans to hold a series of hearings in Nevada and a statutory public consultation and comment period. A formal recommendation to the President will then follow, containing comments from the hearings and accompanied by a final Environmental Impact Statement (EIS). If, as expected, the State of Nevada refuses to support the recommendation to the President, the issue must be debated in the Congress. The current schedule assumes approval in 2001; a licensing phase 2002–2005; construction 2005–2008 and operation in 2010.

The design being envisaged at the present time was put forward in an interim Viability Assessment in 1998. A thermal load strategy was presented in which waste canisters were placed close together, ensuring that the decay heat from the spent fuel raised the temperature of the surrounding rock to above 100°C, thereby boiling off any water which might corrode them and expose the waste in the short-term. At the repeated urging of the NWTRB to analyse alternatives to the current repository design, the DOE recently undertook to evaluate and analyse a low temperature thermal load strategy. If and when completed, this analysis would produce approximately 25 % of the previous thermal output (approximately 15 kW/acre, i.e. around 40 kW/hectare).

It is proposed to keep the repository open and accessible for 100 years from the start of waste emplacement, to allow the decision on backfilling and sealing to be made by future generations. The repository if therefore being referred to as a *'monitored geologic repository'.*

The failed search for a candidate site for the MRS attempted to introduce a volunteer process using a staged package of financial incentives, aimed at assisting interested communities and First Nation groups to participate in the review process.

However, because of political developments in those States where volunteers came forward (up to 19, mostly First Nation groups), the process stalled and was ultimately abandoned in 1995 without a potential site ever being identified. This process is continuing to some extent with various proposals for commercial facilities, the most recent being in Utah and Wyoming, where financial income is the main driving force.

The DOE is also assessing four possible sites with the aim of constructing three facilities for treatment and interim storage of recycled plutonium from dismantled nuclear weapons.

8.11 International organisations

8.11.1 Nuclear Energy Agency

At OECD/NEA, the Radioactive Waste Management Committee (RWMC) is supervising the work within the nuclear waste area. That work is mainly divided into three areas, each of them supervised by a Working Party:

- The Integration Group for the Safety Case (IGSC);
- Forum on Stakeholder Confidence (FSC);
- Working Party on Decommissioning and Dismantling (WPDD).

In addition to these groups there is also a Co-operative Programme on Decommissioning Projects (CPD) and a Regulators Forum.

RWMC has initiated discussions on issues like retrievability, the use of underground laboratories, stepwise decision making, etc. RWMC has also organised international peer review groups. On request by the Swedish Nuclear Inspectorate (SKI), one of these groups made a peer review of SKB's safety analysis SR-97 during spring of 2000.

The Integration Group for the Safety Case (IGSC) works in a discipline-oriented way on technical safety for final repositories with questions such as, for instance, development of performance analysis and how this can be used in order to communicate technical information and develop confidence between concerned stakeholders, how safety assessments may be used as a basis for decisions, development of scenarios, etc. Within IGSC there is also a special collaboration group for countries which have clay as their main geological alternative for a final repository for radioactive waste.

Forum on Stakeholder Confidence FSC had its first workshop in August 2000 (ref. 1). The workshop addressed a variety of topics ranging from the evolving participatory democracy, stakeholders' identity, trust in institutional frameworks, to the role of open dialogue in all aspects of radioactive waste management and disposal. The meeting brought together government nominated participants as well as a considerable number of stakeholders, including academics, sociologists, representatives of mediation and review groups, and elected political representatives. Invited presentations outlined the experience of different national waste management programmes in the area of public perception and stakeholder confidence.

The Forum is intended to help the NEA meet its obligation to Member countries to take up the challenges of understanding the needs of stakeholders and provide a neutral forum where the various actors can exchange information, identify working priorities and suggest best practices for decision-making in radioactive waste management and disposal.

WPDD's task is to co-operate in policy issues on decommissioning and dismantling. Reports are issued on dismantling experiences within CPD as well as from other countries.

Within CPD, more than 20 years' experience from decommissioning and dismantling of nuclear facilities has been collected. In total around 40 projects are included. In addition to the exchange of experiences and technical collaboration, CPD also provides reports on radiological data from dismantling of reactors.

During 1999 and 2000 the NEA has produced a number of status reports summarising the position, as perceived by the organisation on the advice of its consultants and experts from Member States, of the current status of deep geological disposal, with respect to work carried out internationally in the last 10 years.

