5. WASTE MANAGEMENT AND DISPOSAL

5.1 Introduction

The different waste types described in Chapter 3 will require long-term management in a final repository. Different types of repositories are considered for the disposal of low- and intermediate-level waste and of high-level waste. This chapter focuses mainly on the disposal of high-level radioactive waste[1] because advanced fuel cycles are expected to have a considerable impact in this regard and the radionuclide inventory of high-level waste can be relatively well estimated from the results of neutronic calculations.

The information collected within this Study on the characteristics and amounts of low- and intermediate-level waste which will arise from the advanced fuel cycles was not very precise or reliable. As shown in Chapter 3, no major variations are expected on LILW volumes, and since activities and heat loads will remain at low levels, no sizeable impact is expected to result from these waste streams that might modify repository concepts. However, the impact of low- and intermediate-level waste will need to be assessed in future studies when information on these waste categories will improve. For example, if novel materials were proposed, their impact on geological repository safety would have to be evaluated early in the decision-making process when reactor designs were being developed and considered. The rigorous assessment processes developed by countries for assessing suitability of radioactive waste would need to be followed once the materials to be used in advanced fuel cycle schemes will be known.

5.2 Disposal of LILW

5.2.1 Disposal of short-lived LILW

Short-lived LILW can be disposed in a near-surface facility or in a mined repository at a depth of typically a few tens of metres (or in a deep geological repository in case of co-disposal with high-level and/or long-lived waste). Disposal facilities for short-lived LILW constructed near the surface or at moderate depth are in operation in various countries, e.g. in France, Spain, Sweden, Finland, United Kingdom, United States, and Japan. Waste acceptance criteria have been drawn up for these facilities.

For the LLW and ILW that are identified in the fuel cycle schemes it can be verified whether they can be considered as LILW-SL and hypothetically could be accepted at the existing disposal facilities or their future equivalents.

5.2.2 Disposal of long-lived LILW

Long-lived LILW has to be disposed of in a deep geological repository. Just two such facilities have been developed and were/are in operation: Morsleben (Germany, ceased accepting waste in

1. For the purposes of this chapter high-level waste includes spent nuclear fuel.

September 1998) and WIPP (United States). A third facility (Konrad, Germany) has been licensed but is not yet operational. Waste acceptance criteria have been drawn up for these facilities.

For the LLW and ILW that are identified in the fuel cycle schemes and that do not satisfy the waste acceptance criteria for short-lived LILW, it can be verified whether they can be considered as long-lived LILW that could hypothetically be accepted at the existing disposal facilities, or their future equivalents.

5.3 Geological disposal of HLW

5.3.1 Objective

Long-lived radioactive waste, such as spent nuclear fuel (if declared as waste) and high-level waste from fuel reprocessing must be contained and isolated from humans and the environment for many thousands of years [1]. Isolation means keeping the waste away from the biosphere by means which do not rely on active measures in the future and making deliberate human intrusion to the waste difficult without special technical capabilities. The avoidance of locations that may attract inadvertent human intrusion is typically a factor in repository siting. Since complete containment cannot be guaranteed for the whole period during which waste represents a potential hazard, any eventual releases from the repository system should present an acceptable risk. The long-term safety of the repository is provided by the protective functions of the geological environment and the engineered barriers placed around the waste, as well as by the stability of the waste form. The disposal of long-lived waste in deep and stable geological formations is a generally accepted option for the long-term management of radioactive waste.

5.3.2 Considered host formations

At present various types of geological formations are considered as possible host formations for deep disposal of long-lived radioactive waste. The main types considered are:

- hard rock formations, mainly granite; this option is currently studied by countries such as Canada, Finland, Japan, Spain, Sweden and Switzerland;
- argillaceous formations including plastic clays, over indurated clays and mudstones; this option is currently studied in counties such as Belgium, France, Japan, the Netherlands, Spain, Switzerland and more recently Germany;
- salt formations: salt layers as well as salt domes are being considered; disposal in salt domes is studied in Germany and the Netherlands;
- volcanic formations including tuff and basalt; the present United States programme for high-level radioactive waste disposal focuses on a welded tuff formation at Yucca Mountain.

5.3.3 Functioning of a geological repository

Repositories are typically sited in stable geological environments that offer favourable conditions in which the waste and engineered barriers are protected over a long time period [1]. This means that key characteristics that provide safety, such as mechanical stability, low groundwater flux and favourable geochemical conditions, should be unlikely to change significantly over relevant timescales.

Robustness of the repository system is favoured by the multi-barrier concept. The barriers should be complementary, with diverse physical and chemical components and processes contributing to safety, so that uncertainties in the performance of one or more components or processes can be compensated for by the performance of the others.

The safety functions of a repository system can be described in relatively simple terms. Different sets of safety functions have been developed within the main national safety cases and a number of international projects. For example, the safety functions that were identified for water bearing formations in the SPIN project [2] of the European Commission are:

- physical containment: a watertight barrier is isolating the radioactive waste from groundwater during the first phase after repository closure; as long as this safety function is effective, no release of radionuclides can occur from the waste form; physical containment makes the disposal system more robust and easier to analyse by preventing dispersion of radionuclides during the strongly transient initial phase of the repository history (re-saturation processes, heat release, strong radiation, pressure rebuilding, etc.);
- slow release: after container failure, when groundwater comes in contact with the conditioned waste, leaching of radionuclides from the waste matrix starts in combination with the degradation of the waste matrix; various physico-chemical processes, such as corrosion of and lixiviation from the waste matrix, precipitation, sorption or co-precipitation strongly limit the radionuclide releases into the surrounding layers;
- retardation: the radionuclides dissolved in the groundwater that is in contact with the waste will start to migrate through the bentonite buffer and the host formation; because of the very low groundwater fluxes in potential host formations, this transport will be very slow; furthermore, many radionuclides will be sorbed onto minerals of the buffer and the host formation; retardation delays the releases and drastically limits the amounts of radionuclides that are released into the biosphere per unit of time;
- dispersion and dilution: once the long-lived radionuclides leave the repository's barrier system, they are released into the overlying or surrounding aquifers and eventually into the accessible environment; the dispersion and dilution processes in the aquifers and surface waters will further reduce the radionuclide concentrations in the waters that are directly accessible by man.

5.3.4 Repository designs

In the case of geological disposal of long-lived radioactive waste, the geological barrier is complemented with a number of engineered barriers which depend on the characteristics of the host formation and of the disposed waste type. For a given repository site, disposal concepts are designed by taking into account waste characteristics such as radionuclide content, heat generation, criticality, radiation field, leaching rate, gas generation, etc.

Two main types of disposal configurations are considered for high-level waste:

- disposal in galleries, where the waste is placed along the axis of a gallery;
- disposal in boreholes, where the waste is placed in horizontal or vertical boreholes that are drilled from a gallery.

A distinction has to be made between repositories located in water bearing formations on the one hand, and repositories located in salt or unsaturated formations on the other hand.

Repository designs developed for water bearing formations, i.e. hard rock and argillaceous formations, typically consist of the following engineered barriers:

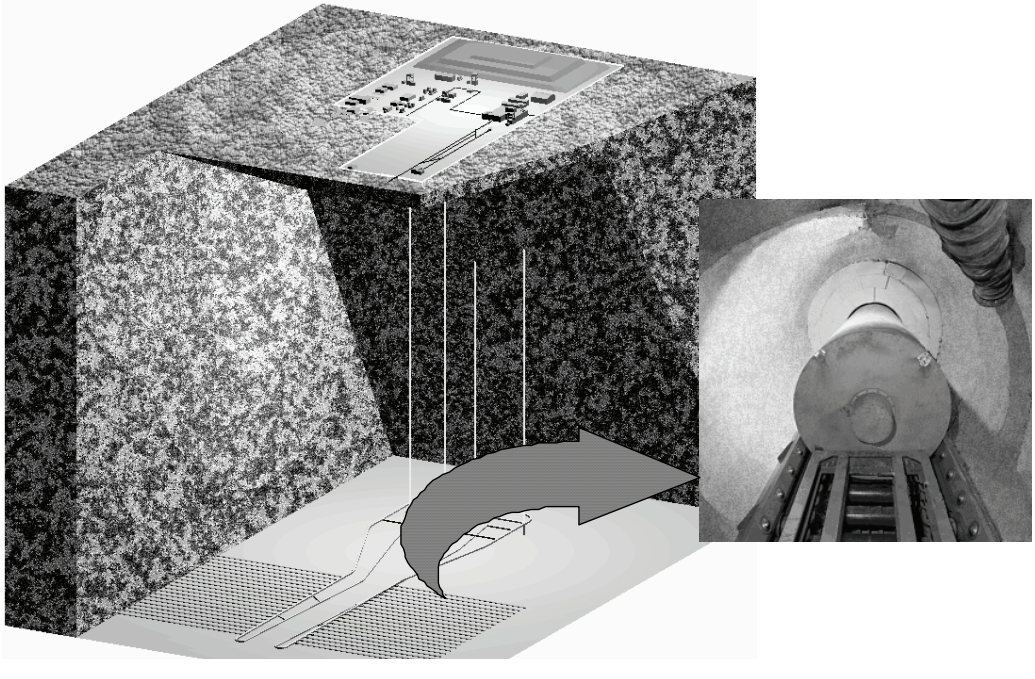
- a metallic container, often called canister or overpack, that has to remain intact during the initial gradient phase of the repository;
- a buffer surrounding the container, which consists in many designs of a swelling clay material, such as bentonite; the buffer has to fill possible voids and fractures and prevent advective water flow around the container;
- a backfill is used to fill transport and access galleries;
- seals and plugs are used to isolate the disposal gallery or borehole from the transport and access galleries and from the part of the host formation that have been disturbed by the excavation of the gallery or borehole.

For repositories located in salt or unsaturated formations several of the above mentioned engineered barriers are also used, but some may not be needed.

a) Example of a repository designed for a granite formation

The reference disposal concept, developed by ENRESA for a high-level waste repository in granite is based on the disposal of four spent fuel assemblies packed in carbon steel canisters in long horizontal galleries. The canisters are surrounded by high density bentonite. The disposal galleries are grouped in two symmetrically arranged disposal areas. Access is accomplished by means of transport galleries, which run perpendicular to the disposal galleries. The transport galleries start from a central area, which includes the required underground infrastructure. Transport between the surface and the central underground area is accomplished by 4 access shafts and a ramp. Figure 5.1 shows a view of the underground installations.

Figure 5.1. Underground installations for a reference repository concept in granite (Spain)



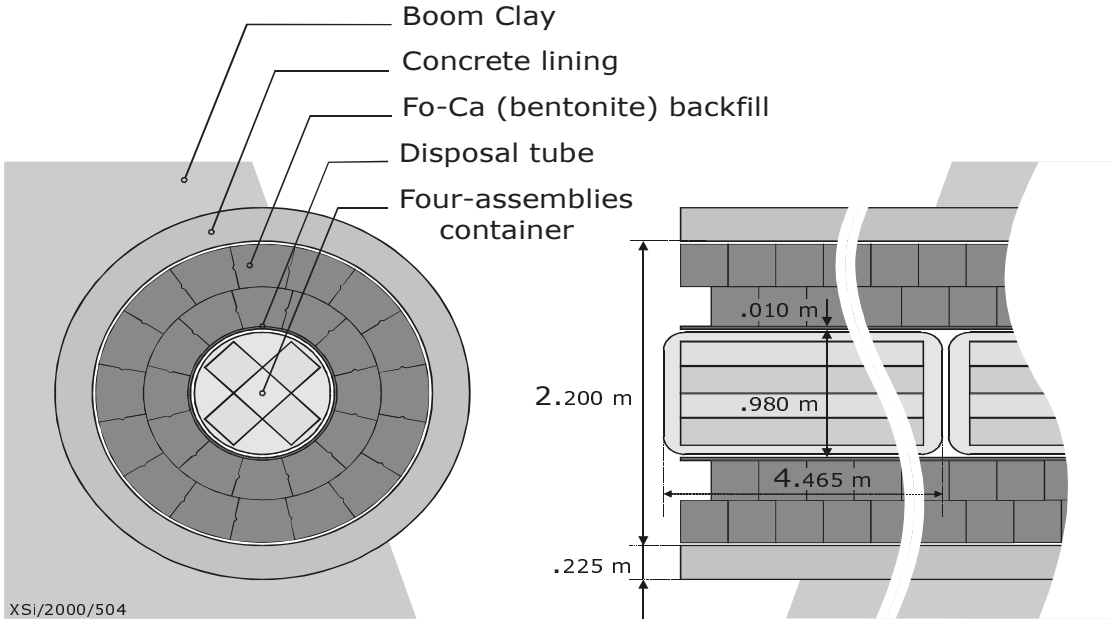
The 10 cm thick canisters are expected to provide a minimum period of containment greater than 1 000 years. The separation between canisters is mainly determined by thermal constraints. A separation of 2 m between canisters and of 35 m between disposal drifts is considered in order not to exceed a temperature of 100°C in the bentonite.

