

## **Chapter 6 Evaluation aimed at optimizing waste disposal**

### **6.1 Basic concept for rationalization**

Following the publication of the 1st TRU progress report in March 2000, the Advisory Committee on Nuclear Fuel Cycle Backend Policy of the Atomic Energy Commission of Japan produced a report on basic considerations of processing and disposal of radioactive waste that contains transuranic nuclides (AEC, 2000). This report stated that knowledge of test data and specific phenomena are important for constructing an assessment model aimed at rationalization and refinement of repository design and improvement of the reliability of safety assessment. This section evaluates how the disposal method for TRU waste can be optimized.

The responsibility of the waste producers to reduce the quantities of waste generated is stated as a general principle of rationalized radioactive waste management by the Atomic Energy Commission and organizations concerned with long-term planning and decision-making associated with nuclear energy production. These bodies also point out that, in order to treat the waste appropriately, it is necessary to consider safe performance, economic factors, the radioactive levels and types of radioactive materials included, so that the waste can be classified appropriately and managed and disposed of in an optimized manner. Hence, it is necessary to consider whether or not to adopt managed disposal at shallow or intermediate depths, or to implement deep geological disposal to ensure isolation from the human environment and to prevent the future influence of human activities.

The sources of radioactive waste are nuclear power generation and reprocessing. It will be necessary to consider appropriate disposal of TRU wastes with wide ranges of nuclide concentrations (cf. Figure 2.3-1). Since Japan is small country, it is considered that repository siting is an important issue and the present disposal concept (clearance, shallow disposal, intermediate-depth disposal, geological disposal) must be implemented effectively and flexibly in order to ensure the safety of the human environment.

Taking these issues into account, this chapter evaluates the validity of several approaches to rationalization, taking into account the quantities of waste material and the disposal concepts presented in Chapters 2 to 5.

The rationalized concepts considered depend on optimizing various factors relating to the waste package, the engineered barriers, the natural barrier and the biosphere that influence exposure of the public (for example quantities of waste, reduction of the amount of radioactive material, improvements in barrier performance). Additionally, as a prerequisite for ensuring safety, irrespective of waste producer and waste origin, the combined efficiency of waste treatment, disposal, investigation work and facilities must be considered. In the latter case, laws must be developed and implemented flexibly. To do this, it will be necessary for the national government to consider applicable policies and institutional frameworks.

Section 6.2 considers the co-location disposal of HLW and TRU wastes, or in other words geological

disposal of these wastes at the same site. It is considered that this concept for geological disposal would result in optimized investigations for selecting a disposal site, sharing of parts of the disposal facility and efficient employment of responsible personnel. Internationally, co-location disposal is considered not only for economic reasons, but also for many other reasons such as the reduction in the number of repositories. The economic rationale for co-location disposal is described a report by FEPC (2003). Hence, in this report, the interaction between HLW and TRU waste are described.

In Section 6.3, the upper limits of concentrations for waste targeted for disposal at intermediate depth and the classification of  $\alpha$ -emitting nuclide concentrations are evaluated. The quantities of waste for intermediate-depth disposal or geological disposal change according to the values used for classification. There are several concepts for defining these classification values. Variations in the amounts and their effects on safety assessments are evaluated by specifying the current dose baseline.

Section 6.4 considers the adoption of alternative approaches for classifying the different forms of waste that are will be returned from foreign reprocessing plants. The idea is that the efficiency of international shipment and storage would be improved by significantly decreasing the quantities of waste and by flexibly designing the layout of the disposal facility. This report evaluates the classification of the amounts of waste in each disposal concept and the effects on safety assessments.

## **6.2 Evaluation of co-location disposal with high-level waste**

In this report, in order to specify a concept for co-location disposal of TRU waste and high-level waste (HLW), interaction between TRU waste disposal facilities and HLW disposal facilities are evaluated by referring to co-location disposal concepts that have been proposed in various countries. To summarize this latter work, consideration is given to thermal effects, organic material, nitrate and a high-pH plume in a general geological environment. Based on the results, the required isolation distance and necessary engineering measures are evaluated and the layout of the disposal facilities is presented. Issues for future consideration are identified and summarized.

### **6.2.1 Disposal conditions specified in other countries**

Countries that are planning, or that have considered, repositories for radioactive waste equivalent to TRU waste include the UK, France, the US, Germany, Switzerland, Belgium, the Netherlands and Spain. Summaries of the repository plans in Switzerland, France, Belgium, Germany and the US are shown in Table 6.2.1-1.

The plans in France and Switzerland are included as representatives of established disposal concepts for TRU waste and HLW. In France, the disposal facility is planned to be partitioned for separate disposal of

TRU waste, HLW and spent fuel. This plan avoids interaction between each waste group. However, there is no detailed description (ANDRA, 2001).

In Switzerland, a disposal facility for long-lived intermediate-level radioactive waste (ILW) is planned to be constructed at a distance of several 100 m from the repository tunnels for HLW and spent fuel. The possibility that a high-pH plume from the long-lived ILW repository tunnels will affect the repository tunnels for HLW and spent fuel is thus prevented (Nagra, 2002) (cf. Figure 6.2.1-1).

The issues presented below, which have been considered in Switzerland, Sweden, France and the UK, are given as examples of interaction between the co-location disposal:

- ① The following 4 items are issues that have been evaluated in connection with co-location disposal in each country. However, detailed results of these evaluations have not been published.
  - Heat generation from HLW or spent fuel
  - Organic material in TRU waste
  - Degradation products in TRU waste
  - a high-pH plume
- ② In Switzerland, it is considered that the host rock will be altered by the high pH-plume originating in the cementitious backfill material and structural components in the tunnels for long-lived ILW. It is also considered that such a plume would alter bentonite in the tunnels for HLW and spent fuel. Hence, in crystalline bedrock, tunnels for long-lived ILW and tunnels for HLW and spent fuel would be isolated using fractures in the host rock. In relatively homogeneous sedimentary rock, it is considered that separations of several 100 m would be sufficient to rule out interaction of these wastes (Nagra, 1994; Nagra, 2002).
- ③ In Sweden, heat from the SFL2 spent fuel disposal facility, corrosion products from SFL3-5 wastes and organic materials from SFL3 wastes have been considered. A separation distance of 1 km would ensure isolation of the wastes and prevent reciprocal interactions occurring between SFL2 and SFL3-5. The tunnels connecting each facility would be isolated using backfill and plugs. It is also planned to construct the facility for SFL3-5 downstream of SFL2 (SKB, 1999).
- ④ In France, in order to decrease interaction of TRU waste, HLW and spent fuel, disposal areas would be separated by several 100 m to ensure isolation of the wastes. However, the details have not yet been reported (ANDRA, 2001).
- ⑤ In the UK, the influence of a high-pH plume from intermediate- and low-level waste on HLW and spent fuel has been considered. A separation distance between the two facilities of 500 m was considered (King and Poole, 2002).

Table 6.2.1-1 Disposal concepts for radioactive waste in other countries

Country	Switzerland	France	Belgium	Germany	America
Disposal concept	<u>Co-location</u> Long-lived intermediate-level waste* High-level waste (vitrified waste and spent fuel)	<u>Co-location</u> Category-B waste* High-level waste (vitrified waste forms and spent fuel)	<u>Co-location</u> Category-B waste* Some Category-C waste* High-level waste (vitrified waste forms and spent fuel)	<u>Co-location</u> Heat-producing waste* Some non-heat-producing waste* High-level waste (vitrified waste forms and spent fuel)	<u>Sole disposal</u> TRU waste*
Lithology, depth	Opalinus Clay: about 650 m Crystalline rock: about 1000 m	Sedimentary rock: about 500 m Crystalline rock: undecided	Boom Clay: about 250 m	Undecided (site selection is being reconsidered.) (In the case of Gorleben the salt formation is at depths of 840 to 1200 m)	In operation (near Carlsbad, New Mexico) Salt formation: depth about 650 m
Layout	Within the same formation, there is a distance of about 500 m between the area with high-level waste and the area with TRU waste (see referenced figure)	Within the same formation, the area with high-level waste and the area with TRU waste are separated (the distance between the areas will be considered later)	Within the same formation, the area with high-level waste and the area with TRU waste are separated (the distance between the areas will be considered later)	Undecided	-

Switzerland: Kristallin-I, NTB 93-09E, May 1994, Nagra.

Project Opalinus Clay, Safety Report, NTB 02-05, December 2002, Nagra.

France “DOSSIER 2001 ARGILE: Progress Report on Feasibility Studies & Research into Deep Geological Disposal of High-Level, Long-Lived Waste (Synthesis report)” (ANDRA, 2001)

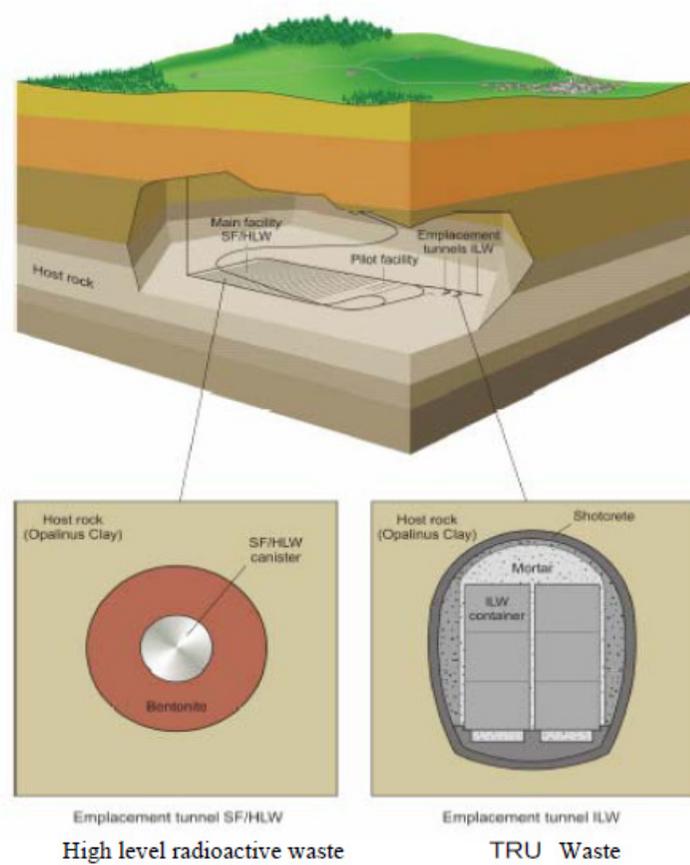
Belgium Safety Assessment and Feasibility Interim Report 2: SAFIR2, ONDRAF/NIRAS (2001)

Germany “Environmental Policy: Joint convention on the safety of spent fuel management and on the safety of radioactive waste management”.

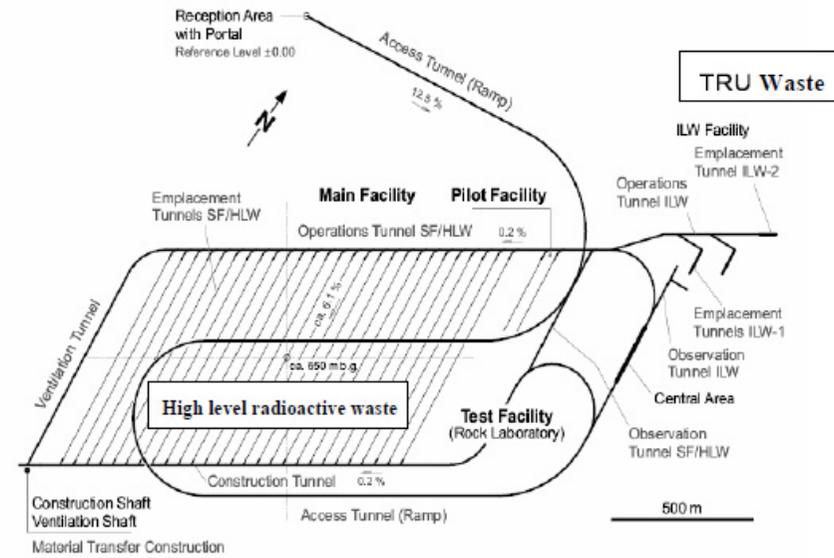
BMU (2003) DBE Web Site: <http://www.dbe.de/>

America WIPP Web Site: <http://www.wipp.ws/>

\*Corresponds to the TRU waste that is assumed for geological disposal in Japan.