For example, in *'Progress Towards Geologic Disposal of Radioactive Waste: Where Do We Stand?'*, published in 1999 (ref. 2), a number of points were identified where the existence of a general consensus amongst workers in the field can be recognised. These include:

- Deep geologic disposal is the most appropriate means of long-term management of the various disposal options considered;
- Significant progress has been made in relevant scientific understanding and in the technology required for geologic disposal in the past ten years;
- The technology for constructing and operating repositories is mature enough for deployment;
- The time-scales envisioned in the past for implementation of geologic disposal were too optimistic;
- There is a high level of confidence amongst the scientific and technical community engaged in waste disposal that geologic disposal is technically safe;
- The broader public, however, does not necessarily share the high level of confidence of the scientific and technical community;
- There is a need for continued high-quality scientific and technical work;
- There is a need for coherent policy and strict regulatory frameworks, with identified decision points, which also allow for public dialogue.

The report points to a number of specific areas where it suggests that significant progress has been made over the last ten years in terms of the technical activities required to implement disposal. These are:

- Development or construction of facilities for the treatment and intermediate storage of waste;
- Experience in laboratory and field experiments, including the study of natural analogues;
- Construction and operation of underground rock laboratories;
- Experience in site characterisation;
- Development of the engineered barrier design;
- Improvement of safety assessment techniques;
- Improved integration of site characterisation, design and safety assessment;
- Development of regulatory frameworks, including requirements for compliance.

In another report, '*Strategic Areas in Radioactive Waste Management*', also published in 1999 (ref. 3), six broad strategic areas have been identified which the RWMC (and its revised sub-groups) is planning to address in the coming years. These areas are:

I Overall waste management approaches
  - Environmental concerns, safety and sustainable development;
  - Comparison of the principles of radioactive and non-radioactive waste management;
  - Economic concerns – evaluation of the impact of financial pressures on waste management programmes.

II The process of repository development for long-lived radioactive wastes
  - Assisting in the resolution of technical issues to promote safety and provide grounds on which to base decision making;

- Developing common understanding between independent bodies such as implementers, regulators and policy makers on the goals to be achieved and respective responsibilities.

III Management of materials from decommissioning and dismantling, and of Very Low Level Waste (VLLW).

IV Public perception and confidence
- Understanding the concerns of stakeholders, communicating effectively, sharing practical experience from outreach/consultation exercises and public decision-making processes (the FSC is an attempt to address this issue).

V Implications of, and participation in, international guidance and agreements
- Identifying implications for waste management programmes of, for example, the new ICRP radiological policy applied to waste disposal and forthcoming update of ICRP 46, and the Joint Convention on Safety of Spent Fuel Management and Radioactive Waste Management.

VI System analysis and technological advances
- Identifying the emerging waste management and disposal technologies, for exchange of information and consideration of their implication at the system level.

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8.11.2 The International Atomic Energy Agency, IAEA

Since the publication in 1995 of the leading document in its Safety Standards Series, ”Principles of Radioactive Waste Management”, the IAEA has given emphasis elaborating the principles contained in it by establishing safety standards covering all major areas of radioactive waste management. This leading document also provides the technical basis for the Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management which was adopted at a diplomatic conference in 1997

The *International Conference on the Safety of Radioactive Waste Management* took place in Córdoba, Spain, from 13 to 17 March 2000 within the framework of the Agency's safety programme for the year 2000. The principal objective of the Conference was to enable members of the scientific community and representatives of waste produces, of bodies responsible for radioactive waste management, of nuclear regulatory bodies and of public interest groups – among others – to engage in an open dialogue. A document containing the conclusions and recommendations of the conference was prepared for consideration by the IAEA Board and General Conference in September 2000. This includes a suggestion to develop the 'Roundtable on Stakeholder Consensus'. The following text is taken from this paper (ref. 1).

> The evolution, under the aegis of the IAEA, of a *de facto* international radiation and nuclear safety regime was noted. In the area of radioactive waste safety, this regime consists of the *Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management* (which, it is hoped, will enter into force soon), the body of international waste safety standards established by the IAEA and other international organizations, and the IAEA's mechanisms for providing for the application of those standards.

Whilst recognizing that it is the responsibility of parliament and government to establish the legislative framework and take the political decisions necessary for the implementation of a national radioactive waste management policy, the conference concluded that such policy should reflect the following considerations:

- The producers of radioactive waste have the prime responsibility for its safe management, and it is they who should propose appropriate options and secure the economic resources necessary in order to discharge that responsibility;
- Radioactive waste management should be dealt with "holistically", so as to avoid actions that, while resolving immediate problems, could constrain future decision-making. However, where the demands of safety are overriding or long-term safety benefits can be secured, the waste may be managed with a view to improving the storage conditions;
- As there are uncertainties – not only scientific and technical, but also legal and political – inherent in the various options for the safe management of radioactive waste, it is necessary to pursue robust management approaches that will be acceptable in a wide range of possible future situations;
- Safety issues should be addressed independently, so as to ensure compliance with regulations and formally defined criteria, which may need periodic revision in order to take into account scientific and technical developments;
- The effective implementation of disposal options requires the clear definition, at the national level, of a step-by-step and transparent approach that enables the different interested parties, including the general public and public institutions, to participate in the decision-making process.