When a disposal drift is filled, it is sealed using blocks of bentonite and structures of concrete at the entry to the drifts. Galleries, shafts and other remaining rock cavities are backfilled, after completion of waste emplacement, with a mixture of bentonite and sand or an appropriate crushed material. The bentonite content of backfilling material is 10%, increasing up to 20% at the top of the galleries. Sealing plugs of galleries and shafts are made of compacted bentonite blocks piled over the whole cross section.

b) Example of a repository designed for an argillaceous formation

The repository configuration that has been developed by the Belgian radioactive waste management agency ONDRAF/NIRAS in the SAFIR 2 study for disposal of high-level radioactive waste in the Boom Clay formation in Belgium is similar to the configuration shown in Figure 5.2.

Figure 5.2. Scheme of the near field of a repository in clay (Belgium)



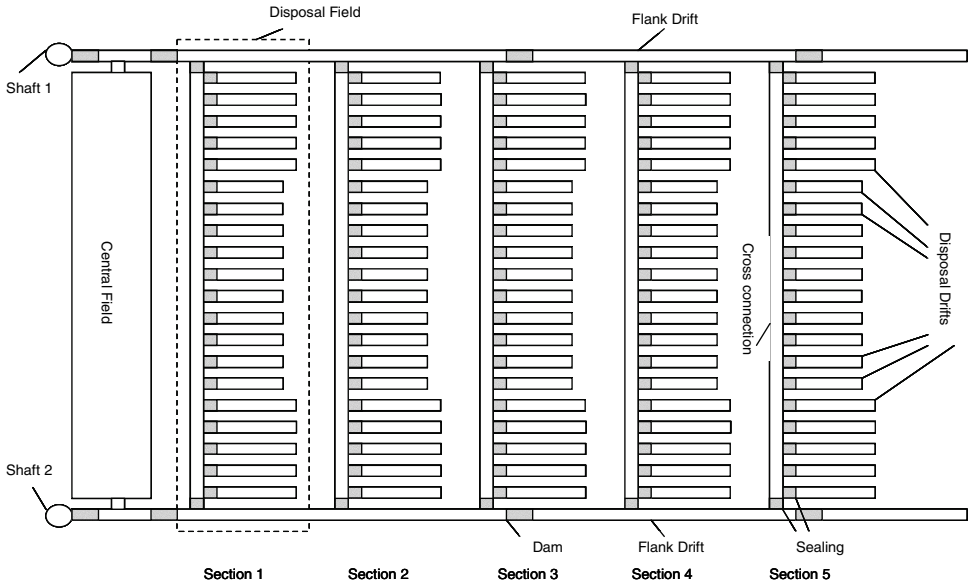
The repository is assumed to be constructed in the middle of the host formation. A central access facility consists of 2 or more vertical shafts connected by the main gallery, from which transport galleries start to the various disposal areas. The disposal galleries are perpendicular to the transport galleries. A scheme of a disposal gallery is shown in Figure 5.2. The spent fuel assemblies or vitrified high-level waste canisters are packed in stainless steel containers, which are placed in a disposal tube. The disposal tube is surrounded by a 60 to 80 centimetres thick bentonite-based buffer. The disposal galleries require the use of a concrete lining because of the high plasticity of the host clay formation.

c) Example of a repository designed for a salt formation

The disposal facility considered by GRS is assumed to be located in a salt dome at a depth of 870 m. The thickness of the overburden is about 300 m. The waste is disposed of in horizontal drifts in a single story (single layer) arrangement. The disposal facility consists of a central field, two flank drifts and five cross connections, with 100 disposal drifts (Figure 5.3). One cross connection and the adjacent 20 disposal drifts and the two segments of the flank drifts form one section. Access to the facility is provided by two shafts.

The volumes and dimensions of the near field assemblies such as flank drifts, disposal drifts, shafts and central field can be found in [14].

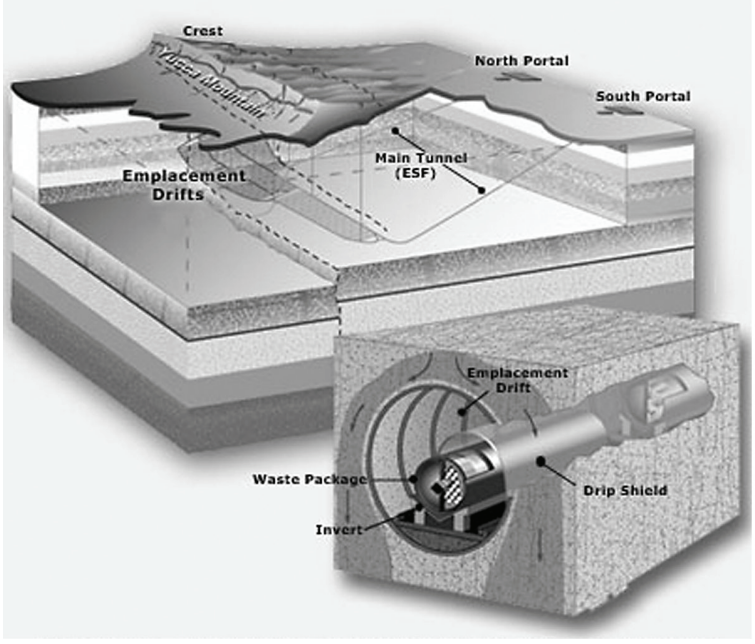
Figure 5.3. Sketch of a disposal facility in salt (Germany)



d) Example of a repository designed for a volcanic formation

The repository concept developed for a repository at the Yucca Mountain site in the United States is located in unsaturated, fractured volcanic tuff. As shown in Figure 5.4, the repository consists of access ramps from the surface and an array of emplacement drifts about 5 metres in diameter, about 2 km long and separated by about 80 metres. The emplacement area covers several km^2 and is located several hundred metres below the surface and several hundred metres above the water table. Both intact spent fuel and vitrified high-level waste would be emplaced in horizontal waste packages set on an engineered invert in the drifts. The waste packages are double shell, with a layer of corrosion resistant nickel-based alloy over a thick structural layer of stainless steel.

Figure 5.4. Conceptual design of a disposal facility in unsaturated tuff (United States)



The density of waste emplacement is constrained to meet thermal design objectives for maximum temperature at the drift wall and at the centre-point between drifts. The engineered system is designed to work with the natural host environment to limit and delay water contact with the waste and subsequent radionuclide release and transport to the accessible environment.

5.4 Evaluations of the impact of advanced fuel cycles on geological disposal of HLW

In a preceding NEA report [3], the impact of advanced fuel cycles on geological disposal was estimated by making extrapolations of the results from existing performance assessments. In the present study, specific evaluations were carried out to allow for a more accurate impact assessment. Modifications of the existing repository concepts for HLW disposal needed to accommodate the HLW types arising from advanced fuel cycles were evaluated and the performance of these concepts for the new waste types were assessed as well the resulting radiological consequences.

5.4.1 Fuel cycles selected for detailed analyses

In Chapter 2, one reference and twelve advanced fuel cycle schemes have been defined. A smaller number of representative schemes were selected for quantitative evaluations.

The schemes selected for the quantitative evaluations are:

- the reference Scheme 1a (PWR, open cycle, UO_2 fuel);
- Scheme 1b (PWR, PUREX reprocessing, single recycling of Pu as MOX) representative of current technology;
- one partially-closed fuel cycle, Scheme 2a (PWR, PUREX reprocessing, multi-recycling of Pu as MOX); and
- one fully-closed fuel cycle, Scheme 3cV1 (GFR, pyroreprocessing, carbide fuel).

The characteristics of the HLW types that have been considered for the analyses are given in Appendix G. For the estimation of those characteristics it was necessary to make a number of assumptions that are listed below.

- It is assumed that 0.1% of the ^{129}I in the spent fuel is present in the vitrified HLW.
- For most of the analyses it is assumed that HLW is disposed after a cooling period of 50 years after discharge from the reactor but in one case (Scheme 1b) it is possible to dispose one of the HLW types (MOX spent fuel) after more than 50 years because temperature limitations could not be respected.
- For the advanced fuel cycles using pyroreprocessing, the exact conditioning method of the HLW is not yet defined. It is assumed that the matrices used have a stability in disposal conditions comparable to the glass matrix that is used at present to condition the vitrified HLW arising from the PUREX reprocessing of uranium oxide fuels. For all conditioned HLW types a matrix lifetime of 100 000 years (70 000 years for Japanese calculations) was assumed. For the spent fuels a matrix lifetime of 200 000 years was assumed.
- For Scheme 3cV1 it is assumed that the noble metal fission products are conditioned together with other metallic waste such as cladding and end-pieces. Although this waste is not HLW, because its heat production is negligible, it is considered in the analyses to include all the fission products. The degradation of the metallic waste matrix is neglected because noble metals have a low solubility.

To illustrate the potential impact of advanced fission product management techniques discussed in Chapter 4, a sub-variant of Scheme 3cV1, which assumes that Cs and Sr are separated from the HLW, is also considered. For this specific scheme the following assumptions are made:

- 99% of the Cs and Sr are separated from the liquid HLW;
- the remaining HLW is vitrified and disposed of in a geological repository after 50 years;
- Sr can be disposed of in a repository for short lived waste after a sufficiently long cooling period;
- Cs is vitrified after an additional cooling period and disposed of in a geological repository after a total cooling period of 100 years.

5.4.2 Repositories considered in the analyses

Five organisations participating in the study volunteered to make numerical analyses of the impact of advanced fuel cycles on HLW disposal. ENRESA made calculations for a hypothetical repository located in a granitic formation. JNC made calculations for a hypothetical repository located in a hard rock formation. SCK•CEN made calculations for a hypothetical repository located in a clay layer. GRS made calculations for a hypothetical repository located in a salt dome. ANL performed analyses of the selected fuel cycle impacts on thermal design limits and long-term performance for a conceptual repository located in fractured unsaturated volcanic tuff.

5.4.3 Impact on repository design

As already mentioned in Section 5.3.4, repository concepts are designed by taking into account waste characteristics such as radionuclide content, heat generation, criticality, radiation fields (γ, neutrons), leaching rate, gas generation, etc. The results given in Chapter 2 (Tables 2.2 and 2.6)

show that the fissile material content of the vitrified HLW arising from reprocessing is much lower than that of spent fuel. Also the activity of and emitted neutrons are lower in the case of HLW from reprocessing than for spent fuel. In both cases, disposal of spent fuel or of vitrified HLW, gas generation in the repository will be due mainly to corrosion of the metallic containers. Consequently, it is not expected that the disposal of vitrified HLW arising from the reprocessing of spent fuel from advanced fuel cycles will require considerable adaptations of the disposal concepts.

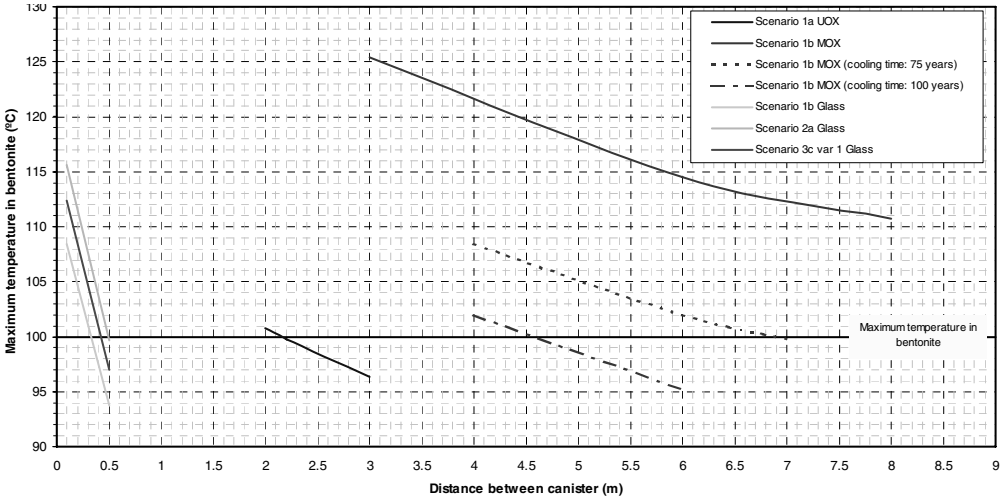
In some fuel cycle schemes considered in this study but that were not selected for detailed analyses, irradiated targets are considered as waste. If these targets are not reprocessed, they can be possibly problematic waste forms. For example, the Am targets arising from Scheme 2c generates 17 times more heat and emits 1 260 times more neutrons per kg than the spent fuel from Scheme 1a. The disposal of such very high-level waste forms will require the development of new repository concepts and of an adapted engineered barrier system. It should also be noted that most spent fuels from fast reactors are significantly hotter than spent fuel from Scheme 1a and therefore, if they would not be reprocessed; their disposal would require considerable adaptations of the repository concept and of the engineered barrier system or a very long interim storage.