Cross section of geologic repository in opalinus clay formation



Layout of geologic repository in opalinus clay formation

Figure 6.2.1-1 Concept for the disposal of HLW, spent fuel and TRU waste developed in Switzerland (Nagra, 2002)

## 6.2.2 Evaluation of the influences of interactions

### 6.2.2.1 Elicitation and classification of influencing factors

Assuming that geological disposal facilities for TRU waste and HLW will be established at the same site, interactions between the TRU waste and HLW, and their potential effects on safety, were identified and summarized.

#### (1) Summary of basic mechanisms of mutual interactions

Mutual interactions considered to have potentially large effects are taken to be the most important.

##### ① Temperature

Direction of effect : HLW disposal facility → TRU disposal facility

Basic mechanism : Disturbance of the geological environment / propagation of energy

Interaction mechanism : Under the disturbed environmental conditions that depend on the different thermo-physical properties of the different rock zones around the HLW facility, spatio-temporal variations in temperature near the TRU waste facility will occur.

##### ② Hydraulics

Direction of effect : HLW disposal facility ↔ TRU disposal facility

Basic mechanism : Disturbance of the geological environment

Interaction mechanism : Under the disturbed environmental conditions that depend on the different hydraulic properties of the rocks around the HLW repository and TRU waste repository, groundwater flow will be affected over a wide area around the HLW / TRU waste disposal facilities.

##### ③ Stress

Direction of effect : HLW disposal facility → TRU disposal facility

Basic mechanism : Disturbance of the geological environment

Interaction mechanism : Under the disturbed environmental conditions that depend on the different mechanical properties of the rocks near the HLW disposal facility, the stress in the vicinity of the TRU waste facility is affected.

##### ④ Chemistry

Direction of effect : HLW disposal facility ← TRU disposal facility

Basic mechanism : Disturbance of the geological environment, mass transport

Interaction mechanism : Under the disturbed environmental conditions that depend on the different chemical characteristics of the rocks near the TRU waste disposal facility, spatio-temporal variations in chemical conditions around the HLW facility occur.

These effects include high pH, organic material, nitrates, cement colloids and gas originating in the TRU waste disposal facility.

#### ⑤ Radiation field

In the separate disposal facilities for TRU waste and HLW, it is considered that there are no obvious effects of radiation. Therefore, interaction connected with the radiation field are considered to be small or negligible.

If the HLW and TRU waste disposal facilities are to be constructed at the same site, the above range of assumed interaction mechanisms are included in scenario analyses. However, the purpose of evaluating interaction mechanisms is to summarize circumstances where there remain many uncertain factors, such as site conditions, layout of the disposal facility, engineered barrier specifications and waste specifications. The aim is to support decision-making connected with establishing a repository and not to evaluate quantitatively all the interaction mechanisms mentioned previously. Instead, the potential influence mechanisms in the previous section were screened and items for quantitative evaluation in the next section were selected.

The mechanisms that perturb a geological environment depend strongly on the geological conditions of a site (stratigraphy, presence or absence of faults, anisotropy, inhomogeneity, etc.). These issues might be important in performance assessments aimed at co-disposal. However, since the site has not yet been decided, these effects are difficult to evaluate quantitatively. Therefore, the nuclide migration distance is conservatively specified to be 100 m so that these effects are excluded.

The effect on an HLW disposal facility of mass transfer by cement colloids originating in the TRU waste disposal facility is also small. If the analysis of the high-pH plume ensures a sufficient isolation distance, the effect of cement is sufficiently covered.

Since the effects of gas generation on the engineered and natural barriers are assumed to be small, gas migration in the HLW disposal facility and TRU waste disposal facility and in the natural barrier is excluded.

Based on these evaluations, the following 4 items are identified and evaluated in the next section.

- ① Heat transfer into the TRU waste disposal facility from the HLW disposal facility
- ② Migration of organic material from the TRU waste disposal facility into the HLW facility and the influence of this material on the HLW disposal facility
- ③ Migration of nitrate from the TRU waste facility into the HLW facility and the influence of these salts on the HLW disposal facility
- ④ Migration of a high-pH plume from the TRU waste facility into the HLW disposal facility,

and the influence of the plume on the HLW disposal facility

### 6.2.2.2 Thermal effects

#### (1) Quantities of heat generated by the waste

The rate of heat generation by vitrified HLW is 100 times higher than that from hulls and ends, which have the highest heat generation rates among the TRU wastes, as shown in Figure 6.2.2.2-1.

In a TRU waste disposal facility, the number of waste packages containing hulls and ends is estimated to be about 29,000. This number is much smaller than the 40,000 vitrified waste forms that would be employed in a HLW disposal facility. Therefore, the heat transfer rate and temperature gradients from the HLW disposal facility are considered to affect the TRU waste facility. Heat transfer rate in the opposite direction is not considered.

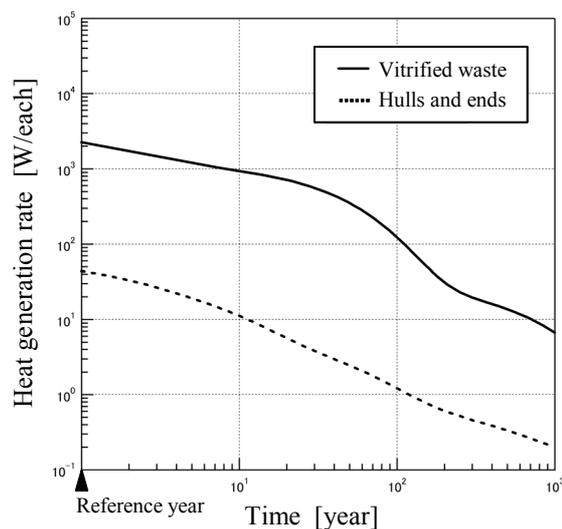


Figure 6.2.2.2-1 Temporal changes in heat generation rates in vitrified waste and in waste packages containing hulls and ends

#### (2) Analytical system and model

The thermal analysis considers both crystalline bedrock and sedimentary rock. The aim is to examine the significance of different configurations of the HLW disposal facility and the differing thermal properties of different host rocks.

The analysis of heat transfer considers 3 types of mechanism: conduction, convection and radiation heat transfer. However, in the geological disposal environment, since there is always basement rock present and the groundwater flow velocity is sufficiently small, heat conduction due to the thermal gradient generated by HLW is considered to dominate. Additionally, since the HLW disposal facility will have dimensions of 2 km in four directions, the facility was represented as a thin plate and a 2-dimensional analysis was carried out.

Hence, this analysis assumed the system shown in Figure 6.2.2.2-2. The heat conduction analysis code TRUMP was used to simulate heat transfer to the TRU waste disposal facility from the HLW disposal facility.

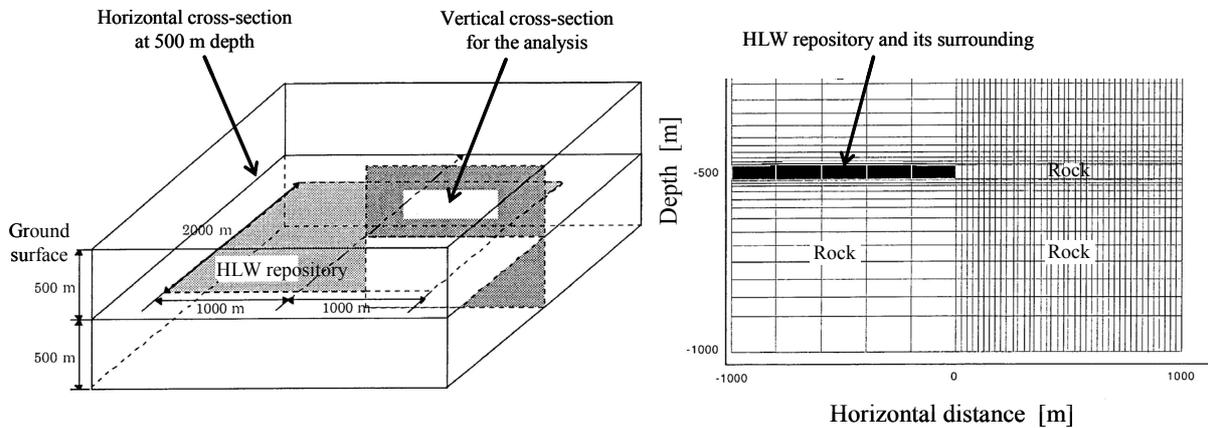


Figure 6.2.2.2-2 Heat conduction analysis model

Additionally, the boundary conditions are specified to be constant temperature (surface temperature: 15°C; temperature at 500 m underground: 30°C; geothermal gradient: 3°C/100 m). It is assumed that an insulating boundary occurs horizontally at a distance of 2,000 m from the center of HLW facility. The values of thermal properties of the engineered barriers and host rocks in this thermal analysis are shown in Table 6.2.2.2-1.

Table 6.2.2.2-1 Thermal properties of the engineered barriers and host rock

Material	Heat conductivity (W/m · K)	Specific heat (J/kg · K)	Unit weight (Mg/m <sup>3</sup> )	Porosity (%)
Vitrified waste	1.2	960	2.8	—
Overpack	51.6	470	7.8	—
Buffer material	0.78	590	1.7	—
Backfill material	0.78	590	1.7	—
Crystalline rock	2.8	1,000	2.7	2
Sedimentary rock (SR-C)	2.2	1,400	2.7	30

### (3) Analytical results

The relationship between the increase in host rock temperature caused by the HLW disposal facility and the separation distance is shown in Figures 6.2.2.2-3 and 6.2.2.2-4. With interim storage for 50 years, at a location that is 100 m from the HLW disposal facility, the temperature increase is estimated to be below

5°C in both crystalline bedrock and sedimentary rock. In the case where the separation is 300 m, the increase is estimated to be below 1°C.

#### **(4) Effect of heat on engineered barrier performance**

If the temperature of the engineered barriers increases as a result of heat from the waste, parameters that affect the migration of radionuclides, such as their solubility, distribution coefficient and diffusion coefficient, may change. In order to prevent alteration of the buffer material in a HLW facility, the maximum temperature is specified to be less than 100°C. Experiments and natural analogue studies show that, if the temperature is not over 100°C, there will be no mineralogical change of smectite, which is the main mineral composing the bentonite (JNC, 2000b).

Since cementitious material and bentonite will be used in the engineered barriers of a TRU waste disposal facility, the maximum temperature is specified to be 80°C. At lower temperatures, there will be no decrease in the sorption capability of radionuclides on the cementitious material. To maintain these lower temperatures, the density of emplacement of waste (hulls and ends) in the disposal tunnels and the separation distance of the tunnels are specified.

#### **(5) Effect of heat generation on natural barrier performance**

Based on a thermal analysis, if HLW is stored for 50 years, the temperature increase near the HLW disposal facility is estimated to be below 20°C. Therefore, it is considered that significant mineral alteration will not occur.

#### **(6) Summary**

The temperature increase due to heat generation in a HLW disposal facility, in either crystalline bedrock or sedimentary rock, is estimated to be below 5°C at a distance of 100 m from the HLW disposal facility. At a distance of 300 m, the temperature increase is estimated to be below 1°C. Hence, a separation distance of 100 m can prevent any temperature influence from HLW on TRU waste in the same facility.