Good progress has been made in the development of technical approaches and in devising sound disposal options for radioactive waste, but further research and development work is always desirable. Irrespective of the option ultimately adopted by each country for High-level and Long-lived waste, there is a need to continue with development and assessments in the field of deep geological disposal since this will be necessary in the future to a greater or lesser extent.

International co-operation is essential to achieving technical and public consensus in support of national programmes. The following mechanisms are especially important in this connection:

- The *Joint Convention*, an incentive legal instrument which presupposes a high level of commitment by Contracting Parties to the safe management of radioactive waste;
- The international safety standards already in place;
- The international mechanisms for providing for application of these international safety standards.

In almost all of the Conference's Technical Sessions, there was discussion of the need to involve all interested parties ("stakeholders") in the decision-making processes related to radioactive waste management. Against that background, the IAEA's initiative in calling for the establishment of an "International Forum" where radioactive waste management safety issues might be discussed in a candid manner by all interested parties was welcomed.

At the same time as the General Conference, a two-day forum was held on the theme of '*Radioactive Waste Management: Turning Options into Solutions*' on the 19th and 20th September 2000.

References:

1. Paper to IAEA Board of Governors General Conference, September 2000. GOV/INF/2000/8-GC(44)/INF/5.

8.11.3 European Commission

In September 2000 much of the responsibility for nuclear safety issues within the Commission was transferred from the Environment Directorate (DG-Env) to the Transport and Energy Directorate (DG-Tren), although radiation protection matters will be unaffected.

Research work on Radioactive Waste Management and Disposal has been part of the European Atomic Energy Community (EURATOM) R&D programmes for more than 25 years, supervised by the Research Directorate. This is part of the general research and technological development (RTD) programme of the European Commission (EC) which covers activities in major fields of science and technology, organised in five-year framework programmes. The programme is performed through 'shared-cost' contracts by national laboratories of the Member States of the European Union (EU) with financial support from the EC (normally up to 50 % of the total costs) or through and in conjunction with the Joint Research Centres (JRC).

Since KASAM's state-of-the-arts report in 1998, the 5th of these so-called Framework Programmes has begun. In this programme (1998–2002) the emphasis is '*on moving from more fundamental research to an integrated approach by demonstrating the capability to solve real problems by providing practical solutions to the outstanding scientific and technical problems and public concern. The research performed should therefore contribute to demonstrate the technical feasibility of geological disposal and improve the scientific basis for the safety assessment. Further methods for achieving public confidence and trust should be established*' (ref.1).

The 5th Programme will include research out on "Radioactive waste management and disposal" under the area "Safety of the fuel cycle" with a total budget available of about 60 million ECU.

In the sub-area of 'waste and spent fuel management and disposal', following a first call of proposals in 1999, 23 projects were selected for further negotiation, several involving Swedish organisations and facilities.

The activities encompass research topics devoted to:

- Repository Technology
  - to test, at the Äspö Hard Rock Laboratory, the necessary steps in the design, construction and operation of a repository in a crystalline formation with the focus on 'engineered barrier system' (EBS) performance;
  - to demonstrate effective sealing and backfilling concepts in various URLs, i.e. of shafts in the HADES-URL's (RESEAL project), in Belgium, and new backfill concepts in the Opalinus Clay at the Mont Terri Rock Laboratory in Switzerland;
  - to assess the hydro-mechanical disturbance of rock mass due to construction of a URL in Eastern France in a deep clay formation.
- Performance Assessment (PA) of repository systems
  - to further develop PA methodologies and strengthening the scientific input basis for models used in PA calculations with the view of the treatment of the bentonite barrier in integrated PA, testing and incorporating of coupled processes in PA modelling;
  - to assess the possible long-term impact due to climate change on the safety of radioactive waste repositories in deep formations with the aim to advance the state-of-the-art of biosphere modelling for PA-use;
  - to identify a number of various safety and performance indicators and to test their suitability for evaluating the long-term performance of disposal systems.
- Long-term behaviour of repository systems
  - to test and assess the different barriers considered in the multi-barrier concept, namely the waste forms, the

buffer/backfill material and the natural (geological) barrier. These will be carried out under conditions, as near as possible to those expected in repositories, as well as under controlled conditions in laboratories.