Temperature limits that have to be respected for a number of the repository components are limiting factors for the allowable disposal density of HLW in a geological repository. In the following sections the maximum allowable disposal density will be estimated for the selected fuel cycle schemes based on temperature limits.

a) Results of thermal calculations for a repository in granite

ENRESA carried out thermal calculations for a hypothetical repository in granite. The calculated temperatures of the buffer in contact with the container are shown in Figure 5.5. It appears that the MOX spent fuel considered for this study, with a burn-up of 60 GWd/tHM, cannot be disposed in this repository concept after a cooling time of 50 years. Calculations were performed also on the temperatures after 75 and 100 years of cooling period. If the MOX spent fuel should be disposed after 50 years, the repository concept would have to be modified, e.g. by enlarging the diameter and/or the length of the container.

Figure 5.5. Calculated maximum temperatures in the bentonite buffer as a function of the distance between 2 high-level waste containers (ENRESA calculations for a repository in granite)



The calculated lengths of the disposal galleries are given in Table 5.1. It may be noted that the reduction of the ratio of the required length of disposal galleries for Schemes 2a and 3cV1 as compared with Scheme 1a, respectively 0.844 and 0.238, is somewhat higher than the ratio of the thermal outputs of the corresponding high-level waste, respectively 0.946 and 0.271.

Table 5.1. Required excavation length for different schemes
(ENRESA calculations for a repository in granite)

Scheme	Cooling times (years)	Container load	Waste production (cont./TWhe)	Container length (m)	Space between containers (m)	Excavation (m/TWhe)	Total excavation (m/TWhe)	Ratio
1a	50	3 UOX	1.327	5.28	2.2	9.925	9.92	1
1b	50	1 MOX	0.437	5.28	t.l.e.[1]	t.l.e.[1]	>4.924[2]	–
		1 Glass	2.592	1.60	0.3	4.924		>0.5[2]
	75	1 MOX	0.437	5.28	7.0	5.365	9.51	0.959
		1 Glass	2.592	1.60	0	4.146		
2a	50	1 Glass	3.987	1.60	0.5	8.373	8.37	0.844
3cV1	50	1 Glass	1.181	1.60	0.4	2.363	2.36	0.238

1. Temperature limit exceeded.
2. Values obtained considering only the HLW glasses.

b) Results of thermal calculations for a repository in clay

Depending on the host formation the repository site and the design, various temperature limits have to be respected. Examples of temperature limitations are:

- Maximum temperature of the buffer at its interface with the container (canister) wall, which has to remain in many repository designs below 100°C.
- Maximum temperature of the host formation.
- Maximum temperature in the aquifers overlying the repository.

For several repository designs the most stringent thermal limitation is the maximum temperature of the buffer where it is in contact with the container. In this case detailed thermal calculations can be satisfactorily estimated by using a linear thermal output limit. Examples of candidate repository sites, for which a linear thermal limit was applied, are:

- The Boom Clay repository design: 300 W/m [5].
- The Opalinus Clay repository design: 200 W/m [6].
- The Yucca Mountain repository design (although it is not in clay), for which a maximum linear thermal output of 900 W/m was applied [4].

An overview of the HLW types arising from the 5 fuel cycle schemes analysed, the corresponding thermal outputs, and the estimated length of disposal galleries in case of disposal in Boom Clay is given in Table 5.2.

Table 5.2. High-level waste types arising, thermal outputs and needed length of disposal galleries for a maximum linear thermal loading of 300 W/m (SCK•CEN calculation for a repository in clay)

	Fuel cycle scheme				
	1a	**1b**	**2a**	**3cV1**	**3cV1 (separation of Cs, Sr)**
SF assemblies (#/TWhe)	3.981	0.437	n.a.	n.a.	n.a.
HLW canisters (#/TWhe)	n.a.	2.592	3.987	1.181	0.705
NM canisters (#/TWhe)	n.a.	n.a.	n.a.	0.085	0.085
Cs-waste canisters (#/TWhe)	n.a.	n.a.	n.a.	n.a.	0.225
Thermal output HLW after 50 years (W/TWhe)	2 110	2 031	1 997	571	12.5
Thermal output Cs-waste after 100 years (W/TWhe)	n.a.	n.a.	n.a.	n.a.	120.9
Length of disposal galleries (m/TWhe)	7.033	6.77	6.657	2.016	0.755
Length of disposal galleries relative to 1a	1	0.963	0.947	0.287	0.107

Table 5.2 shows that single or multiple recycling of Pu does not significantly reduce the thermal output of the high-level waste. On the other hand a fully-closed cycle such as Scheme 3cV1 leads to a reduction of the required length of disposal galleries by 3.5 as compared with Scheme 1a. Separation of Cs and Sr results in a drastic reduction of the thermal output and consequently the required length of disposal galleries is reduced by a factor 10 as compared with Scheme 1a. An optimisation of the waste packages might allow a further reduction of the required length of disposal galleries.

c) Results of thermal loading calculations for a repository in tuff

Thermal design goals restrict repository utilisation by limiting the loading density of the waste streams. In Yucca Mountain's nominal, high-temperature operating mode (HTOM), there are two thermal design limits that affect potential repository capacity:

- The host rock must remain below 200°C everywhere in the repository to limit mineral alteration. Meeting this limit at the drift wall results in limits on the linear thermal loading density along a drift, driven primarily by early-time fission product decay heat.
- The temperature midway between storage drifts must remain below the boiling temperature of 96°C at all times to permit drainage of water through the fracture network between drifts. This results in limits on areal thermal loading density primarily driven by long-term actinide decay heat. Drifts in Yucca Mountain are assumed separated by 81 m.

The decay heat from waste stream was calculated for each scheme, in W/TWhe, for up to 10 000 years using the ASIDE code for isotopic decay [7]. The maximum allowed loading density for each scheme was calculated using the AFCI Repository Thermal Model [8]. Results of these calculations are shown in Table 5.3, where calculated utilisation limits are expressed as linear and areal densities of waste arising from the generation of the indicated TWhe of electricity. Relative to Scheme 1a, all recycle schemes raise utilisation limits and the most prominent increase is associated with the Scheme 3cV1. For Schemes 1a, 1b, and 2a, the long-term a real limit driven by actinides determines the results. For the complete actinide recycle, Scheme 3cV1, the actinide decay heat drops dramatically. The results shown assume extended cooling or other thermal management of the short-term fission products to achieve the utilisation limits shown. In addition, since the models used

represent realistically only the central region of a very large repository, the absolute loading limits shown in Table 5.3 are likely to be conservatively low.

Table 5.3. Calculated utilisation limits in an unsaturated tuff repository

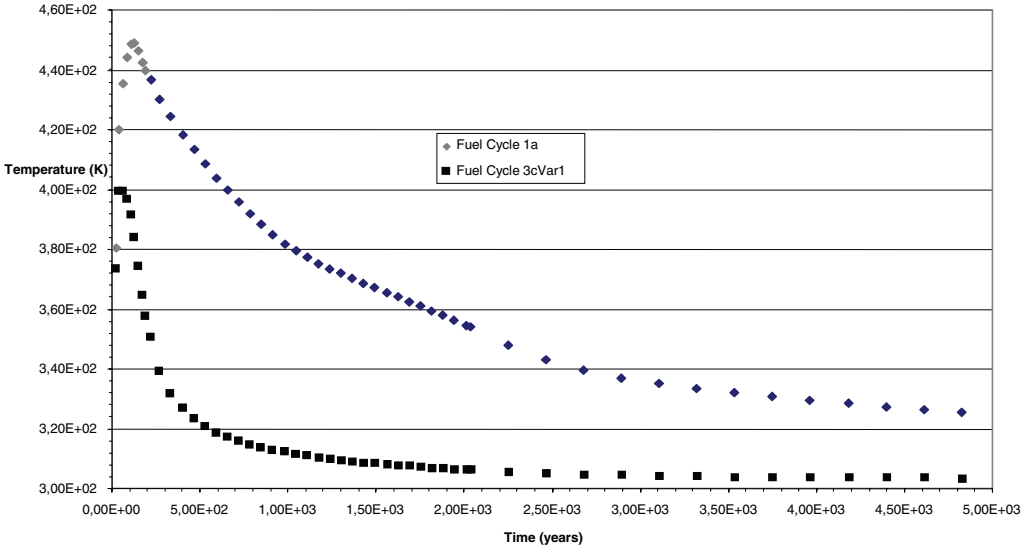
Scheme	1a	1b	2a	3cV1
Linear (TWhe/m)	0.52	0.60	0.85	8.27
Areal (TWhe/m^2)	6.42E-03	7.35E-03	1.05E-02	1.02E-01
Linear relative to Scheme 1a	1.00	1.14	1.63	15.90

d) Results of thermal calculations for a repository in salt

In the German concept, the maximum temperature for a repository in salt is expected to be limited by future regulations to 200°C at the contact surface between a disposal cask and the rock salt. Within the estimation of the temperature field a bulk temperature was calculated. The maximum bulk temperature has been fixed at 180°C. Starting from this maximum bulk temperature the upper boundary of the heat load for the disposal drifts has been determined. The maximum heat load for a short disposal drift (length 219.5 m) has been estimated to be in the range of 130 to 135 kW while the maximum heat load for a long disposal drift (length 229.5 m) has to be in the range of 140 to 145 kW.

Figure 5.6 shows the evolution of the temperature (in Kelvin) at a central point of the disposal field for waste from Schemes 1a and 3cV1. The temperature for disposal of spent fuel from Sheme 1a reaches its maximum of 450K (177°C) at 129 years and is decreasing to 340K after 2 680 years. The temperature for disposal of high-level waste from Scheme 3cV1 reaches its maximum of 400K (127°C) at 64 years and is decreasing rapidly to 340K after 593 years. The ambient temperature of the host rock is assumed to be 303K (30°C). The fast decrease of the temperature of Scheme 3cV1 waste is due to the fact that the heat load of this waste is decreasing drastically. Fifty years after unloading from the reactor the heat load is 518 W per TWhe and after 400 years it is 1.7 W per TWhe.

Figure 5.6. Evolution of temperature relevant for salt creep calculation



The two completely different shapes of the temperature curves have a significant influence on the behaviour of the disposal facility in salt. For Scheme 1a the salt creep will be drastically increased due to increasing temperature and will stay for a very long time at a relatively high level. For Scheme 3cV1 the salt creep will also increase but not to such a high extent. After a relatively short time of increased salt creep, the creep will decrease and remain for a long time at the level of unaffected salt. This is important because the salt creep will result in a convergence of the repository vaults and thus contribute to the containment of the waste.

Because of similar heat load characteristics the shapes of the temperature curves of Schemes 1b and 2a are somehow similar to the temperature curve of Scheme 1a. Therefore, the four estimates to be carried out may be grouped into two categories: three analyses of disposal "hot" waste (Schemes 1a, 1b and 2a) and one analysis of disposal of "cold" waste (Scheme 3cV1).

e) Impact of extending the cooling time

In the analyses presented above it was generally assumed that the high-level waste is disposed of after 50 years of cooling. Increasing the cooling time might allow for a considerable reduction in the size of the repository in the case of advanced fuel cycles because the thermal output of their high-level waste is decreasing much faster than it is the case for the spent fuel in Scheme 1a. An overview of the thermal outputs of the schemes considered s given in Table 5.3.