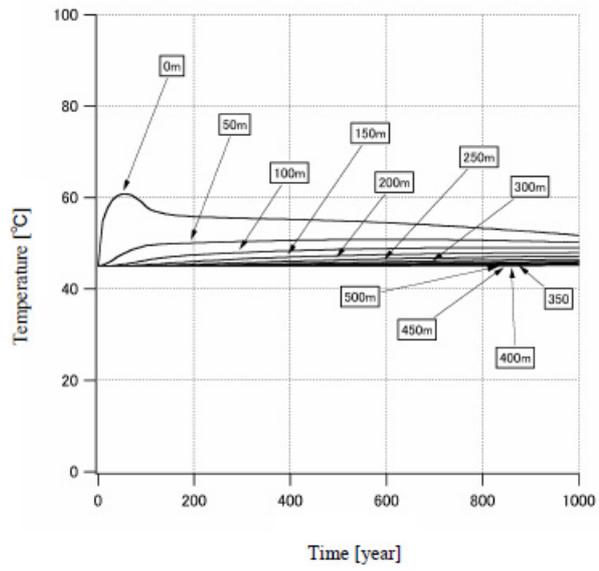


Figure 6.2.2.2-3 Temperature change with respect to separation distance and time in crystalline rock

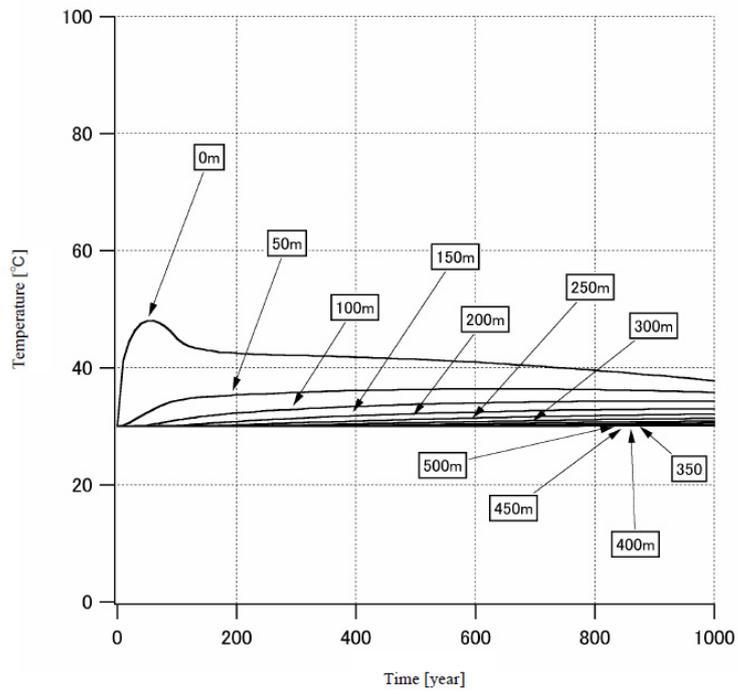


Figure 6.2.2.2-4 Temperature change with respect to separation distance and time in sedimentary rock

### 6.2.2.3 Effects of organic material

#### (1) Types of organic material in waste

The information in Chapter 2 shows that some types of organic material are included in TRU waste. However, the effects of bitumen and spent solvent in the waste on nuclide solubility and sorption behaviour are considered to be small. In this section, cellulose-type organic material, which is included in hulls and ends (JNC), is evaluated.

Based on experimental studies by AEA in the UK and PSI in Switzerland, the organic material which affects the distribution coefficient and solubility of radionuclides is considered to be Iso-Saccharinic Acid (ISA). This material is considered in the safety assessments that have been carried out in the UK, Switzerland and Sweden (Bradbury and Sarott, 1994). Basic structure of cellulose is  $(C_6H_{10}O_5)_n$  as shown in Figure 6.2.2.3-1 and ISA are its degradation products.

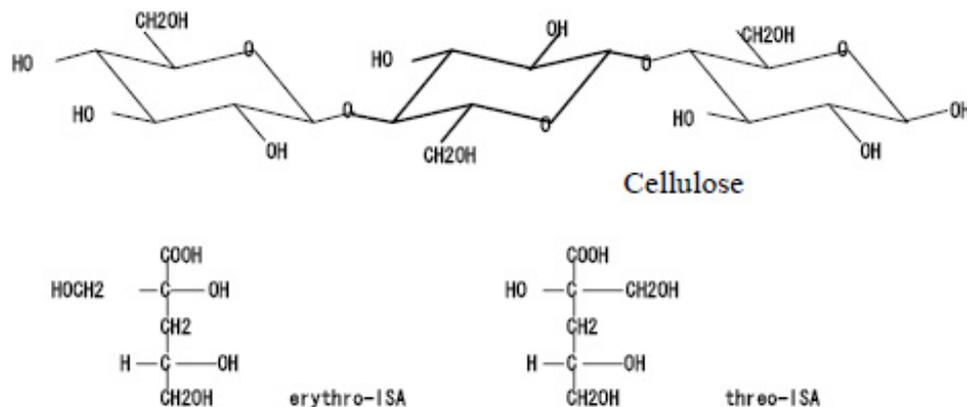


Figure 6.2.2.3-1 Chemical structures of cellulose and ISA (Greenfield and Holtom, 1995)

#### (2) Evaluation of the organic material plume

Organic material in the hulls and ends of JNC is considered to be cellulose-type organic material and the recovery rate of ISA is assumed to be 100%. The behaviour of an ISA plume in the host rock depends on the following conditions:

- ① Inventory of ISA
- ② Sorption distribution coefficient of ISA for cementitious material
- ③ Sorption distribution coefficient of ISA for the host rock
- ④ Groundwater flow direction and flow velocity in host rock

In the analysis of an organic material plume, in order to understand the significance of differences in the hydraulic properties of the host rock, crystalline bedrock and sedimentary rock were both considered. It is likely that ISA partly sorbs in the engineered barriers and partly in the natural barrier. However, in this

analysis, it was assumed that the ISA sorbs onto cementitious material, but not onto the host rock. The ISA was simply diluted by groundwater in the host rock. Since the disposal facility and the surrounding rock are treated as a two-dimensional system, the 2D mass transfer analysis code "AZURE" was used. The model used in this analysis is shown in Figure 6.2.2.3-2. The values of parameters, including the hydraulic properties of the engineered barrier and host rock, are given in Table 6.2.2.3-1.

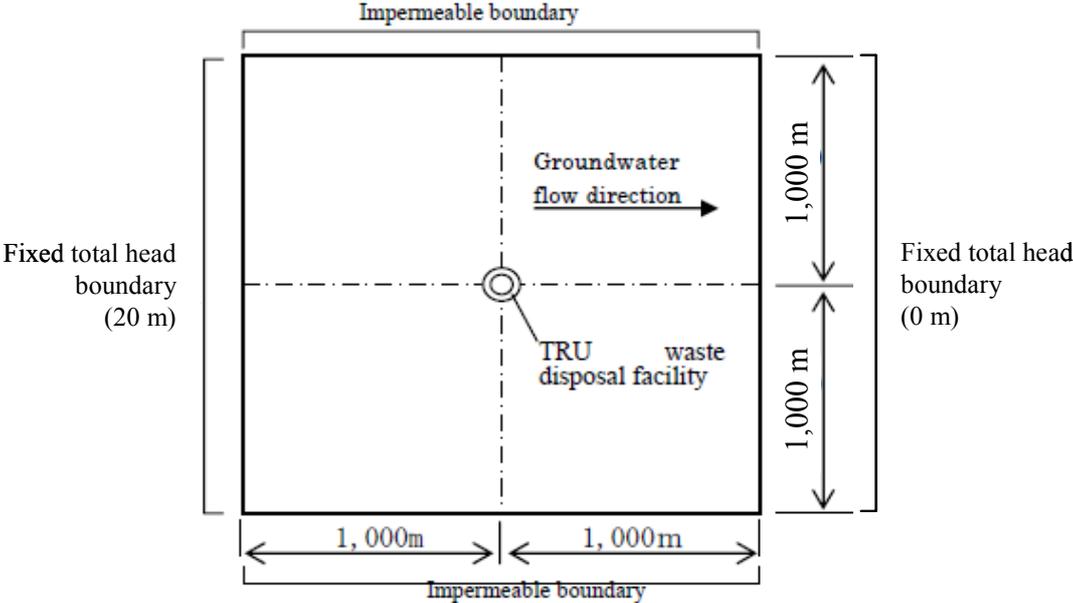


Figure 6.2.2.3-2 Illustration of the analytical model

Table 6.2.2.3-1 Parameters and values used

Parameter		Unit	Crystalline rock	Sedimentary rock
Quantity of organic material	ISA amount	kg	525 (= 0.5 kg × 1,050)	
	Hydraulic conductivity			
	Hydraulic conductivity of cement	m s <sup>-1</sup>	4×10 <sup>-6</sup>	
	Hydraulic conductivity of bentonite	m s <sup>-1</sup>	2×10 <sup>-11</sup>	
Diffusion coefficient	Effective diffusion coefficient in cement	m <sup>2</sup> /s	8×10 <sup>-10</sup>	
	Effective diffusion coefficient in bentonite	m <sup>2</sup> /s	3×10 <sup>-10</sup>	
	Effective diffusion coefficient in host rock	m <sup>2</sup> /s	8×10 <sup>-11</sup>	1.2×10 <sup>-9</sup>
Sorption distribution coefficient	Sorption distribution coefficient in cement	m <sup>3</sup> /kg	0.17	
	Sorption distribution coefficient in bentonite	m <sup>3</sup> /kg	0	
	Sorption distribution coefficient in host rock	m <sup>3</sup> /kg	0	
Form of engineered barriers	Target disposal facility	—	Waste Group 2	
	Size of disposal facility	m	8.8 Φ	11.3 Φ
Natural barrier	Hydraulic conductivity	m s <sup>-1</sup>	1×10 <sup>-8</sup> , 1×10 <sup>-9</sup> , 1×10 <sup>-10</sup>	
	Porosity	—	0.02	0.30
	True density	Mg/m <sup>3</sup>	2.7	2.7
	Hydraulic gradient	m/m	0.01	
	Dispersion length	—	1/10 of migration distance	

### (3) Analytical results

The results of the 2D mass transfer analysis for organic material are shown in Figures 6.2.2.3-3 and 6.2.2.3-4. The behaviour of the organic material plume is different in crystalline and sedimentary rock, since the porosities and effective diffusion coefficients are different. In crystalline rock, the part of the plume with a concentration of more than 1×10<sup>-6</sup> mol/dm<sup>3</sup> is limited to the vicinity of the facility. Concentration contours of 1×10<sup>-7</sup> and 1×10<sup>-8</sup> mol/dm<sup>3</sup> occur within 100 m of the disposal facility on the upstream side and at 200 and 300 m respectively on the downstream side.

In sedimentary rock, the porosity is larger and, in a given volume of rock, a greater quantity of groundwater may dilute the plume than in the crystalline rock. As in the crystalline rock, in the sedimentary rock the part of the plume with a concentration of more than 1×10<sup>-6</sup> mol/dm<sup>3</sup> again occurs within the disposal facility. However, the 1×10<sup>-7</sup> and 1×10<sup>-8</sup> mol/dm<sup>3</sup> concentration contours in the sedimentary rock occur closer to the facility, at distances of 50 m and 100 m respectively, in both the upstream and downstream directions.

#### **(4) Effects of organic material on engineered and natural barriers**

ISA functions as a ligand and its presence in pore water is changed by interaction with nuclides. As a result, an increase in solubility and decrease in distribution coefficient will occur and nuclide migration retardation function of the engineered and natural barriers might be compromised. Some experimental data in which ISA effects solubility of nuclides were reported. Based on Greenfield and Holtom (1995), the solubility of Pu starts increasing when the ISA concentration is greater than  $1 \times 10^{-6} \text{ mol/dm}^3$ , as shown in Figure 6.2.2.3-5.

Tits et al. (2002) evaluated the effect of ISA on sorption of Eu(III) and Th(IV) on calcite. In this report, it is concluded that the distribution coefficient of Eu(III) and Th(IV) decreases if the ISA concentration is greater than  $1 \times 10^{-5} \text{ mol/dm}^3$  and  $2 \times 10^{-5} \text{ mol/dm}^3$ , respectively (cf. Figure 6.2.2.3-6).

Bradbury and Sarott (1994) considered the effect of ISA concentration on the solubility of major elements as shown in Table 6.2.2.3-2. The lower limit of ISA concentration is estimated to be  $1 \times 10^{-6} \text{ mol/dm}^3$ .

#### **(5) Summary**

Based on international studies, the concentration above which ISA affects the solubility and sorption behaviour of nuclides is considered to be  $1 \times 10^{-6} \text{ mol/dm}^3$ . Given this analytical result, and since the expansion of a plume with an ISA concentration of more than  $1 \times 10^{-6} \text{ mol/dm}^3$  would be restricted to the vicinity of the disposal facility, the effect of ISA can be ignored if a separation distance of 100 m between the TRU waste and HLW disposal facilities can be ensured.