In addition, a network (CROP) will be established as a forum for pooling and assessing experiences from various URL's in Europe, USA and Canada. Particular interest will be given to construction, instrumentation and correlation of theoretical models with field data, especially concerning the 'engineered barrier system' (EBS).

As regards the 4th Framework Programme, now complete, a review conference was held in Luxembourg in November 1999 (ref. 2). The general topics covered, and the relevant conclusions presented are reproduced below:

> The European Commission-supported R&D activities on "Radioactive Waste Management and Disposal" have contributed to further develop a common understanding and consensus within the European Union on the key issues. Research, together with other activities such as the "Community Plan of Action", promoted international co-operation and provided a forum for exchange of information and expertise at high level between the various groups involved inside and outside the European Union.
>
> Important developments and achievements in "Radioactive Waste Management and Disposal" have been reached with regard to the practical demonstration of the technical feasibility of disposal concepts and the assessment of the long-term behaviour of repository components through large field experiments in Underground Research Laboratories. Specific investigations of engineered barriers and the geological environment contributed to extend the scientific knowledge and database on the processes of relevance to assess the confinement properties of the repository.
>
> The results achieved have also contributed to improve the confidence in the methodological approaches, the tools and reliability in modelling results and predictive capabilities of performance assessment (PA) calculations, for evaluating the performance of the repository system.

In most European countries developing deep underground repositories for the final disposal of their long-living radioactive waste, discussion about acceptable solutions in waste management strategies have involved the possible implications of retrievability.

In order to compare approaches to this issue, a Concerted Action (CA) on *"The retrievability of long-lived radioactive waste in deep underground repositories"* was set up by the EC (see also chapter 2 of this report). This reviewed the current understanding and different views of retrievability, which led to

- an overview of how different countries are considering the issue;
- the establishment of an interpretation and working definition of the retrievability concept;
- a methodology allowing the assessment of how retrievability varies between different disposal concepts;
- identification of the implications on repository design and long term safety.

Socio-political, monitoring and safeguards-related aspects were also addressed (ref. 3).

International co-operation and partnerships in research projects carried out in the URLs and the exchange of information were stimulated via the establishment of a **CL**ub of **U**nderground **S**torage, **TE**sting and **R**esearch facilities (**CLUSTER** – URL's) in which the Äspö URL was represented.

Preparations are now underway to go beyond the current 5th EURATOM Framework Programme (1998–2002) by reflecting on the strategic issues to be covered by a future European research programme, or 6th Framework Programme. The EURATOM Scientific and Technical Committee (STC) has prepared a strategic discussion paper.

Considering that in the present EU, more than one third of the electricity is generated in nuclear power plants, that EU electricity demand can be expected to continue to increase in the future and to ensure that future generations have a real selection of available technologies to choose from when they have to decide on the energy supply system that would best suit their needs and acceptance criteria, the STC strongly recommends an increase in the European nuclear R&D budget for the period following the 5th EURATOM Framework Programme. With respect to research issues related to waste management, the STC recommends the development of a common scientific understanding of the presently used concept of *"reasonable assurance of safety"* for the long-term post-closure phase of repositories, in order to ensure reasonably equivalent legal interpretations among EU countries in licensing procedures. It is also felt desirable to develop a common understanding of long-term safety aspects related to the concepts of *'indefinite storage'* and *'retrievable disposal'*.

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KASAM, the Swedish National Council for Nuclear Waste – established in 1985 – is an independent committee attached to the Ministry of the Environment. KASAM's task is to investigate issues relating to nuclear waste and the decommissioning of nuclear installations and to provide the Government and certain regulatory authorities with advice on these issues.

KASAM's members – who largely comprise professors from Swedish and Nordic universities and institutes of technology – represent independent expertise within different areas of importance for the final disposal of radioactive waste, not only in natural science and technology but also in areas such as ethics, psychology, law and social sciences.

KASAM is responsible for evaluating the programme for research and development – concerning the final disposal of spent nuclear fuel – which the Swedish nuclear power utilities present every three years.

An important activity carried out by KASAM is that of providing a forum for different viewpoints and for experts in Sweden and abroad on the topic of nuclear waste and related issues. A number of seminars on various themes have therefore been held.

KASAM is also responsible for presenting a special independent evaluation of the state of knowledge within the nuclear waste area every three years. This publication contains the state-of-the-art reports for 2004.

The report is available in Swedish and in English.

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