Table 5.4. Thermal outputs from HLW after 50 and 200 years of cooling

		Fuel cycle scenario			
	Unit	**1a**	**1b**	**2a**	**3cV1**
A. Thermal output HLW (50 y)	W/TWhe	2 110	2 031	2 000	571
B. Thermal output HLW (200 y)	W/TWhe	591	506	337	19.3
B/A		3.57	4.01	5.93	29.59
A/A for Scheme 1a		1	0.963	0.948	0.271
B/A for Scheme 1a		0.280	0.240	0.160	0.009

Table 5.4 shows that if the cooling time is extended from 50 to 200 years, the thermal output of the spent fuel is reduced by a factor 3.6 while for Scheme 3cV1 the thermal output of the HLW is reduced by a factor 30. The standardised thermal outputs after 50 and 200 years cooling time (standardised by dividing by the thermal output of the reference fuel cycle 1a after 50 years cooling time) show that in the case of advanced fuel cycles the needed size of the repository can be drastically reduced. However in this case the linear relationship between the thermal output and the repository size can no longer be applied. If longer cooling times are considered, it is necessary also to reconsider the cooling time before conditioning (e.g. vitrification) of the waste. Moreover, such long cooling times might be seen as contrary to the precautionary principle due to the delay before the waste will become passively safe.

5.4.4 Assessment of long-term consequences

a) Objective and limitations of the calculations, and uncertainty

The main objective of the calculations carried out in this study is to evaluate the impact of advanced fuel cycles on the long-term consequences of the disposition of HLW in the geological repository concepts that have been developed in recent years.

Results obtained for different repositories cannot be compared with each other in particular because of the uncertainties prevailing in the assessments. In performance assessments a distinction is often made between scenario uncertainty, uncertainty in conceptual models and uncertainty in parameters. Some of the calculations presented are carried out for generic repository locations, others are site specific and the level of characterisation of the repository systems considered can be very different. No attempt was made to harmonise the treatment of uncertainty in the reported assessments.

In a safety case stochastic calculations are often performed to take account of the uncertainty in parameter values. However, as most indicators considered in the present study are relative to the value obtained for the reference fuel cycle scheme, deterministic calculations have been carried out for the impact assessments, with the exception of the repository in tuff.

b) Considered repository evolution scenarios

The calculations that are presented in the following sections were done for the expected evolution of the repository system, except in the case of the repository in salt. For the salt repository, as long as the salt dome remains intact, it is assumed that no groundwater will come in contact with the disposed waste and consequently no radionuclide would be released from the disposal system. Therefore, the consequences of the occurrence of one of the least unlikely events that can result in an intrusion of groundwater into the repository – a brine intrusion – have been chosen as basis for the analysis and an alternative less likely evolution has been studied instead of the expected one.

Human intrusion scenarios, which assume that a future human action short circuits several barriers of the repository system due to reduction of the radiotoxic inventory, have not been analysed in this study. Advanced fuel cycles might reduce significantly the long-term consequences from such scenarios.

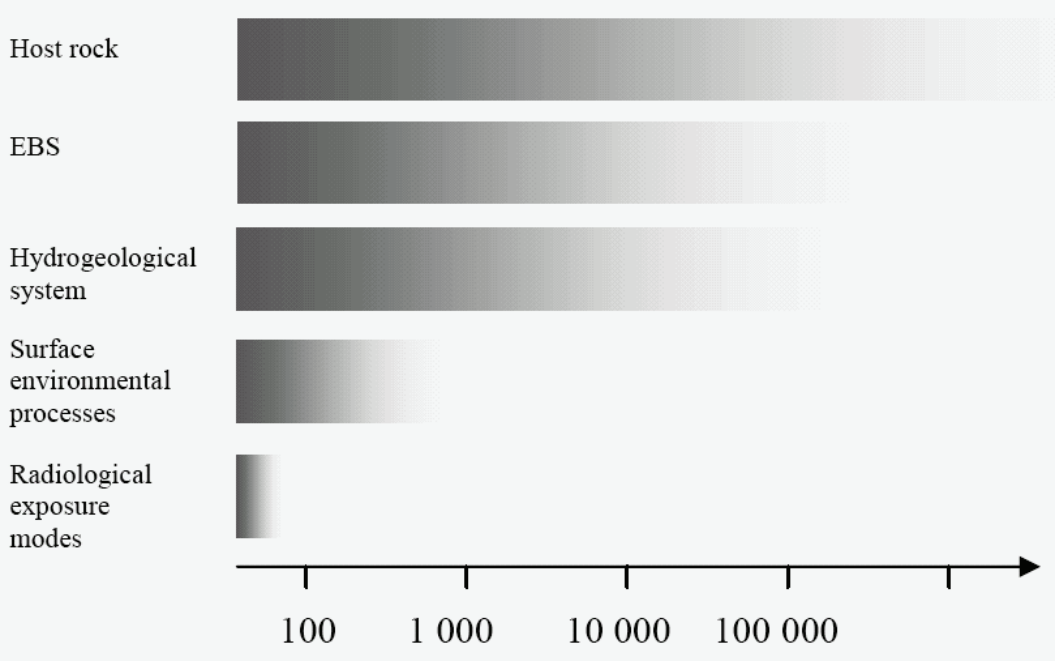
c) Indicators

The main safety indicator, which has been used for many years in safety assessments of radioactive waste disposal systems, is the annual effective dose (or dose rate) to an average member of the critical group affected by the repository [9]. To evaluate the safety of a repository the maximum dose rate can be compared with a dose constraint determined by the radiological protection authorities. The ICRP recommends an upper value for the dose constraint of 0.3 mSv/year in normal exposure situations. However, the evaluation of the safety of a repository cannot be reduced to a simple comparison of the calculated dose with the dose (or risk) constraint; the time and duration of exposure are to be taken into consideration also.

For exposure situations with a probability of occurrence lower than one, the annual risk defined as the product of the dose rate by the probability of occurrence of that situation can be used instead of dose rate.

During the time scales that have to be considered in the safety assessments of HWL repositories, various components of the repository system are expected to undergo significant changes. Figure 5.7 illustrates the limits of predictability of various components of a geological repository system [10]. It indicates that radiological exposure modes and surface environmental processes are expected to change considerably after a few decades or centuries, respectively, and that the hydrogeological system is expected to change significantly after a few thousands of years. These limits of predictability raise problems for calculating long-term doses. This has led to the development of complementary safety indicators that are independent of the biosphere and the hydrogeology [2,11]. One of these indicators is the radiotoxicity flux released from the host formation or into the biosphere and another is the integrated radiotoxicity flux over time which takes into account the duration of the radionuclide releases.

Figure 5.7. Schematic illustration of the predictability of various components of a geological repository system over time [10]



The results of the assessments are presented in details in Appendices H to K. A set of normalised (per produced TWhe) indicators has been calculated to illustrate various aspects of the long term consequences:

- radionuclide fluxes released from the host formation;
- radiotoxicity flux, together with its main contributors, released from the host formation;
- integrated radiotoxicity flux, together with its main contributors, released from the host formation; and
- dose.

For the disposal facility in the salt rock host formation only the dose (not normalised to the produced energy) is presented.

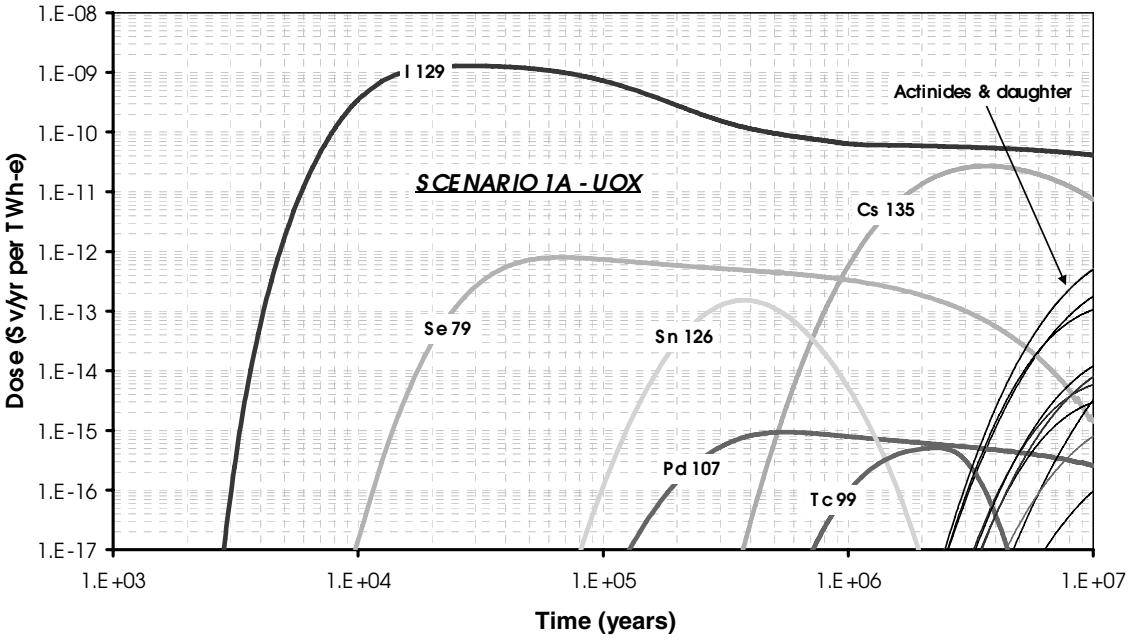
The following sections focus on the calculated doses.

d) Results of calculations for a repository in granite

The evaluations are based on the computer codes and parameter values that have been used for the assessment study of the ENRESA reference disposal concept.

Figure 5.8 shows the doses per radionuclide for Scheme 1a. ^{129}I controls the peak doses and the contribution of ^{135}Cs becomes significant after 1 million years. Doses due to actinides and their daughter products become significant only after several million years because these radionuclides are strongly sorbed in the bentonite barrier and the geo-sphere.

Figure 5.8. Dose per radionuclide in Scheme 1a



The doses per radionuclide due to the MOX fuel in Scheme 1b are similar to those in Scheme 1a. The difference in peak dose is just a factor of 2. Although the inventory of fission products per TWhe in the MOX fuel is roughly one order of magnitude smaller than in the UOX, the matrix alteration rate of MOX fuel calculated with the ENRESA model [12] is 5 times greater than for UOX fuel(due to greater alpha activity.

Figure 5.9 shows the doses per radionuclide in Scheme 1b due to the vitrified waste only, not including doses due to MOX spent fuel. Since the inventory of ^{129}I is only 0.1% of the inventory in Scheme 1a, doses due to ^{129}I are much smaller than in that scheme. Doses due to the remaining fission products are greater than in Scheme 1a because in the calculations the glass is assumed to dissolve completely in 72 000 years while the UO_2 (and even the MOX) matrices last much longer.

The doses per radionuclide for fuel cycles 2a and 3cV1 are similar to those shown in Figure 5.9 for the vitrified HLW of fuel cycle 1b. This is logical, because the glass matrix alteration model is the same in the 3 cases (1b-vitrified waste, 2a and 3cV1) and there are only small differences in the inventory of fission products (a factor of 2 at most).