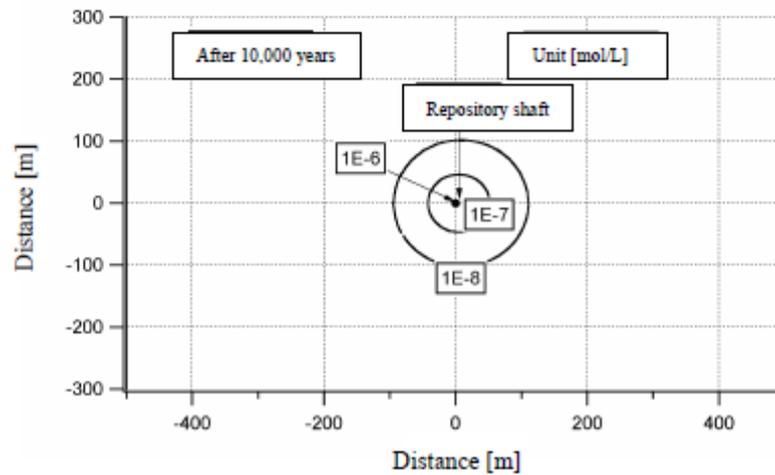
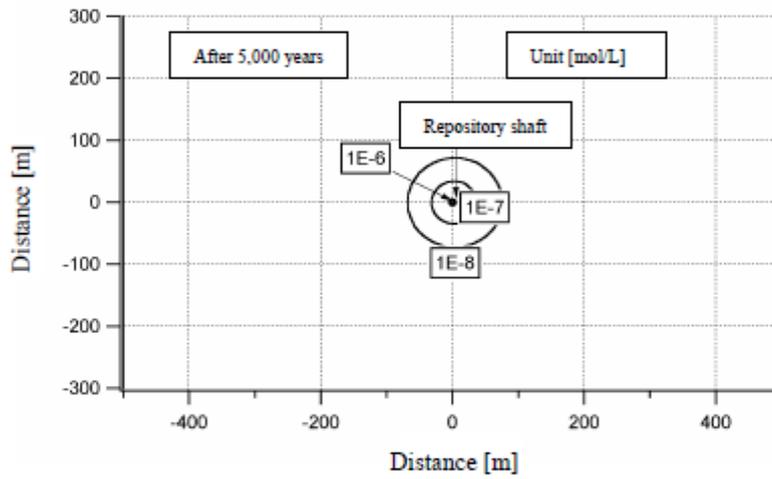
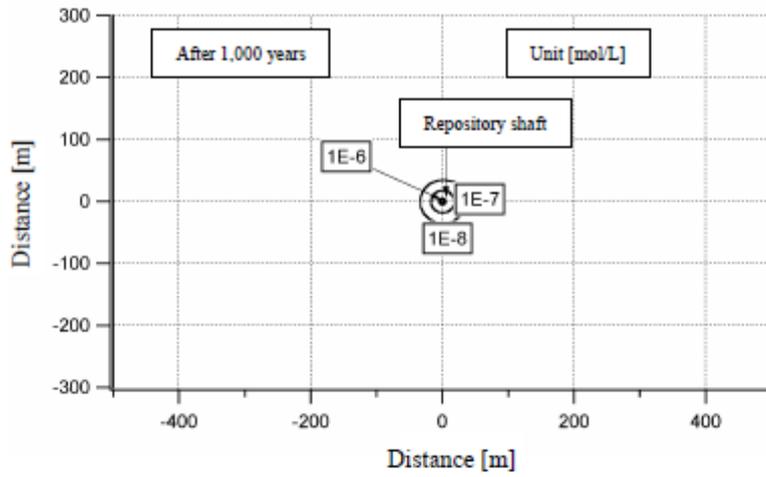


Figure 6.2.2.3-3 Results of the simulation of an organic plume in crystalline rock (hydraulic conductivity  $1 \times 10^{-9} \text{ m s}^{-1}$ )

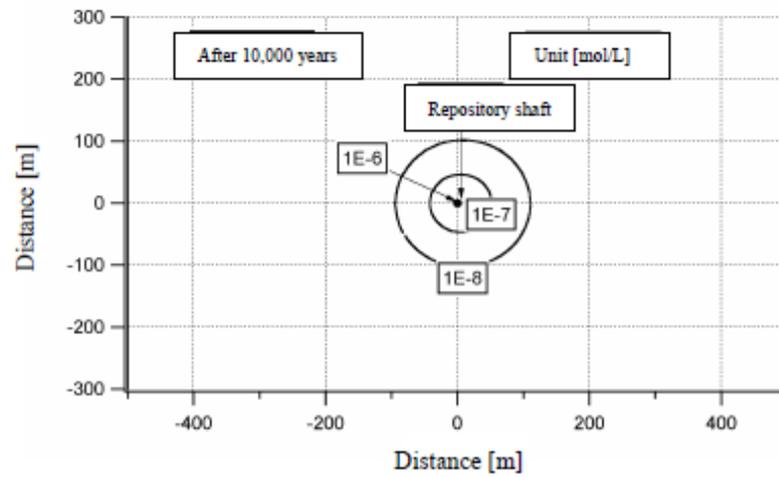
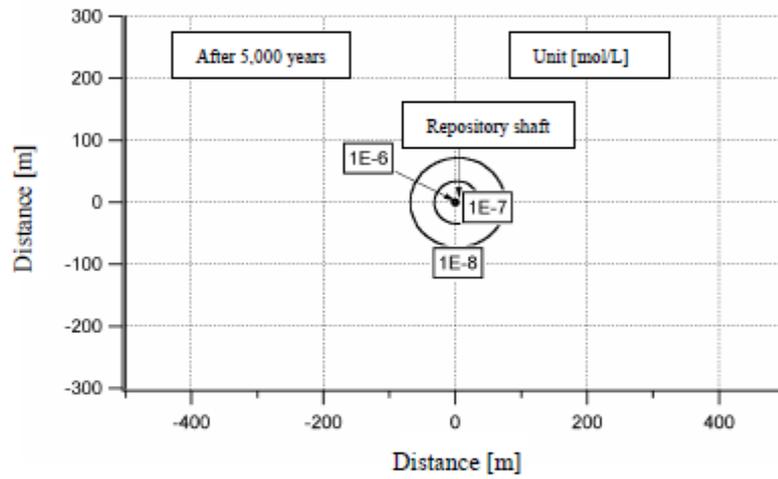
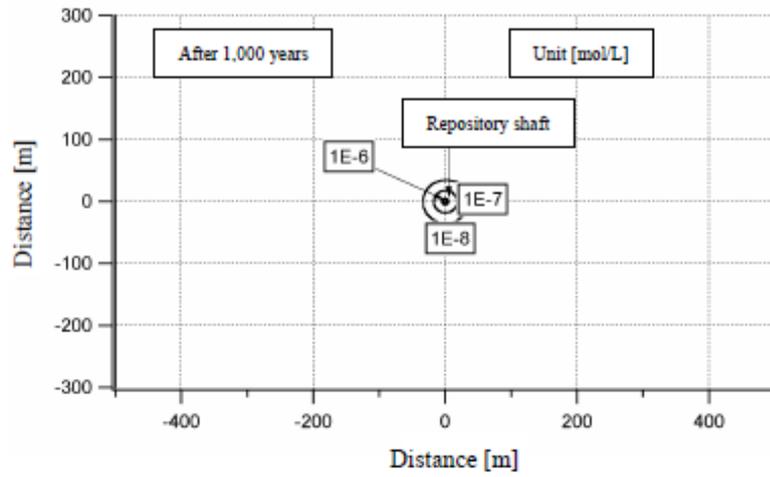


Figure 6.2.2.3-4 Results of the simulation of an organic plume in sedimentary rock (hydraulic conductivity  $1 \times 10^{-9} \text{ m s}^{-1}$ )

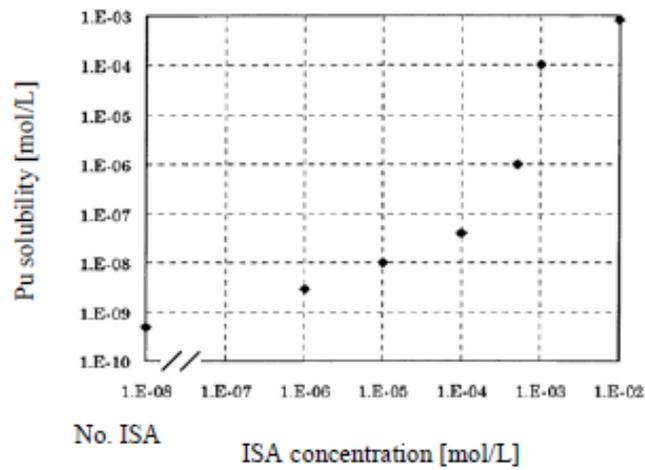


Figure 6.2.2.3-5 Relationship between Pu solubility and ISA concentration (Greenfield and Holtom, 1995)

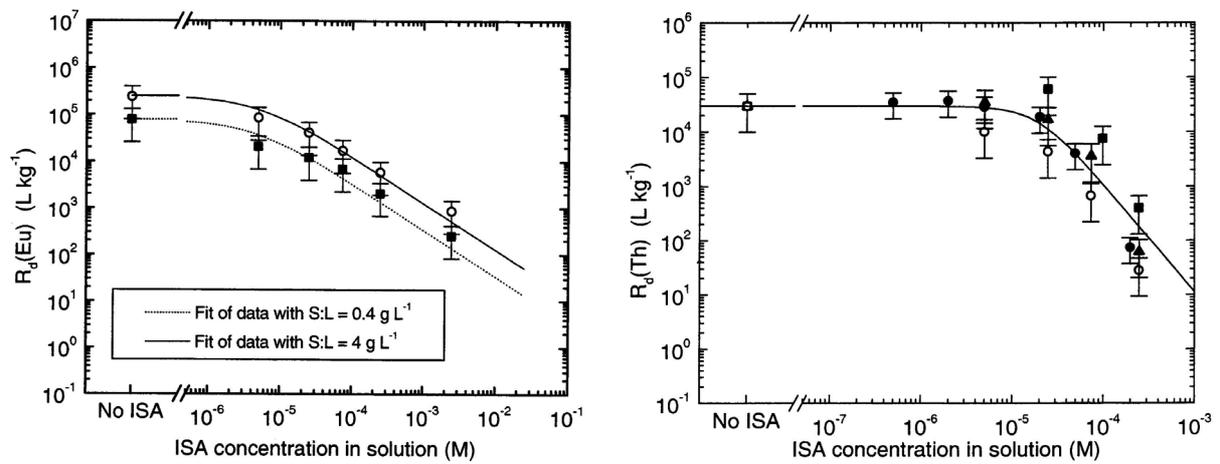


Figure 6.2.2.3-6 Relationships between distribution coefficients of Eu and Th and ISA concentrations (Tits et al., 2002)

Table 6.2.2.3-2 Relationships between increases in the concentrations of major elements and ISA concentrations

ISA concentration (mol/dm <sup>3</sup> )	Increase (solubility enhancement factor)					
	Pu(IV)	Am(III)	Th(IV)	U(VI)	Np(IV)	Tc(IV)
1E-06	1	1	1	1	1	1
1E-05	20	20	20	1	1	1
1E-04	500	500	500	~2	~2	1
1E-03	2E+04	2E+04	2E+04	~100	~100	~10
1E-02	1E+06	1E+06	1E+06	~5E+03	~5E+03	~500

#### 6.2.2.4 Effects of nitrate

In TRU waste, nitrate that includes organic material mainly takes the form of liquid waste from reprocessing. The total amount of nitrate in the waste destined for geological disposal is estimated to be equivalent to about  $3.25 \times 10^6$  kg  $\text{NaNO}_3$ , as described in Section 4.4.9. The concentration of nitrate in the disposal facility is estimated to be  $7.1 \text{ mol/dm}^3$ \*<sup>1</sup>, in the case where the pore water concentration in the facility is calculated from the amount of waste and nitrate. The concentration is more or less equal to the degree of saturated solution of nitrate ( $7.7 \text{ mol/dm}^3$  at  $25^\circ\text{C}$ ) since nitrate is readily dissolved by groundwater.