Figure 5.9. Dose per radionuclide due to vitrified waste in Scheme 1b

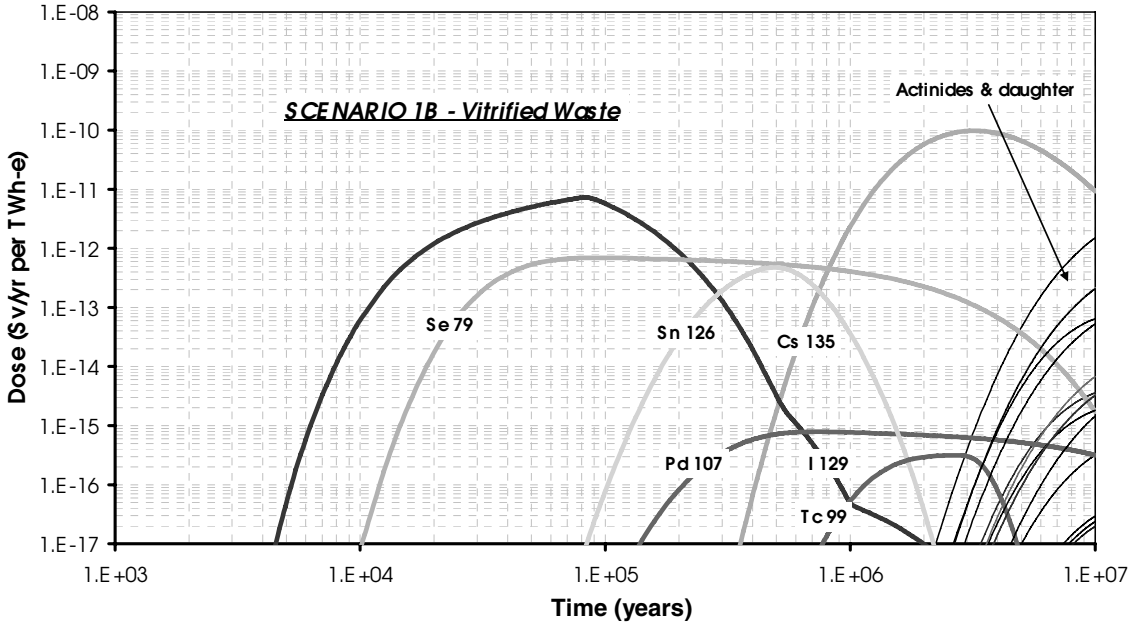
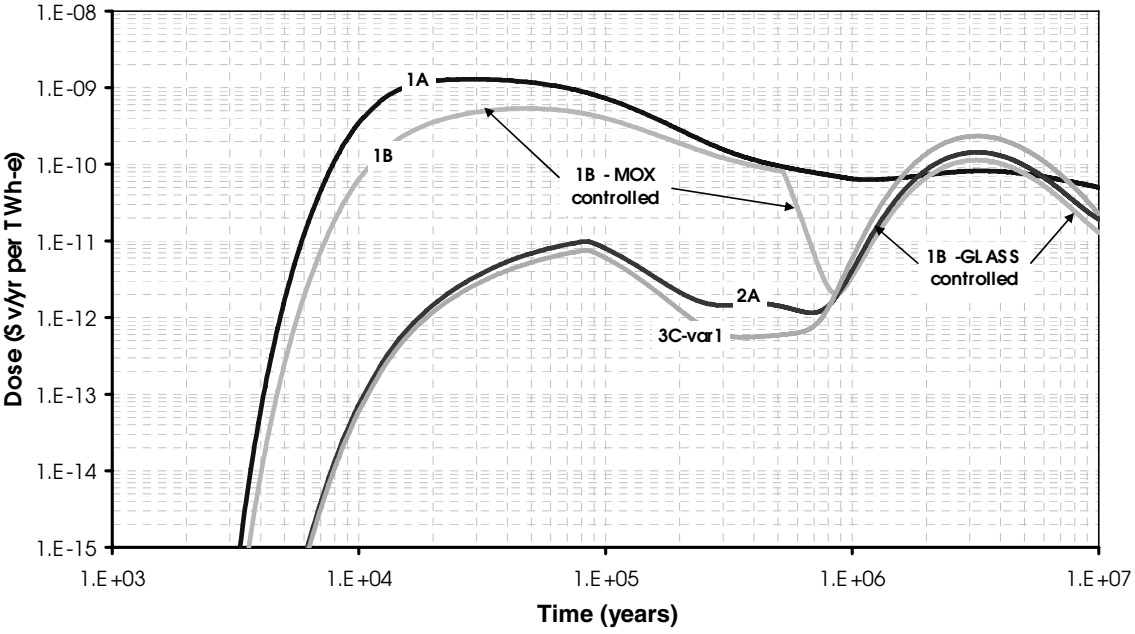


Figure 5.10 shows the total doses in the 4 schemes analysed. The curve for Scheme 1b is the superposition of two curves: one for the MOX fuel that controls the doses up to 800 000 years, and another for the vitrified waste which is dominant after 1 million years.

Figure 5.10. Normalised total doses for the different schemes



The dose curves for Schemes 2a, 3cV1 and the vitrified waste in Scheme 1b are quite similar as shown in Figure 5.9 because in the 3 schemes the same model for the glass matrix degradation is used and the inventories of fission products are quite similar. However, the maximum dose due to HLW

from Scheme 3cV1 is about 2 times higher than the maximum dose from Scheme 2a; this is due to the higher ^{135}Cs production in fast reactors than in light-water reactors. The higher ^{135}Cs production can be explained by different fission yields of ^{235}U (LWR) and ^{239}Pu (FR) and spectrum effects (^{135}Xe decays to ^{135}Cs but it can also capture a neutron; in a fast spectrum ^{135}Xe capture is lower and thus the ^{135}Cs production is higher).

e) Results of calculations for a repository in granite

The evaluations made by JNC are based on the computer codes and parameter values that have been used for the H12 report [13]. Because the disposal of spent fuel was not considered in the H12 study, JNC did not make calculations for the reference Scheme 1a, nor for the MOX spent fuel of Scheme 1b. CASE 1 corresponds to the vitrified HLW arising from the reprocessing of the uranium oxide spent fuel from Scheme 1b (excluding spent MOX fuel); CASE 2 corresponds to the HLW that will arise from Scheme 2a if the liquid waste from reprocessing of UOX and MOX spent fuels are mixed prior to vitrification; CASE 3 and CASE 4 correspond to the HLW that will arise from the separate vitrification of, respectively, the UOX and MOX spent fuels from Scheme 2a.

JNC took over a number of assumptions from the H12 study including:

- A zero remaining ^{129}I fraction in the vitrified high-level waste (hence in these calculation there was no contribution from ^{129}I to the dose from HLW).
- A 1 000 years of over-pack lifetime.
- A glass matrix lifetime of about 70 000 years.

Total doses calculated for the four considered cases have very similar values within a factor of 2 (see Figure 5.11).

Figure 5.11. Comparison of total dose

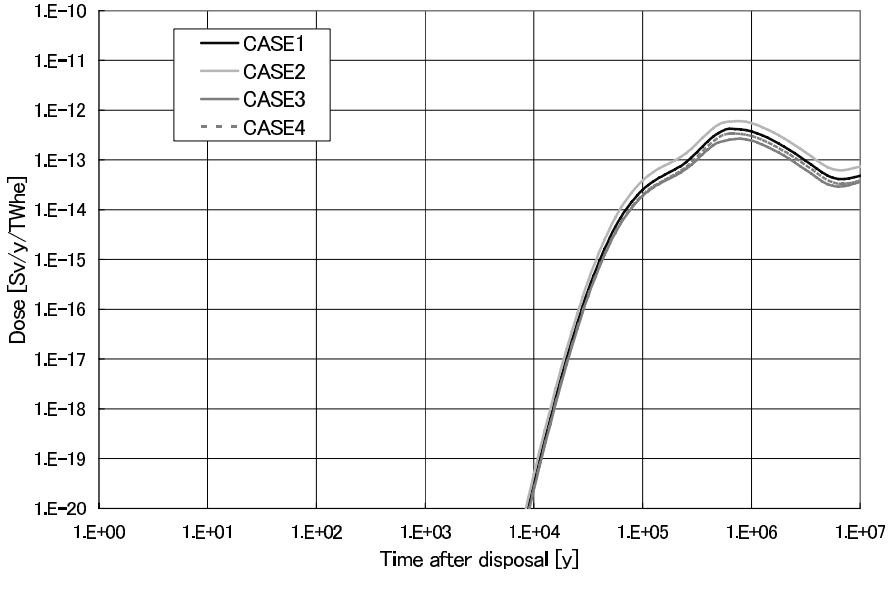


Figure 5.12 which shows the dose for each radionuclide in CASE1 (1b) is representative of the 4 cases considered.

Figure 5.12. Dose for each radionuclide for Scheme 1b

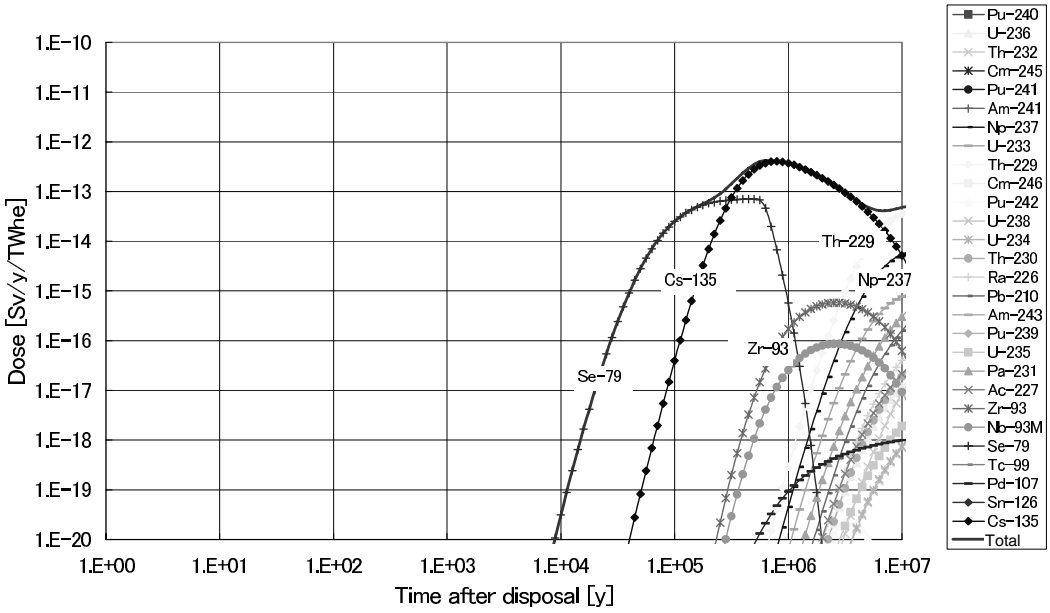


Table 5.5 shows the main contributors to the total dose for the 4 cases, ^{79}Se, ^{135}Cs and ^{229}Th; the underlined radionuclide is the main contributor to the maximum total dose which is ^{135}Cs for all cases. For ^{135}Cs which is soluble and ^{93}Zr which did not reach its solubility limit the calculated doses are proportional to the inventory. For ^{107}Pd, ^{79}Se, ^{99}Tc, and ^{237}Np and ^{238}U the solubility limits were reached. For ^{229}Th, the release rate is the same as for ^{237}Np due to radioactive equilibrium.

Table 5.5. Main contributors to the total dose

	Before 100 000 years	**From 100 000 to 1 000 000 years**
CASE1 (1b)	1st : ^{79}Se	1st: ^{135}**Cs** -> ^{229}Th
CASE2 (2a)	1st : ^{79}Se	1st: ^{135}**Cs** -> ^{229}Th
CASE3 (2a UOX)	1st : ^{79}Se	1st: ^{135}**Cs** -> ^{229}Th
CASE4 (2a MOX)	1st : ^{79}Se	1st: ^{135}**Cs** -> ^{229}Th

The importance of ^{135}Cs as main contributor highlights the value of reducing the inventory of long-lived fission products in HLW rather than reducing the inventory of actinides. Finally, it is concluded that the calculated doses are not significantly different for the four types of vitrified HLW (Cases 1-4) arising from the fuel cycle schemes considered in the study.

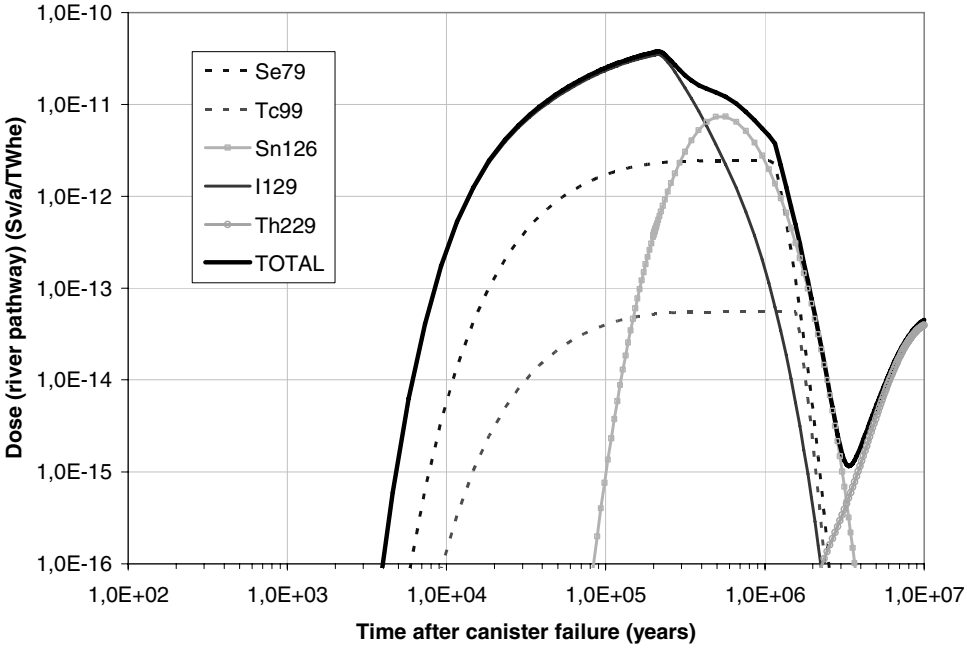
f) Results of calculations for a repository in clay

The evaluations made by SCK•CEN are based on the computer codes and parameter values that have been used for the SAFIR 2 report [5].