The following effects of nitrate on the barrier performance of the HLW disposal system are considered:

- effects on the mineralogical alteration of bentonite material and host rock
- effects on the permeability of bentonite material
- effects on the ability of the engineered and natural barriers to sorb radionuclides
- effects on overpack corrosion
- variations in redox conditions in the geological environment and elemental oxidization by nitrate ion.

The possibility that variations in redox conditions, including elemental oxidization, may occur is pointed out in Section 4.4.9. Improving knowledge of these issues in the future is considered to be important for making quantitative decisions.

This section evaluates the nitrate plume which is generated by dissolution of nitrate and associated cations in pore water. The effect on barrier performance of nitrate is described in Section 4.4.9. In the present section, the nitrate plume is described and the effect of the nitrate salts is evaluated.

##### (1) Evaluation of the nitrate plume

The effect of nitrate from the TRU waste disposal facility depends on its concentration in the groundwater. The nitrate plume from the TRU waste disposal facility is evaluated using hydrological mass transport analysis.

##### a. Analytical conditions

The nitrate plume is affected by the following conditions.

---

\*<sup>1</sup>: Pore water concentration of nitrate in the facility is calculated using a value for the total amount of nitrate of  $3.25 \times 10^6$  kg/ (volume of waste ( $5,701 \text{ m}^3$ ) $\times$ porosity of cement material (0.19)) $\times$ filling ratio (0.2)/molecular weight of  $\text{NaNO}_3$  ( $8.5 \times 10^{-2}$  kg/mol) .

- ① Nitrate concentration in the disposal facility
- ② Performance of the waste matrix
- ③ Sorption distribution coefficient of nitrate in cementitious material
- ④ Sorption distribution coefficient of nitrate in host rock
- ⑤ Groundwater flow direction and flow velocity in the host rock
- ⑥ Biodegradation in the host rock

A period of 1000 years was specified for the leaching of nitrates from the matrices of solidified bitumen. The activity of microbes, which decompose nitrates, in geological media is considered in Section 4.4.7 but, since microbial activity is strongly dependent on site conditions, the effects are not considered in this evaluation. As described in Section 4.4.9,  $\text{NO}_3^-$  generated by dissolution of nitrates is partly reduced by reactions with metals and the chemical form of the nitrate may thus be changed. However, this evaluation assumes that nitrate remains as  $\text{NO}_3^-$ , which is only poorly sorbed in barrier materials.

The sorption distribution coefficient of  $\text{NO}_3^-$  in barrier material is considered to be the same as that of  $\Gamma$ . The sorption distribution coefficient of  $\Gamma$  obtained using the JNC-SDB database (Shibutani et al., 1999; Suyama and Sasamoto, 2004) is summarised in Figure 6.2.2.4-1. The compositions of the liquid phase were specified to be NaCl solution,  $\text{NaHCO}_3$  solution or groundwater. The solid phase was specified as having the composition of granitic rock, sandstone or amphibole. The minerals considered belonged to the feldspar group, augite group and biotite group. The  $\text{Na}^+$  concentration in Figure 6.2.2.4-1 is lower than the nitrate saturation concentration. However, the sorption distribution coefficient of  $\Gamma$  in liquids with different  $\text{Na}^+$  concentrations has a value greater than  $0.0001 \text{ m}^3/\text{kg}$ . Hence, in the evaluation of the nitrate plume,  $0.0001 \text{ m}^3/\text{kg}$  was specified as the sorption distribution coefficient in the barrier material.

The analytical model was a 2D vertical cross-section model, which considered the rock surrounding the TRU disposal facility between 300 m upstream and 500 m downstream (Figure 6.2.2.4-2). The 2D hydrological mass transport analysis code Dtransu-2D·EL (e.g. Hishiya et al., 1999) was used. The analysis used the parameters values, including physical properties of the engineered barriers and the host rock, which are shown in Table 6.2.2.4-1.

#### **b. Analytical results**

Figures 6.2.2.4-3 and 6.2.2.4-4 show the temporal changes in nitrate concentrations given by analyses for crystalline and sedimentary type basement rock respectively. In each case, the hydraulic conductivity of the rock was  $1 \times 10^{-9} \text{ m s}^{-1}$ .

Tables 6.2.2.4-3 and 6.2.2.4-4 show the maximum concentrations of nitrates attained during a period of 100,000 years at various distances (50 m, 100 m, 200 m, 300 m) from the TRU waste disposal facility in the following directions:

- ① upstream along the groundwater flow path
- ② perpendicular to groundwater flow
- ③ downstream along the groundwater flow path

In cases ① and ②, the largest concentrations of nitrate in the plume are in the order  $0.01 \text{ mol/dm}^3$  at a distance of 100 m from the facility. However, at a distance of 300 m from the facility the concentrations are in the order  $1 \times 10^{-4} \text{ mol/dm}^3$ . Additionally, in case ③, particularly when the rock is low-porosity crystalline bedrock, the plume shows high concentrations of nitrate at locations that are far from the TRU waste disposal facility. At a distance of 500 m from the facility, the maximum concentrations of nitrate are in the order  $0.01 \text{ mol/dm}^3$ . However, the time taken to reach these concentrations is estimated to be  $1 \times 10^4$  years.

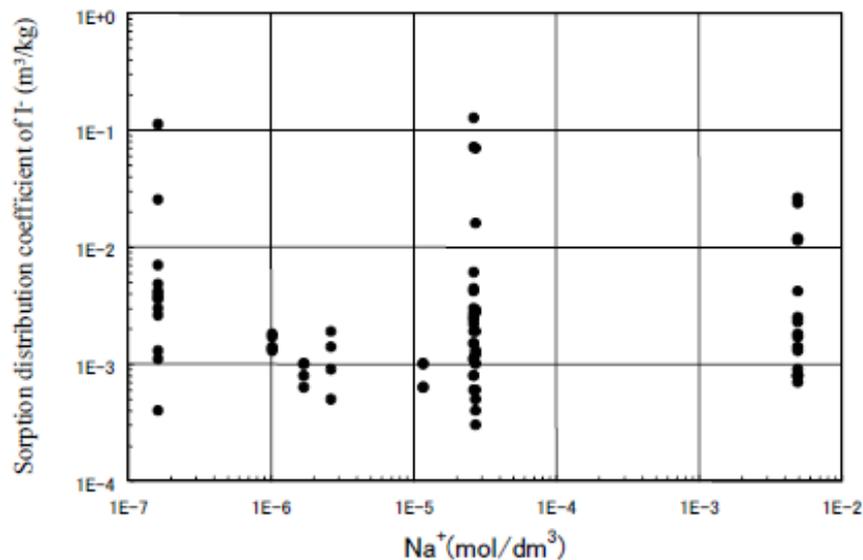


Figure 6.2.2.4-1 Relationship between sorption distribution coefficient of  $\text{I}^-$  on rock and  $\text{Na}^+$  concentration in liquid

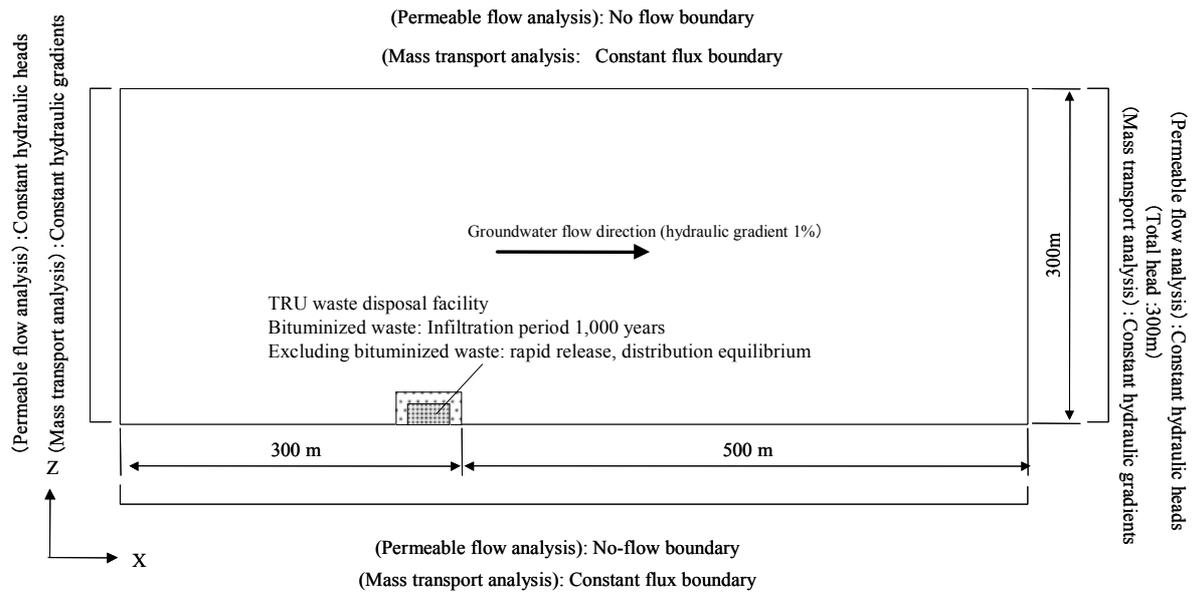
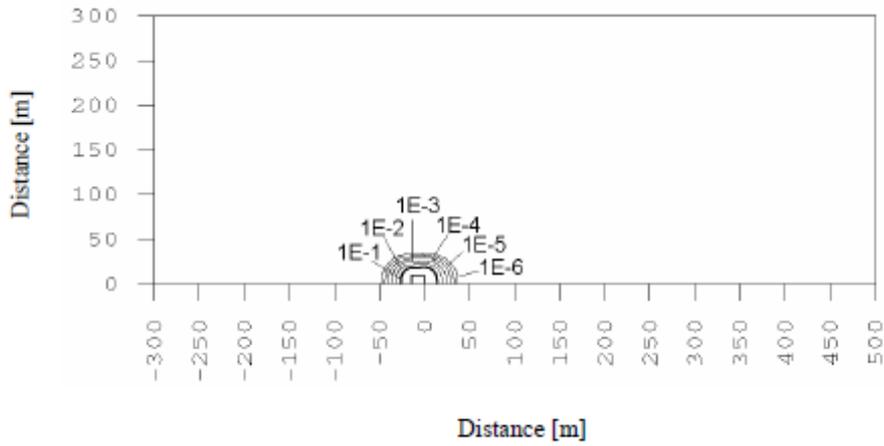