Figure 5.13 shows the doses per radionuclide for Scheme 1a. The total dose is dominated by ^{129}I. Smaller contributions are due to ^{126}Sn, ^{79}Se and ^{99}Tc. Doses due to actinides and their daughters become significant only after several million years because these radionuclides are strongly sorbed in the clay host formation.

The doses calculated for the disposal of the MOX spent fuel from Scheme 1b are comparable with those shown in Figure 5.13.

Figure 5.13. Total dose and its main contributors calculated for Scheme 1a



When the spent fuels are reprocessed, which is the case for Schemes 1b, 2a and 3cV1, most of the iodine escapes as gas during the fuel dissolution phase; it was assumed that 0.1 % of the ^{129}I present in the spent fuel is transferred to the vitrified HLW. For the vitrified HLW (see for example Figure 5.14, which shows the doses calculated for Scheme 3cV1) the calculated total doses are mainly due to ^{126}Sn and ^{79}Se; ^{129}I and ^{99}Tc give smaller contributions. Doses due to actinides become significant only after a few million years.

Figure 5.14. Total dose and its main contributors calculated for Scheme 3cV1

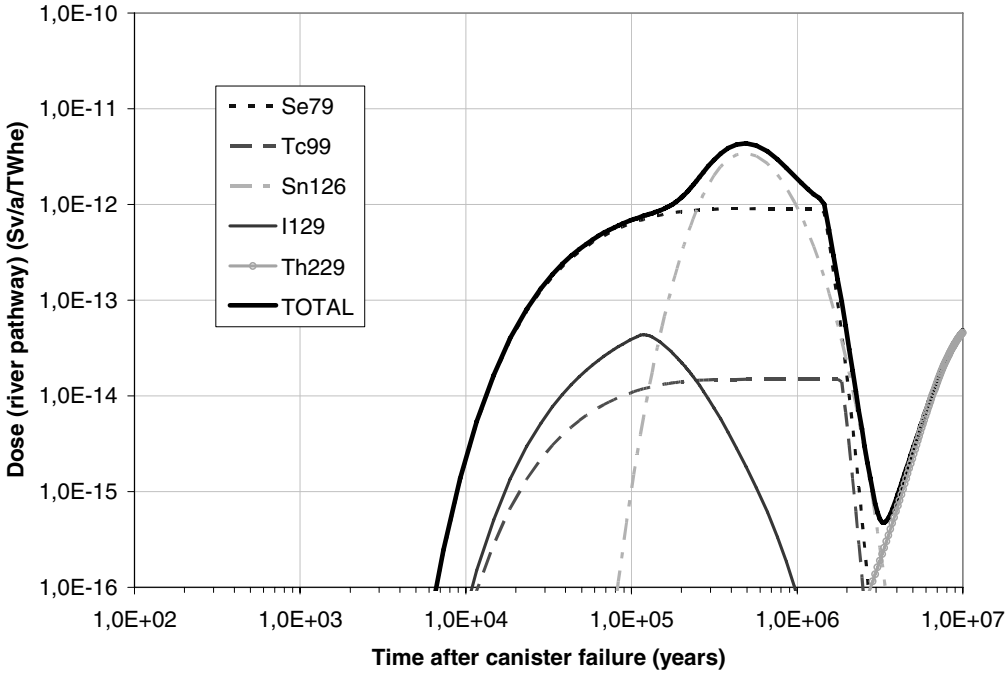


Figure 5.15. Total dose calculated for Schemes 1a, 1b, 2a, 3cV1 and 3cV1 with separation of Cs/Sr

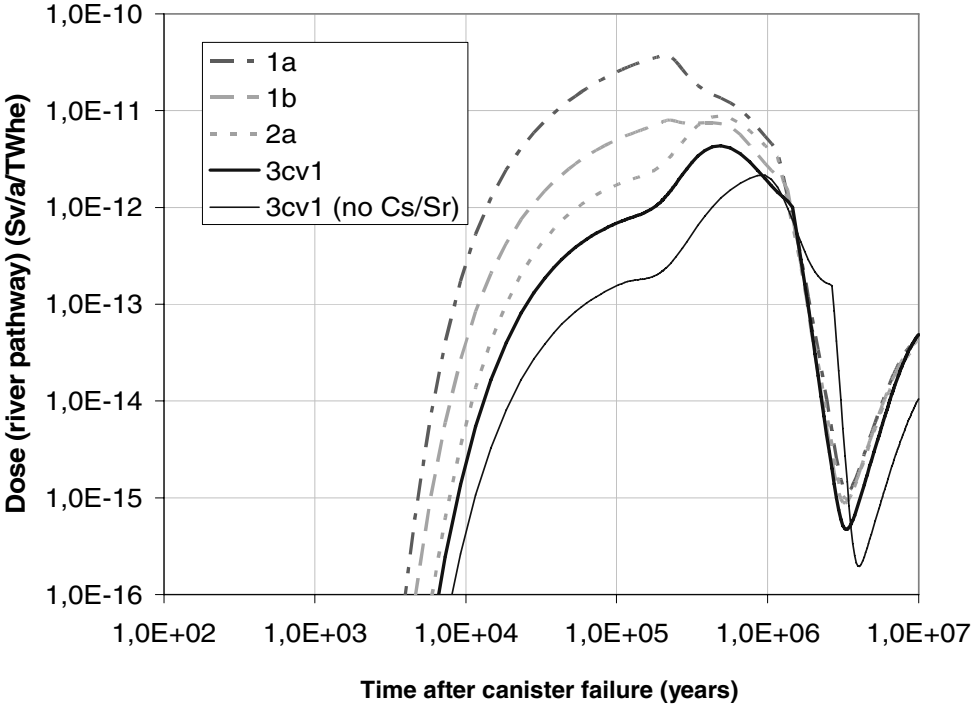


Figure 5.15 shows the total doses calculated for the 5 considered fuel cycle schemes. Scheme 1a has the highest maximum calculated dose because all the generated iodine is going into the geological repository. In the case of Scheme 1b a large fraction of the iodine was released from the HLW during reprocessing of the uranium oxide spent fuel, but the disposed MOX spent fuel still contains all the generated iodine. For the three schemes for which only conditioned HLW has to be disposed of, the maximum calculated doses have a similar shape. The calculated dose for Scheme 3cV1 is somewhat lower than the one for Scheme 2a because of the better efficiency of the GFR. The lower calculated dose for Scheme 3cV1 with separation of Cs and Sr is essentially explained by the more compact repository configuration which strengthens the contribution of the solubility limits for Se, Tc, Sn and the actinides, for example.

g) Results of calculations for a repository in salt

The results of the performance assessment carried out by GRS for a repository in salt in the cases of Schemes 1a, 2a and 3cV1 are shown in Figures 5.16 to 5.18. The calculated dose rates are extremely low. The maximum is about eight orders of magnitude below the regulatory limit recently applicable in Germany, i.e., 3 x 10^{-4} Sv per/year. The difference between the three curves is smaller than two orders of magnitude; the calculated dose is not normalised to energy produced.

Figure 5.16. Calculated dose for waste from Scheme 1a

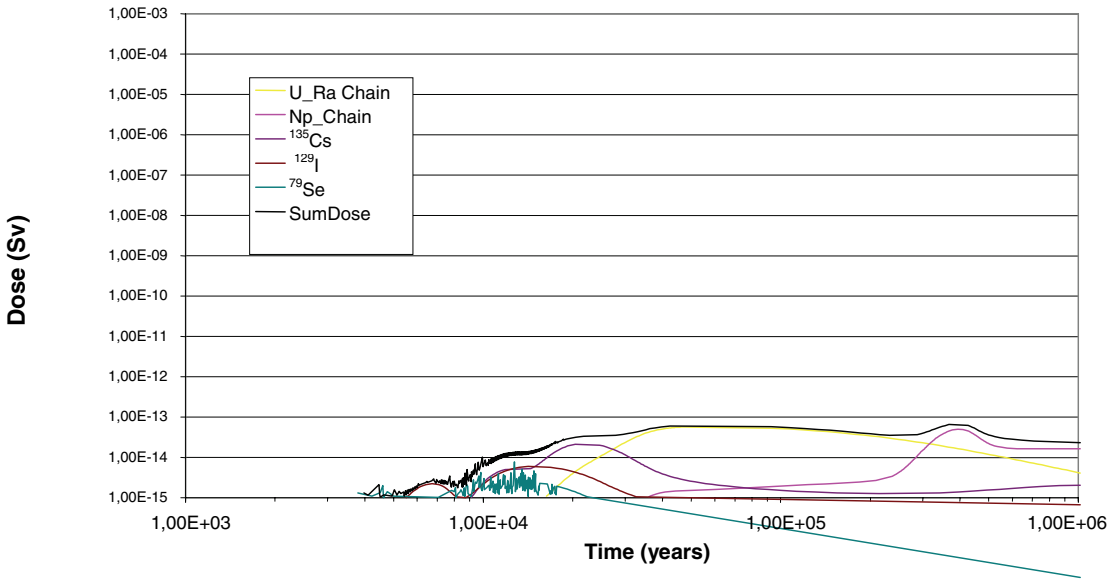


Figure 5.17. Calculated dose for waste from Scheme 2a

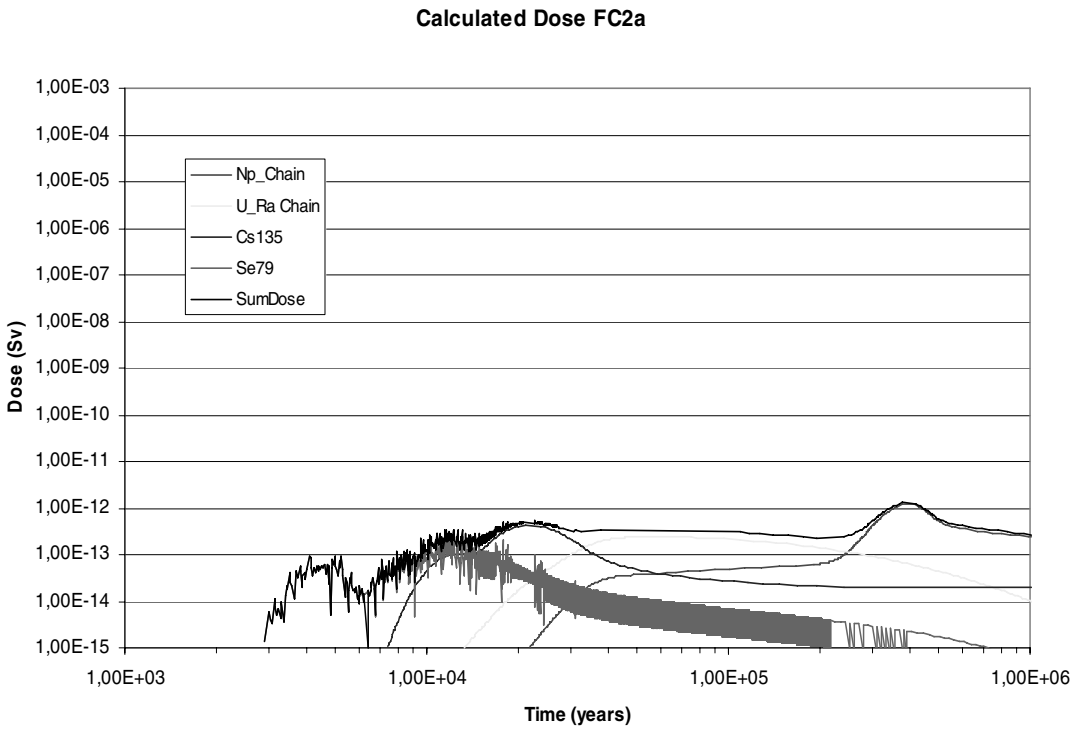
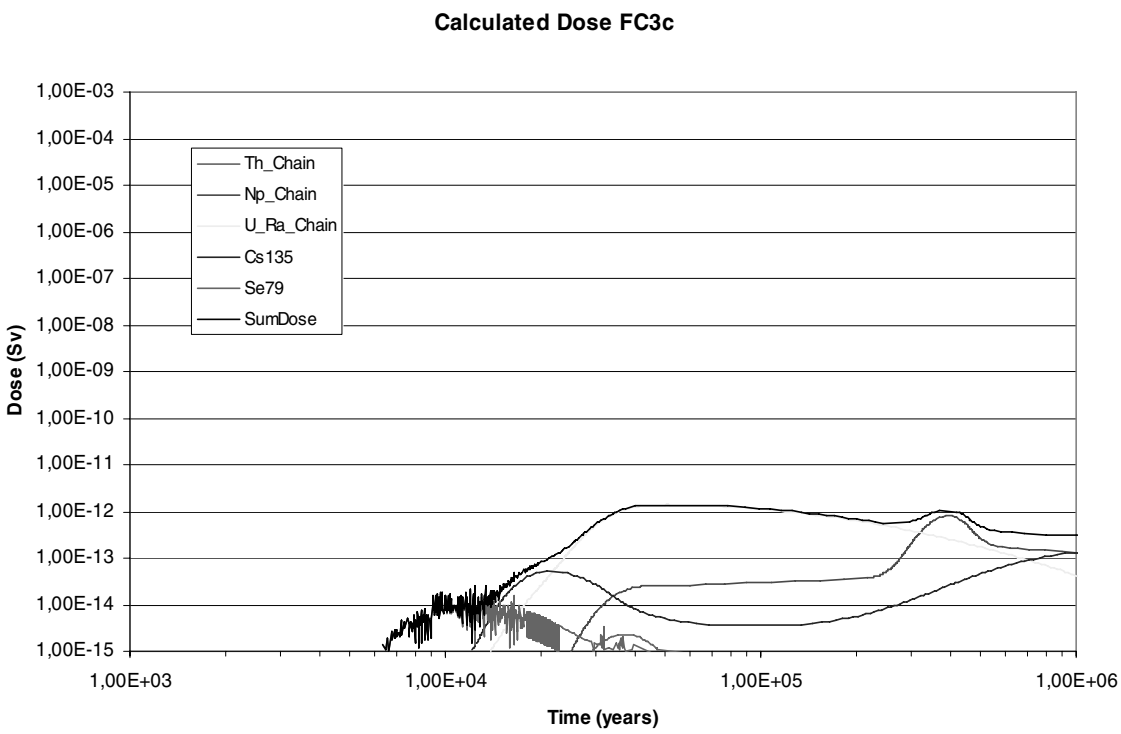


Figure 5.18. Calculated dose for waste from Scheme 3cV1



The assessments presented here are meant to explore whether disposal of the resulting waste is feasible and can be implemented in a safe manner. In addition, it might be desirable to identify advantages and drawbacks of the considered fuel cycles. The question arises, whether the radiological dose calculation described above might serve for such a comparison.

A central point for the performance assessment in salt is that the scenario under investigation is not the normal evolution but one alternative evolution scenario. It should be kept in mind that the normal evolution scenario in salt leads inevitably to zero release. Some conditions for the alternative scenario studied here have therefore been chosen arbitrarily. One of these conditions is for example the point at which the anhydrite vein is connected to the disposal field. A change of this connection point can easily result in a difference in the calculated dose of some orders of magnitude. Parameter uncertainties have been investigated in previous studies (e.g. SPA project) [14]. The parameters with the highest impact on the result are related to the description of the salt convergence (e.g. initial convergence rate, temperature, pressure). Small uncertainties on these parameters can easily result in a difference in the calculated dose of some orders of magnitude.

Considering the spread of the calculated doses due to possible uncertainties and the absence of optimisation for Scheme 3cV1 disposal facility, it can be concluded that the two orders of magnitude difference between the calculated dose curves is too small to be significant. The calculated doses can serve as indicators on whether safe disposal can be implemented for waste resulting from each scheme but cannot serve as comparative criteria.

The dose curves for the performance assessment in salt show that some radionuclides which are relevant in other host rocks, such as ^{93}Zr, ^{99}Tc, ^{107}Pd, ^{126}Sn, ^{147}Sm, Ac-Chain and Th-Chain, do not give a significant contribution to the total dose in this case.

Due to the increasing uncertainty with time, the Np-peak (after 200 000 years) is not significant. The uranium-radium decay chain is dominating the calculated total dose over a very long time for the three schemes considered. The amount of ^{238}U in the disposal facility must be reduced below the solubility limit through P&T strategies to successfully reduce the dose. The considered efficiency of the recovery of U during reprocessing (residue of 0.1%) is not sufficient to reach this objective.

Also, ^{79}Se, ^{129}I and ^{135}Cs are considered candidates for an effective dose reduction. Figures 5.16 to 5.18 show that ^{79}Se and ^{135}Cs dominate the dose at early times. In the dose curve of Scheme 1a the dose coming from ^{129}I gives a significant contribution to the total dose at early times. For Schemes 2a and 3cV1 ^{129}I does not appear in the dose curve because iodine is separated from the high-level waste flow during the reprocessing.

It has been considered to separate the heat-producing fission products – Sr and Cs – from the waste, to store them for about 300 years and to dispose them afterwards. However, this is not necessarily an interesting option for disposal in rock salt because the heat generation of the disposed waste guarantees a fast salt creep and void volume reduction.

h) Results of calculations for a repository in tuff

To evaluate the effect of Schemes 1a, 1b, 2a and 3cV1 on an unsaturated tuff repository, performance assessment calculations for radionuclide release and absorbed dose were carried out at ANL with the GOLDSIM [15] code using the "simplified site recommendation model" [16]. The results are presented in relative terms, normalised to the reference Scheme 1a, rather than in absolute terms because the simplified approach adopted is not designed to provide absolute values.

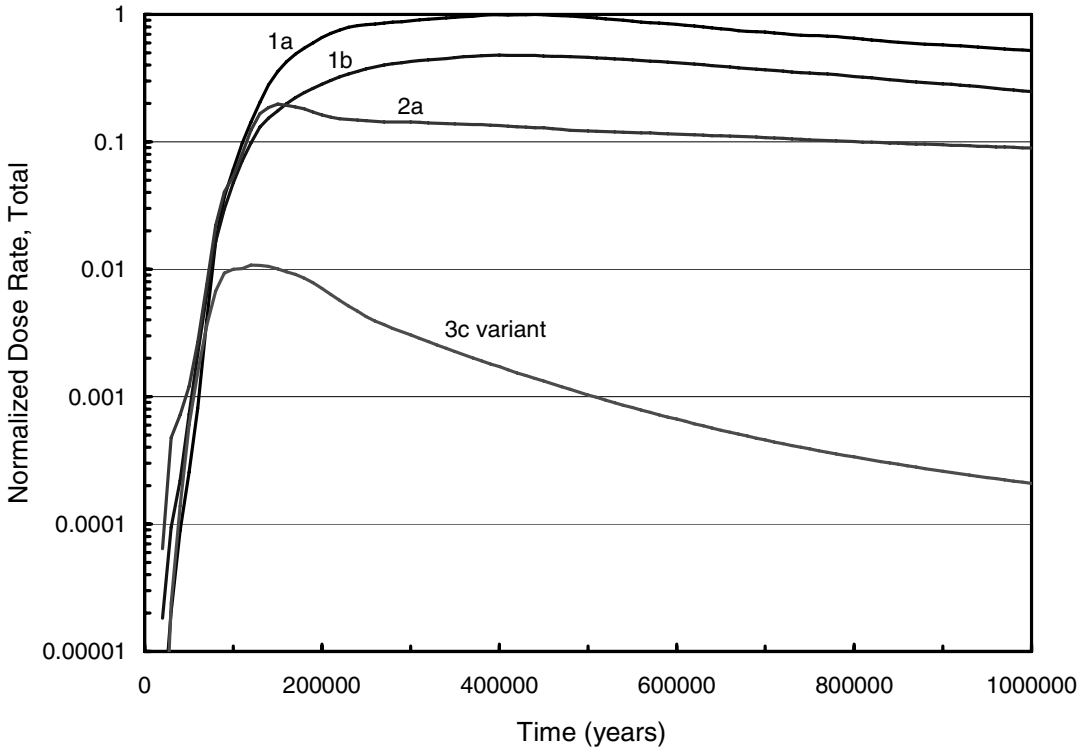
In the simplified model, waste streams make use of two types of waste packages with different performance characteristics. Spent fuel that is removed from the reactor and sent to the repository without processing is directly disposed in commercial spent nuclear fuel waste packages [17]. Alternatively, the waste stream from processed spent fuel is vitrified into waste forms made of high-level waste glass before placement in co-disposal waste packages.

The performance assessment calculations reported here utilised an area-loading pattern. Scheme 1a was chosen as the reference case. The simplified model of direct disposal allows for disposal of 70 000 Mt of commercial spent nuclear fuel (the statutory limit) within 11 770 commercial spent fuel waste packages spread over a repository footprint area of 4.02×10^6 m^2, corresponding to an average area loading of 17.4 kg/m^2. Directly substituting 70 000 Mt of spent fuel from Scheme 1a into this model, the total electrical energy generated (3.44×10^4 TWhe) spread over this same footprint results in a loading density of 8.56×10^{-3} TWhe/m^2. This value is around 34% above the conservatively-low utilisation limit calculated for Scheme 1a in Table 5.3.

For the waste streams of the remaining three fuel cycle schemes, the same total electrical generation (3.44×10^4 TWhe), repository footprint, and number of waste packages (11 770) as for the reference scheme are assumed. Also, each waste package is assumed to contain the waste from generating the same amount of electrical energy, i.e. $3.44\times10^4/11\ 770 = 2.92$ TWhe. With Scheme 1b, the vitrified waste was assigned to 10 231 co-disposal waste packages and the spent fuel to 1 539 commercial spent fuel waste packages. For Schemes 2a and 3cV1, the vitrified waste was placed in 11 770 co-disposal waste packages. In all schemes the assumed average loading density is 8.56×10^{-3} TWhe/m^2. This average loading density is slightly above the conservative utilisation limit given in Table 5.3 for Scheme 1b and well below the density for Schemes 2a and 3cV1. Thus, the following results do not reflect any increased repository utilisation (loading density) made possible in Schemes 2a and 3cV1, as the design analysis needed for repository re-optimisation is beyond the scope of this study.

Figure 5.19 shows the total dose rate for the four schemes. All the results are averages based on 1 000 independent samples of the stochastic input parameters and are normalised to the peak dose rate for Scheme 1a. The curves in Figure 5.19 indicate that Schemes 1b, 2a, and 3cV1 lower the dose rate as compared to the direct disposal case, Scheme 1a. However, the greatest reduction in the dose rate occurs for Scheme 3cV1. These results are not surprising since for Scheme 1a, the fraction of the dose rate contributed by actinides and their decay products is always greater than 90% after about 80 000 years. After 200 000 years the dose-rate contribution from actinides is essentially 100% for Schemes 1a, 1b, and 2a, and even for Scheme 3cV1 is more than 70%. During much of the time period prior to 60 000 years, the dose rate is dominated by ^{99}Tc and, to a lesser extent by ^{129}I. The peak dose rate from fission products is higher for Schemes 1b, 2a, and 3cV1 than for Scheme 1a because all or most of the fission products are in the glass waste form. In the site recommendation total system performance assessment model, the glass waste form is not as durable as directly disposed commercial spent nuclear fuel with credit taken for cladding. The fission product contribution to the dose rate is lower in Scheme 3cV1 than in Schemes 1b and 2a because of the greater thermal efficiency specified for the gas-cooled fast reactor.