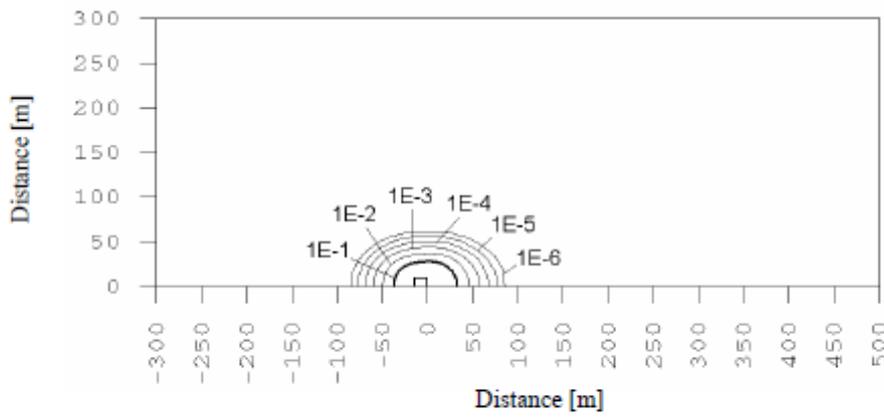
Figure 6.2.2.4-2 Analytical model

Table 6.2.2.4-1 Parameters and values used

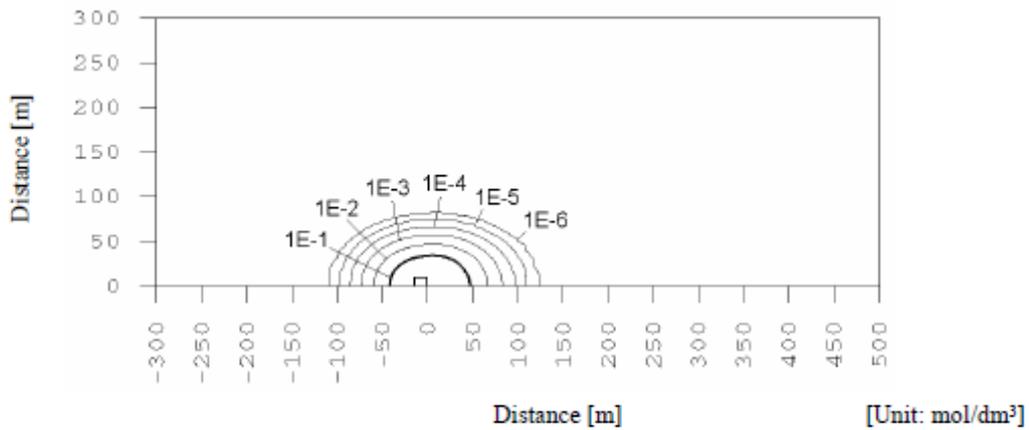
Parameter		Unit	Crystalline rock	Sedimentary rock
Quantity of nitrate		kg	3.25×10 <sup>6</sup>	
Hydraulic conductivity	Hydraulic conductivity of cement	m s <sup>-1</sup>	4×10 <sup>-6</sup>	
Diffusion coefficient	Effective diffusion coefficient in cement	m <sup>2</sup> /s	8×10 <sup>-10</sup>	
	Effective diffusion coefficient in host rock	m <sup>2</sup> /s	8×10 <sup>-11</sup>	1.2×10 <sup>-9</sup>
Sorption distribution coefficient	Sorption distribution coefficient in cement	m <sup>3</sup> /kg	0.0001	
	Sorption distribution coefficient in host rock	m <sup>3</sup> /kg	0.0001	
Disposal facility		—	Waste Group 3	
Natural barrier	Hydraulic conductivity	m s <sup>-1</sup>	1×10 <sup>-8</sup> , 1×10 <sup>-9</sup> , 1×10 <sup>-10</sup>	
	Porosity	—	0.02	0.30
	True density	Mg/m <sup>3</sup>	2.7	2.7
	Hydraulic gradient	m/m	0.01	
	Dispersion length	—	1/10 of migration distance	



(a) After 1,000 years



(b) After 5,000 years



(c) After 10,000 years

Figure 6.2.2.4-3 Temporal variations of nitrate distribution  
(crystalline bedrock: hydraulic conductivity of host rock  $1 \times 10^{-9} \text{ m s}^{-1}$ )

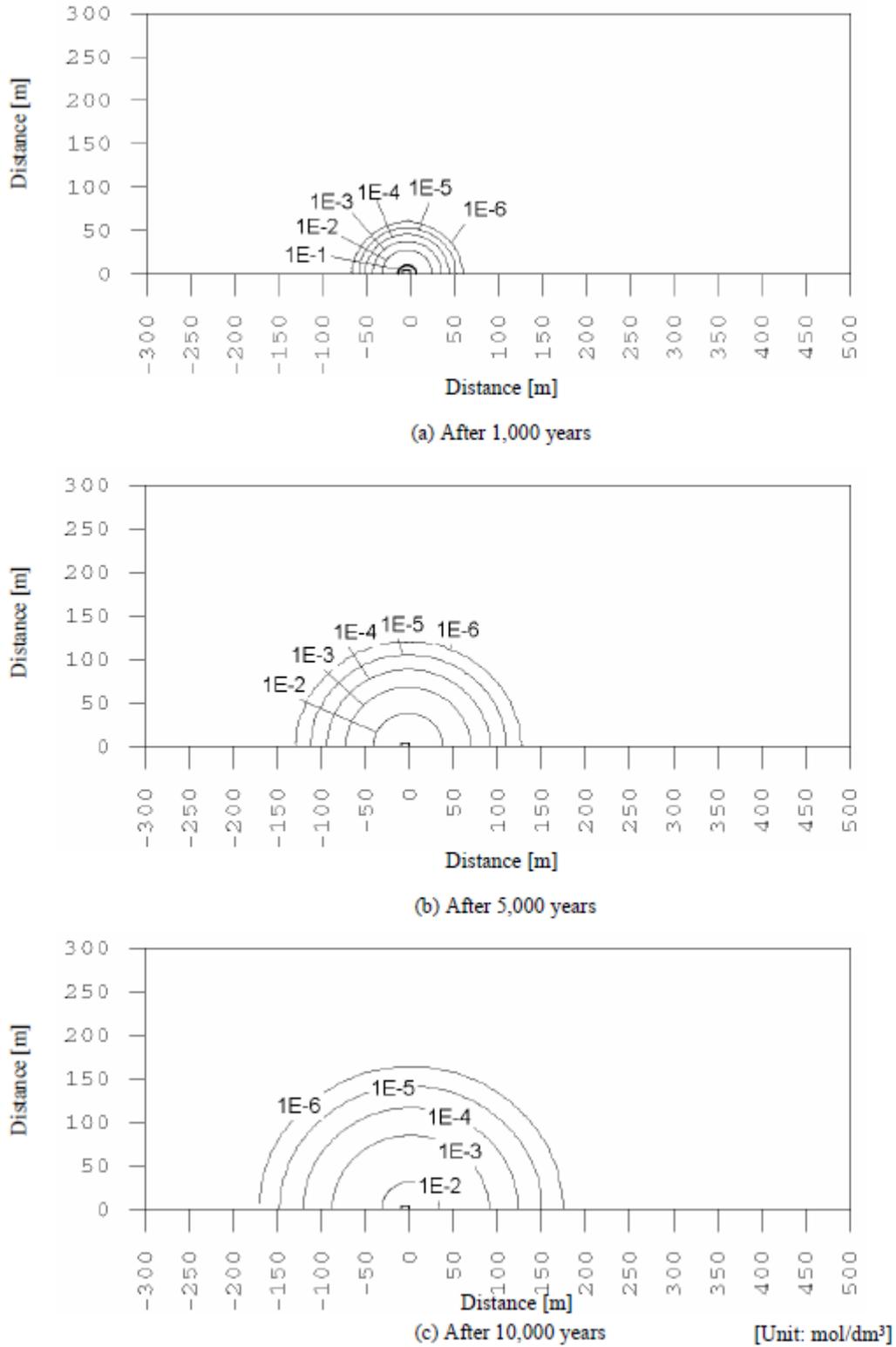


Figure 6.2.2.4-4 Temporal variations of nitrate distribution  
(sedimentary rock: hydraulic conductivity of host rock  $1 \times 10^{-9} \text{ m s}^{-1}$ )

Table 6.2.2.4-2 Maximum nitrate concentrations attained at different points in the plume upstream from the TRU waste disposal facility during a period of 100,000 years

Rock type: crystalline bedrock (Unit : mol/ dm <sup>3</sup> )			
Isolation distance (m)	Reference case	Sensitivity analysis case	
	Hydraulic conductivity 1E-9 m/s	Hydraulic conductivity 1E-8 m/s	Hydraulic conductivity 1E-10 m/s
50	5.1E-2	1.4E-2	1.4E-1
100	1.6E-3	9.7E-5	1.5E-2
200	3.0E-6	3.3E-8	7.7E-6
300	0.0E+0	0.0E+0	0.0E+0
Rock type: sedimentary rock (Unit : mol/ dm <sup>3</sup> )			
Isolation distance (m)	Reference case	Sensitivity analysis case	
	Hydraulic conductivity 1E-9 m/s	Hydraulic conductivity 1E-8 m/s	Hydraulic conductivity 1E-10 m/s
50	5.9E-3	2.1E-3	6.8E-3
100	1.2E-3	1.3E-4	1.6E-3
200	1.7E-4	1.3E-6	3.4E-4
300	0.0E+0	0.0E+0	0.0E+0

Table 6.2.2.4-3 Maximum nitrate concentrations attained at different points in the plume perpendicular to the flow direction during a period of 100,000 years

Rock type : crystalline bedrock (Unit : mol/dm <sup>3</sup> )			
Isolation distance (m)	Reference case	Sensitivity analysis case	
	Hydraulic conductivity 1E-9 m/s	Hydraulic conductivity 1E-8 m/s	Hydraulic conductivity 1E-10 m/s
50	6.3E-2	3.7E-3	1.3E-1
100	3.2E-3	4.0E-6	1.1E-2
200	2.7E-6	1.2E-11	4.3E-6
300	5.2E-11	1.2E-16	3.6E-11
Rock type : sedimentary rock (Unit : mol/dm <sup>3</sup> )			
Isolation distance (m)	Reference case	Sensitivity analysis case	
	Hydraulic conductivity 1E-9 m/s	Hydraulic conductivity 1E-8 m/s	Hydraulic conductivity 1E-10 m/s
50	6.0E-3	3.3E-3	6.1E-3
100	1.3E-3	2.7E-4	1.4E-3
200	3.1E-4	9.3E-6	3.5E-4
300	9.3E-5	8.6E-7	1.0E-4

Table 6.2.2.4-4 Maximum nitrate concentrations of the plume attained at different points on the downstream side from TRU waste disposal facility over a period of 100,000 years

Rock style : crystalline bedrock (Unit : mol/dm <sup>3</sup> )			
Isolation distance (m)	Reference case	Sensitivity analysis case	
	Hydraulic conductivity 1E-9 m/s	Hydraulic conductivity 1E-8 m/s	Hydraulic conductivity 1E-10 m/s
50	3.7E-1	7.7E-1	1.3E-1
100	1.9E-1	4.6E-1	2.0E-2
200	4.7E-2	2.5E-1	1.7E-5
300	1.5E-3	1.7E-1	2.5E-10
400	7.8E-6	1.2E-1	3.4E-16
500	1.5E-8	1.8E-2	0.0E+0
Rock style : sedimentary rock (Unit : mol/dm <sup>3</sup> )			
Isolation distance (m)	Reference case	Sensitivity analysis case	
	Hydraulic conductivity 1E-9 m/s	Hydraulic conductivity 1E-8 m/s	Hydraulic conductivity 1E-10 m/s
50	6.2E-3	1.7E-2	4.9E-3
100	2.1E-3	7.4E-3	1.4E-3
200	6.9E-4	3.2E-3	3.5E-4
300	1.8E-4	2.1E-3	5.5E-5
400	2.5E-5	1.5E-3	4.2E-6
500	3.2E-6	1.3E-3	3.2E-7

**(2) Summary of present knowledge concerning the effects of nitrates**

The effects of nitrates in Group 3 waste packages, notably on the alteration of cementitious material and nuclide migration parameters such as radionuclide solubilities are described in Section 4.4.9. Here, knowledge concerning the buffer material and host rock and its relevance for a HLW disposal facility is summarized.

**a. Mineralogical effects on bentonite material and host rock**

Tests were performed over a period of 18 months to investigate the mineralogical effects of NaNO<sub>3</sub> on bentonite material and host rock (Nakazawa et al., 2004). Cases in which the NaNO<sub>3</sub> concentration was at saturation or at 1/1,000 of saturation were considered. Figure 6.2.2.4-5 shows the result of an X-ray diffraction analysis of a bentonite sample before and after alteration. No significant alteration is indicated when comparing cases in which NaNO<sub>3</sub> is present and absent. However, the bentonite may undergo cation exchange reactions. If NaNO<sub>3</sub> is at saturation, the Na<sup>+</sup> cations will be exchanged for alkali cations originating in the cement. However, when the concentrations of nitrate were 0.01 mol/dm<sup>3</sup> and 0 mol/dm<sup>3</sup>, the Ca<sup>2+</sup> type bentonite remained dominant.

Additionally, in each of the 2 types of sedimentary rock considered there was no significant alteration of the initial minerals such as quartz and plagioclase, even when  $\text{NaNO}_3$  was present.