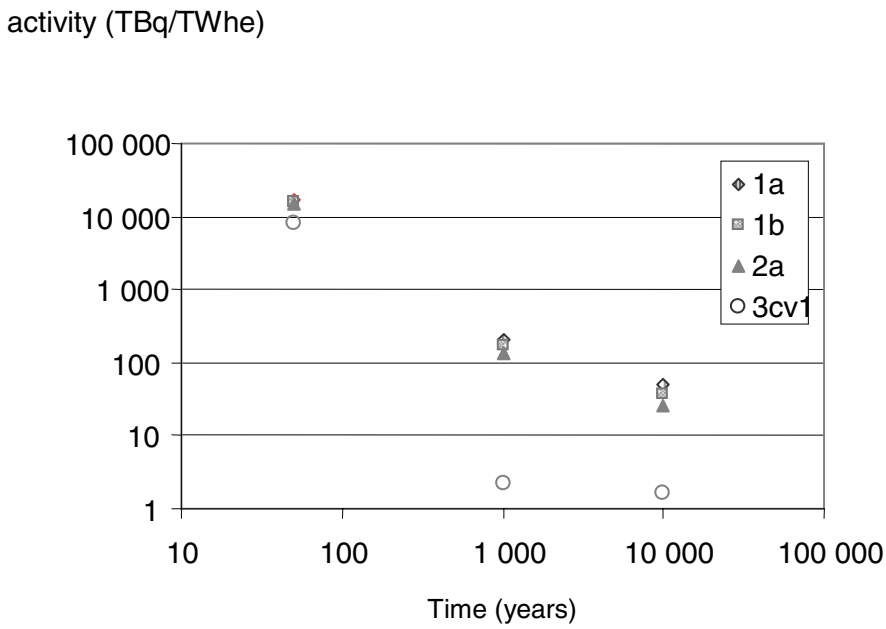
Figure 5.19. Normalised dose rates for Schemes 1a, 1b, 2a, and 3cV1 in an unsaturated tuff repository



i) Activity in the disposed waste

The above reported analyses assume that the main components of the disposal system will function as expected. In that case the disposal system confines the disposed waste in a very efficient way and the releases of radionuclides into the environment occur after long time periods and are very small. However, it cannot be ruled out that future human actions can strongly perturb and even short-circuit one or several barriers of the disposal system. The consequences of such intrusions would depend on the activity present in the disposed waste. Figure 5.20 shows the evolution of the activity in the waste as function of time. For fuel cycle Schemes 1a, 1b and 2a the activity decays by a factor of about 100 from 50 to 1 000 years; for the fully closed fuel cycle Scenario 3cV1 the activity decays by a factor of 3 500.

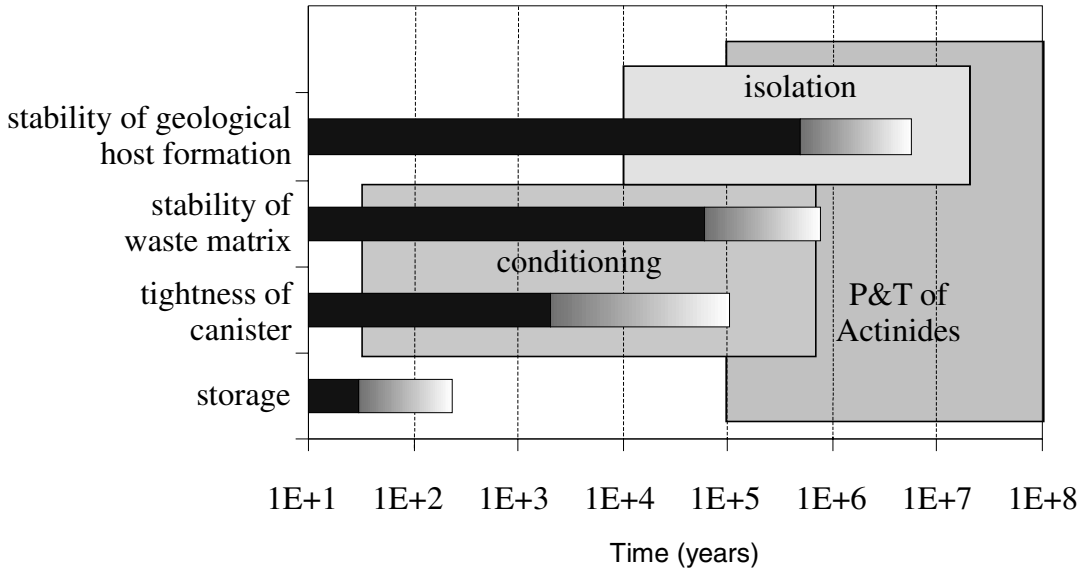
Figure 5.20. Evolution of the activity in the waste as function of time



5.4.5 Complementarities of conditioning, geological disposal and P&T in handling uncertainties

Figure 5.7 introduced qualitatively the notion of a limited predictability for every component of a geological repository. Using the same graphic support, Figure 5.21 illustrates the time range of relevance for the three major courses of action in waste management: conditioning, geological disposal, partitioning and transmutation.

Figure 5.21. Schematic illustration of the coverage of the expected reliability of the main components of a geological repository system by different elements of the waste management policies



Within the shortest time range – tens to hundreds of years – processing of spent fuel is likely to increase the risk of radiological exposure of workers. Additional measures of radiation protection (shielding, process optimisation) will keep this risk at accepted levels, inducing extra costs that are expected to be marginal. No benefit is yet brought by advanced fuel cycles, except through the possibility to isolate and store heat-producing isotopes such as ^{137}Cs and ^{90}Sr; this opens the way to optimised repository concepts and the related savings at repository construction time, i.e. within the same time frame.

The stability of geological systems (the host rock) can be assessed for hundreds of thousands of years and more. This time scale is the one for migration of the most mobile elements in all candidate host rocks (except salt, where transport of radionuclides can only occur after an accidental water intrusion). The amount of migrating radionuclides is here controlled by the waste-embedding matrix that resists to water leaching, together with the selection of a stable host rock of sufficient dimensions, homogeneity, and capability to retard radionuclide migration processes. In this context, it is a consistent finding that several fission products are more mobile than the actinides. The situation is even more contrasted for rocks where reducing conditions prevail (clay, granite); the actinides are in this case strongly sorbed and solubility limited. The radionuclide release to the biosphere in normal conditions, is dominated by the most mobile FPs, such as ^{129}I and ^{79}Se.

The great reliability of geological formations as retention media (the rock itself) becomes more delicate to assess on the very long time scales: millions to tens of millions of years. Despite the fact that the presence of the geological layers themselves is witness of their stability over hundreds of millions of years, predictions of their continued presence in the future do not exceed a few million years. At these time scales and beyond, very basic phenomena on very simple systems still can be relied upon. The decay of a radioisotope is one of those totally predictable physical behaviours, since it depends on one single parameter, insensitive to any change in the environment: the half-life of the radioisotope. Whenever no other information on the surrounding systems is available, the activity of the radionuclide source term is a good indication for the potential radiotoxicity of the remaining radionuclides. Transmutation is the sole physical process that can modify the source term.

Severe accidental situations such as human intrusion or seismic event leading to breaches into the repository are not taken into account in this study. These very unlikely events would establish pathways for radionuclide to reach the biosphere much earlier than the evolutions described here. They would increase the importance of the physical source term, i.e. the potential radiotoxicity.

Figure 5.21 illustrates in a qualitative way the decrease in reliability of components by a decrease in shade of grey; the range of action (time wise) attributed to the elements of a waste management policy appears as labelled superimposed boxes. These boxes also illustrate for any specific value of time the number of redundant barriers available to secure and immobilise the waste. It appears that P&T does cover the region of gradual decrease of reliance on chemical and geochemical means of retention, by providing means to reduce the waste before its physical emplacement in the repository.

Therefore, the various courses of action on waste management are complementary means to achieve redundant confinement of waste, either during the natural loss of predictability of the retention barriers or during their accidental disturbance: partitioning followed by transmutation, storing in surface halls, embedding radionuclides in durable matrices, conditioning in canisters, emplacing canisters in underground galleries within selected host rocks, sealing galleries with engineered barriers, all these actions contribute to devise safe ways of dealing with radioactive waste. Many technical solutions achieve fulfilment of radiation protection criteria over the decay times of the very long-lived radioisotopes.

5.4.6 Challenges

In general the disposal of HLW that is expected to arise from advanced fuel cycles appears to be less complicated than the disposal of spent fuel from the reference fuel cycle, because the disposed waste contains less fissile materials, generates less heat and emits in some cases less γ radiation and neutrons. However, some special waste types, such as targets, might not be disposable using currently considered technologies.

Spent fuels from advanced fuel cycles based on the use of FRs with high burn-up and/or ADSs might not be reprocessed with the PUREX process because the thermal output and the γ radiation of the spent fuels are too high. The reprocessing of such spent fuels may require the application of pyrochemical reprocessing. This technology is based on the dissolution of the spent fuel in molten salts, such as fluorides and chlorides. How the HLW arising from pyrochemical reprocessing will be conditioned is not yet known. It should be noted that the presence of fluorides or chlorides in HLW can strongly complicate the design of the near field barriers of the repository system. Indeed, the leakage of fluoride and chloride to the water in the near field would strongly accelerate the corrosion of many alloys that are considered in existing repository designs as candidate canister materials. It will be necessary to try to separate fluorides or chlorides from the HLW prior to conditioning in a solid matrix or to develop a matrix material that can guarantee a stable inclusion of F or Cl during very long time spans.

There is, at present, a lack of information on the intermediate level waste. Further work in that field is needed to complete that assessment of the impact of advanced fuel cycle schemes on overall waste management policies.

5.5 Conclusions

The HLW arising from advanced fuel cycle schemes generates less heat than the spent fuel arising from the reference Scheme 1a. For disposal in hard rock, clay and tuff formations the maximum allowable disposal density is determined by thermal limitations. Especially in the case of a fully-closed fuel cycle, the considerably smaller thermal output of the HLW after 50 years of cooling allows a significant reduction in the total length of disposal galleries needed. Separation of Cs and Sr allows for further reduction in the HLW repository size. For example, in the case of disposal in a clay formation the required length of the HLW disposal galleries is reduced by a factor of 3.5 for the fully-closed Scheme 3cV1, as compared with Scheme 1a, and by the reduction factor reaches 9 when Cs and Sr are separated. The separated Cs will include not only the short-lived ^{137}Cs but similar amounts (in mass) of the long-lived ^{135}Cs that should be managed somehow.

For advanced fuel cycle schemes, extending the cooling time from 50 to 200 years will result in a drastic reduction of the thermal output of the HLW and, consequently, of the required size of the repository.

In the case of disposal in rock salt the heat generation of the disposed waste contributes to a fast salt creep and void volume reduction. Therefore, a lowering of the thermal output of the HLW forms necessitates an optimisation of the waste packages and of the disposal configuration.

The increase in size of any ILW repositories due to the reprocessing of spent nuclear fuel or separation of Cs or Sr which has not been considered in this study deserves further investigation.

For all the repositories considered the maximum dose resulting from the disposal of HLW from the fuel cycle schemes evaluated does not change significantly. The dose reduction factor resulting from reprocessing is at most 8 and mainly results from the removal of ^{129}I from the liquid HLW during reprocessing. Should ^{129}I be captured and disposed of in the HLW repository, the doses resulting from all schemes would be about equal.

The releases of solubility-limited fission products are somewhat lower in the case of advanced fuel cycle schemes, because the denser repository configuration amplifies the contribution of the solubility limitation to the repository performance.

Overall, disposal in reducing conditions results in very late, i.e. after a few million years, and very small releases of actinides into the biosphere. Reducing conditions exists in most hard rock or argillaceous candidate host formations.

In the very long-term, i.e. after a few million years, the total dose is somewhat lower in the case of the fully-closed fuel cycle schemes, because much smaller amounts of actinides have to be disposed of in the repository. However, this effect is not proportional to the amount of disposed actinides because most of them are strongly solubility limited.

The activity of the HLW arising from advanced fuel cycles decreases faster than that of the reference fuel cycle. This can considerably limit the eventual consequences of less probable repository evolution scenarios, such as future human intrusions, that might strongly perturb the functioning of the HLW repository.

Advanced fuel cycles based on fast reactors or accelerator driven systems may require the application of pyroprocessing. The composition of the conditioned HLW forms that will arise from pyroprocessing is not yet known. The presence of corrosion accelerating substances, such as fluorides or chlorides, can complicate the design of an appropriate engineered barrier system. There is also currently a lack of information on the amount and characteristics of LILW types that will arise from pyroprocessing.

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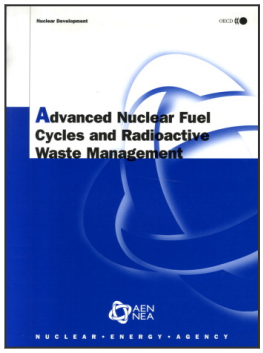

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