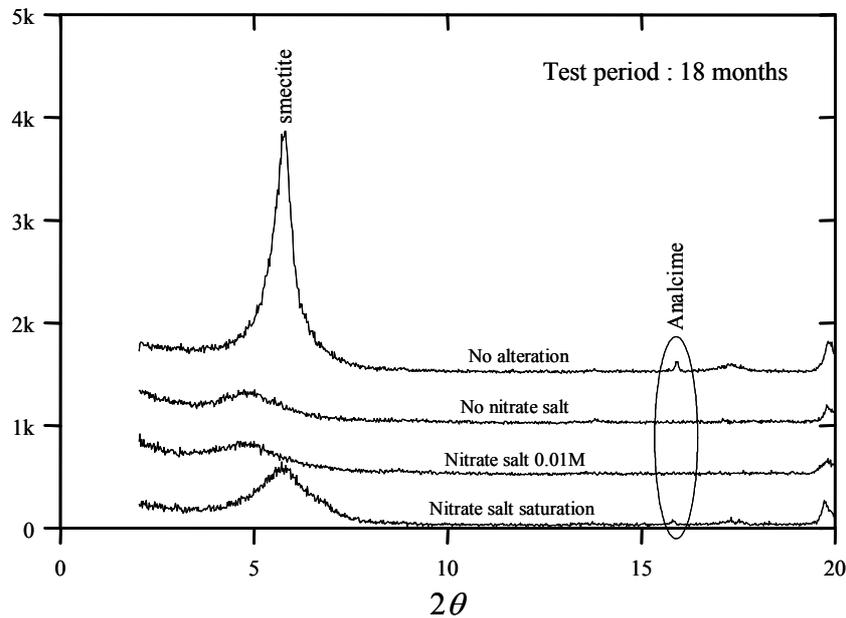


Figure 6.2.2.4-5 Results of X-ray diffraction analyses of bentonite before and after alteration tests (Nakazawa et al., 2004)

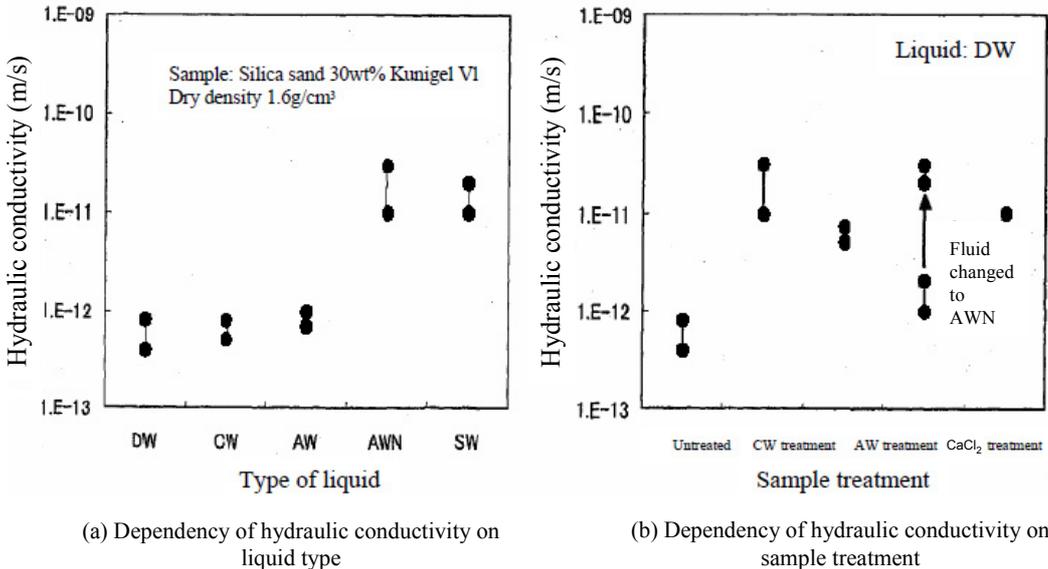
**b. Effect on the permeability of bentonite material**

It is reported that, if the concentrations of dissolved ions are high, as in synthetic sea water, then the hydraulic conductivity of bentonite will increase (JNC, 2002). This change is due to a decrease in the swelling capability of the bentonite, caused by the aggregation of bentonite particles. A permeability test using a  $3 \text{ mol/dm}^3$  solution of  $\text{NaNO}_3$  (pH 13) showed that the hydraulic conductivity of bentonite increased in the same way as when synthetic sea water (SW) was used (Figure 6.2.2.4-6; Mihara et al., 2004). However, this phenomenon is not specific to the presence of nitrate. It is considered that the influence on the performance of a HLW disposal facility would be similar to that shown by the performance assessment in the H12 report when water with sea water composition is assumed (JNC, 2000b).

**c. Effects on sorption properties of radionuclides in the engineered and natural barriers**

The effects of nitrate on the sorption of radionuclides onto cementitious material and host rock are described in Section 4.4.9. Saso et al. (2004) evaluated the sorption distribution coefficient of organic C-14, Cl-36 and Np-237 for cement (OPC and OPC/BFS=1/9) and the sorption distribution coefficient of organic C-14 in sedimentary rock. No significant decrease in the sorption of organic C-14 and Cl-36 was observed when the  $\text{NaNO}_3$  concentration was  $0.007 \text{ mol/dm}^3$  (1/1000 times  $\text{NaNO}_3$  saturation). However, when the concentration of  $\text{NaNO}_3$  is at saturation, the sorption distribution coefficient was decreased by 1 order of magnitude compared to the value when there is no  $\text{NaNO}_3$ . In contrast, no significant decrease in the

sorption of Np-237 on cementitious material was observed in a saturated NaNO<sub>3</sub> solution. Similarly, no significant decrease in the sorption of organic C-14 on sedimentary rock or cementitious material was observed when the NaNO<sub>3</sub> concentration was 0.007 mol/dm<sup>3</sup>. However, in a saturated NaNO<sub>3</sub> solution the sorption distribution coefficient decreased by 1 order of magnitude compared to the case when no NaNO<sub>3</sub> was present. There is no information concerning the sorption distribution coefficient of radionuclides in barrier materials in the NaNO<sub>3</sub> concentration range from 0.01 mol/dm<sup>3</sup> to saturation. Figure 6.2.2.4-7 summarizes the relationship between Na<sup>+</sup> concentration and the distribution coefficient for Cs<sup>+</sup>, obtained using the JNC-SDB database (Shibutani et al., 1999; Suyama and Sasamoto, 2004). Solutions with different concentrations of NaCl, Na<sub>2</sub>CO<sub>3</sub>, Na<sub>2</sub>SO<sub>4</sub> and NaHCO<sub>3</sub> were specified. The selected solids were basalt, granite and granodiorite, shale, hematite and microcline. These materials are included in the basement rocks that form the natural barrier. According to Figure 6.2.2.4-7, the distribution coefficient of Cs<sup>+</sup> is slightly decreased when the Na<sup>+</sup> concentration is higher than 0.1 mol/dm<sup>3</sup>.



Note: DW : De-aerated, deionized water, CW : highly alkaline solution at pH 12.5, AW : highly alkaline solution at pH 13.2, AWN: solution produced by adding 3 mol/dm<sup>3</sup> NaNO<sub>3</sub> to AW, SW: synthetic sea water.

Figure 6.2.2.4-6 Permeability test using sodium nitrate solution (3 mol/dm<sup>3</sup>)  
(partly revised from Mihara et al., 2004)

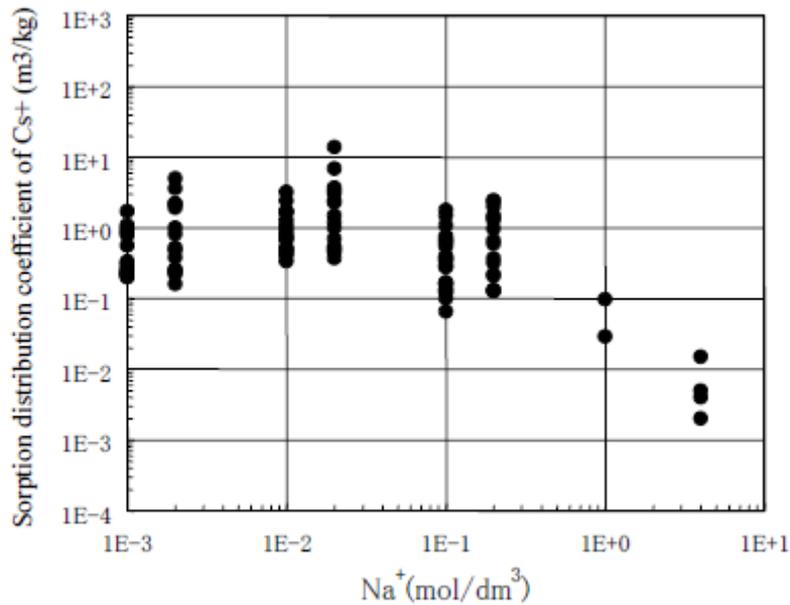


Figure 6.2.2.4-7 Relationship between the sorption distribution coefficient of  $\text{Cs}^+$  on rock and minerals and the  $\text{Na}^+$  concentration of liquid

**d. Effects on corrosion of metal such as carbon steel**

Consideration is given to the possibility that  $\text{NO}_3^-$  might promote the partial corrosion and stress-corrosion cracking of the carbon steel overpack used in a HLW repository.

In the H12 report, the total concentration of nitrogen-bearing ions and radicals generated by radiolysis is assumed to be  $2 \times 10^{-4} \text{ mol/dm}^3$ . It is considered that the possibility of stress-corrosion cracking due to nitrate in the disposal facility is low. Stress-corrosion cracking did not occur in a low strain rate experiment using  $0.001 \text{ mol/dm}^3$  of boiling  $\text{KNO}_3$  solution (Beavers et al., 1987). Additionally, stress-corrosion cracking is suppressed in the presence of carbonate or chloride (Koiwai, 1983).

In the latter case, the concept of radiolysis-induced corrosion in the H12 report (JNC, 2000a) serves as a useful reference. According to this concept, if the cathodic current density resulting from the oxidizing chemical species produced by radiolysis does not exceed the passive current density, radiolysis will not drive partial corrosion. The same concept can be applied to the  $\text{NO}_3^-$  ion. It is assumed that the  $\text{NO}_3^-$  is reduced to ammonia at the overpack surface, thereby producing a cathodic electric current. It is considered that the  $\text{NO}_3^-$  will not cause localized corrosion if the  $\text{NO}_3^-$  concentration does not exceed  $4.5 \times 10^{-4} \text{ mol/dm}^3$ . This latter concentration is obtained by assuming that  $\text{NO}_3^-$  diffuses into the buffer material.

**(3) Summary**

The HLW repository should be constructed in such a way as to prevent it from being influenced directly by

the TRU waste facility. The distance of the HLW repository from the TRU waste facility and the groundwater flow directions should be taken into account. The effects of nitrates are considered to depend on where, within the flow system, the HLW is located relative to the TRU waste. The effect will depend on the widening of the nitrate-bearing plume. The maximum concentration of nitrate in the plume, at a distance of 300 m upstream from the TRU waste facility was estimated to be  $1 \times 10^{-4}$  mol/dm<sup>3</sup>. Taking into account current knowledge about the effects of nitrate, it is considered that, at such a concentration, there will be no significant influence on the mineralogical alteration of barrier material in a HLW disposal facility, i.e. the permeability of bentonite material, sorption distribution coefficients of barrier materials and corrosion of the carbon steel overpack will not be significantly affected.

The position within the flow system of a TRU waste disposal facility relative to a HLW disposal facility might change if groundwater flow directions change due to long-term geological processes such as topographic or structural variations. Therefore, the most extreme case in which the HLW facility lies downstream of the TRU waste facility was considered. In this case, a higher nitrate concentration is expected than in the cases where a nitrate-bearing plume expands upstream and laterally. Among the cases evaluated, the nitrate-bearing plume reached the greatest distance downstream in the case that considered crystalline rock with a hydraulic conductivity of  $1 \times 10^{-8}$  m s<sup>-1</sup>. However, assuming that the HLW facility is constructed 500 m from the TRU waste facility, the nitrate concentration took about 20,000 years to reach more than  $1 \times 10^{-4}$  mol/dm<sup>3</sup>. Furthermore, the extreme geological changes required to produce this result would actually require a very long time period. Taken together, these considerations imply that any effects from nitrate would be small during the period for which the integrity of the overpack can be guaranteed.

It considered possible to avoid any significant effects of nitrate by constructing the TRU waste disposal facility several 100 m downstream of the HLW disposal facility.

#### **6.2.2.5 Effects of high pH**

As described in sections 4.4.2 and 4.4.3, large amounts of cement would be used in a TRU waste disposal facility. Due to the dissolution of cementitious materials, pore water in the facility would reach high pH values. Initially, the pore water in cementitious material reaches  $\text{pH} > 12.5$  due to the dissolution of Na- and K- hydroxides. When the Na and K are lost by dispersion into the natural barrier, the pore water pH is controlled by the solubility of portlandite ( $\text{Ca}(\text{OH})_2$ ) and decreases to about 12.5. Ca becomes dominant. If  $\text{Ca}(\text{OH})_2$  disappears, pH is then controlled by calcium-silica-hydrate gel (C-S-H gel). If the Ca/Si ratio in the C-S-H gel gradually decreases due to the selective leaching of Ca, the pore water pH also gradually decreases. The alkaline component of the cement material is dissolved by the groundwater and gradually migrates into the natural barrier.

Hence, it is considered that a region will develop in the natural barrier where the pH of the groundwater will become high. The following effects of such a high-pH plume on barrier performance are considered:

- effects on the mass transport characteristics of bentonite material
- Ca-type montmorillonite formation
- other mineralogical effects in bentonite material
- influence on the corrosion of carbon steel used for the overpack
- mineralogical influences in the natural barrier
- influences on nuclide migration in the natural barrier

In this section, the high-pH plume that would migrate from a TRU waste disposal facility is evaluated and the effects of the plume are described.

**(1) Evaluation of the high-pH plume**

In a TRU waste disposal facility, a large amount of cementitious material will be used to solidify waste, as a filling material, as structural components such as drift supports and to plug and grout excavations. On the long term, the cementitious material used for these purposes will be altered by reaction with groundwater. As a result, the pore water that contacts the cement will become highly alkaline and will react with the surrounding bentonite material and the natural barrier. In order to evaluate the consequences of alkaline pore water migration and reaction with minerals, a 2D geochemical-hydrogeological coupled mass transport analysis code, “PHREEQC-TRANS”, was used.

**a. Target facility for analysis**

A disposal facility for waste Group 4 (without buffer material) was evaluated. The facility, which is shown in Figure 4.2.4-1 to have a circular cross-section of 12 m diameter, is represented by a square in the illustration of the 1D model shown in Figure 6.2.2.5-1. A high-pH plume was allowed to migrate from the facility into the upstream and downstream regions of the host rock. Each of these regions was 200 m long. During the analysis, fixed heads and solute concentrations were specified to occur at the extreme upstream and downstream boundaries, as shown in Figure 6.2.2.5-1.

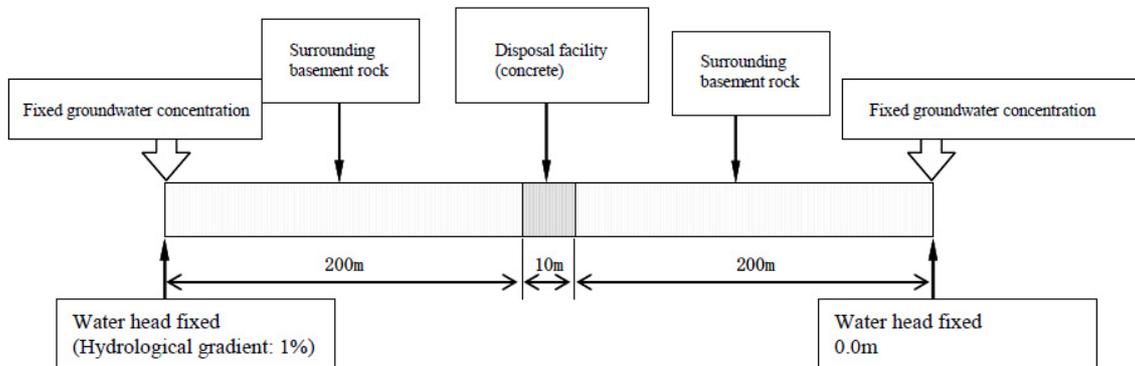


Figure 6.2.2.5-1 Illustration of the modeled regions

**b. Specification of host rock mineral composition**

In the region representing the host rock around the disposal facility, reactions between the alkaline components from the facility and the minerals composing the host rock were simulated. Dissolution of rock-forming minerals, generation of secondary minerals and changes in pore water composition occurred. However, the minerals forming the host rock will depend on the site conditions. Since the disposal site has not yet been decided, the actual conditions cannot be specified at present. Hence, a hypothetical basement rock was assumed in this evaluation. This basement rock was assumed to contain silica minerals that may readily react with the alkaline components originating in the TRU waste repository. These minerals include several forms of silica, such as quartz, cristobalite, chalcedony and amorphous silica (Miyashiro and Kushiro, 1972). From these minerals, chalcedony was selected for consideration. This mineral may occur in a fresh water groundwater setting and is an accessory mineral in the bentonite.

Additionally, fractures along the flow path are also considered to be filled by amorphous silica (Miyashiro and Kushiro, 1972) and clay minerals (JNC, 2000b). The clay minerals were evaluated using thermodynamic data for montmorillonite. The quantities of these minerals that react are assumed to be trivial and the amounts of reaction were therefore all specified to be 0.1wt% (cf. Table 6.2.2.5-1). Insignificant amounts of other minerals in the solid phase were considered to react.

Table 6.2.2.5-1 Minerals which contribute to the geochemical reaction

	Composition	Amount of reaction (wt%)
Chalcedony	SiO <sub>2</sub>	10
Amorphous silica	SiO <sub>2</sub>	0.1
Calcite	CaCO <sub>3</sub>	0.1
Montmorillonite	Na <sub>0.33</sub> Mg <sub>0.33</sub> Al <sub>1.67</sub> Si <sub>4</sub> H <sub>2</sub> O <sub>12</sub>	0.1

**c. Specification of cement hydrate minerals in the disposal facility**

In this evaluation, all cementitious materials are treated as mortar. The modeled mortar is based on the usual specifications for cementitious materials used the various types of evaluation in this report (Appendix 3B). The specified mixture, mineral composition and chemical composition of the model mortar are shown in Tables 6.2.2.5-2 to 6.2.2.5-4. Although Na and K are free alkali components of the cement which are regulated for fixed periods by sorption onto C-S-H gels, it is assumed that their total amount migrates instantaneously.

Table 6.2.2.5-2 Chemical composition of cement (wt%)

	SiO <sub>2</sub>	Al <sub>2</sub> O <sub>3</sub>	Fe <sub>2</sub> O <sub>3</sub>	CaO	MgO	SO <sub>3</sub>	Na <sub>2</sub> O	K <sub>2</sub> O
Ordinary Portland Cement	21.3	5.31	2.57	64.8	1.95	1.94	0.24	0.56

Table 6.2.2.5-3 Specified mix of model mortar

Water-cement ratio W/C (%)	Unit quantity (kg/m <sup>3</sup> )		
	Water (W)	OPC (C)	Amount of fine aggregate
55	266	483	1,449

Table 6.2.2.5-4 Mineral composition of cement (mol/dm<sup>3</sup> water)

	Ettringite	C <sub>3</sub> FH <sub>6</sub>	C <sub>3</sub> AH <sub>6</sub>	C-S-H gel	Portlandite	Na <sub>2</sub> O	K <sub>2</sub> O
Mortar	0.211	0.416	1.132	16.4	7.45	0.097	0.15

#### d. Groundwater composition

The high-pH plume from the TRU waste disposal facility is affected by chemical reactions as well as by the surrounding groundwater flow conditions. In addition to groundwater of fresh water origin, groundwater of seawater origin might also occur in the basement rock. Hence, both these types of groundwater are evaluated here.

#### e. Model for reaction of silica minerals

This evaluation assumed that the reactive silica minerals in the basement rock are chalcedony and amorphous silica. When calculating the chemical environment and long-term performance of the barrier material, it was assumed that chalcedony is an accessory mineral in the buffer material. Mass transport in the buffer material will occur predominantly by diffusion, due to the low permeability of the material. Consequently, the time needed to transport dissolved silica away from the site of dissolution will be sufficiently long, relative to the time required for the solubility limit of the silica to be reached, that the chemical reaction can be treated as being at instantaneous equilibrium. In contrast, the hydraulic conductivity of the basement rock evaluated here was sufficiently high that cases where there was insufficient time for chemical equilibrium to be reached had to be considered. It was assumed that the amount of chalcedony reacted would depend on its dissolution rate. However, the rate of amorphous silica dissolution was considered to be sufficiently high that this mineral could be treated as being at instantaneous equilibrium, even though the hydraulic conductivity of the basement rock was high.

The minerals considered in each barrier material are shown in Table 6.2.2.5-5.

Table 6.2.2.5-5 Minerals considered in each barrier material

Barrier material	Initial minerals	Secondary minerals	
Cementitious material	Portlandite	C <sub>3</sub> ASH <sub>4</sub>	Friedel salt
	C-S-H gel	C <sub>2</sub> ASH <sub>8</sub>	Calcite
	C <sub>3</sub> AH <sub>6</sub>	C <sub>3</sub> AS <sub>3</sub>	Chalcedony
	Ettringite	Kaolinite	Analcite
	Brucite	Pyrophyllite	Laumontite
Host rock	Chalcedony	C-S-H gel	Brucite
	Amorphous silica	C <sub>3</sub> ASH <sub>4</sub>	Analcite
	Calcite	C <sub>2</sub> ASH <sub>8</sub>	Laumontite
	Montmorillonite	C <sub>3</sub> AS <sub>3</sub>	Cristobalite

#### f. Analytical cases

To evaluate the dispersion of the alkaline component from the disposal facility under different conditions, different cases assumed that the host rock was crystalline or sedimentary. Three different hydraulic conductivities were specified. In order to understand the effect of different groundwater compositions, fresh water groundwater and seawater type groundwater were assumed. Each evaluation treated the rock as an equivalent porous medium, but different porosities were used for the different rock types.

Parameters used in the simulations and the hydraulic characteristics of the engineered barriers and basement rock that were represented are summarized in Table 6.2.2.5-6.

Table 6.2.2.5-6 Parameter values, including physical properties, used in the simulations of a high-pH plume

Parameter		Unit	Crystalline rock	Sedimentary rock
Hydraulic conductivity	Hydraulic conductivity of cement	m s <sup>-1</sup>	4×10 <sup>-6</sup>	
Partition coefficient	Effective diffusion coefficient in cement	m <sup>2</sup> /s	8×10 <sup>-10</sup>	
	Effective diffusion coefficient in host rock	m <sup>2</sup> /s	8×10 <sup>-11</sup>	1.2×10 <sup>-9</sup>
Type	Characteristics of disposal facility	—	Waste Group 4	
Natural barrier	Hydraulic conductivity	m s <sup>-1</sup>	1×10 <sup>-8</sup> , 1×10 <sup>-9</sup> , 1×10 <sup>-10</sup>	
	Porosity	—	0.02	0.30
	True density	Mg/m <sup>3</sup>	2.7	2.7
	Hydraulic gradient	m/m	0.01	

#### g. Analytical results

For the crystalline bedrock case in which the hydraulic conductivity was 1×10<sup>-9</sup> m s<sup>-1</sup>, variations in the concentrations of solutes in the liquid phase and volumetric fractions of minerals in the solid phase near

the facility after 10,000 years are shown in Figures 6.2.2.5-2 to 6.2.2.5-5. The variations in pH are shown in Figures 6.2.2.5-6 and 6.2.2.5-7. Additionally, the relationships between temporal and spatial variations in pH in the repository and host rock are shown in Figure 6.2.2.5-8.

In the simulations, clinoptilolite was generated in the basement rock by alteration of clay minerals when free alkaline components such as Na and K migrated from the cement. In the case where the groundwater was of seawater type, since the Na concentration in the groundwater was high, after the dissipation of the free alkaline component from the cement, analcite was generated near the disposal facility by the gradual dissolution of clinoptilolite.

Additionally, in the region where the dissolution of amorphous silica continued, C-S-H gel with low Ca/Si ratio (Ca/Si=0.4) was generated. In the region where amorphous silica disappeared, the Ca/Si ratio gradually increased and C-S-H gel (Ca/Si=0.9) was generated by dissolution of chalcedony.

The pH in the host rock became high due to the migration of the free alkaline component from the cement. However, since clinoptilolite was generated when this free alkaline component migrated, K was fixed in the mineral phase and the fronts of Na and K migrated at different rates. Hence, the effect of the free alkalis on pH was decreased. After dissipation of the free alkalis, the pH reached 10.0 due to equilibrium with the C-S-H gel (Ca/Si=0.4) that was generated by the dissolution of amorphous silica. However, in the region where amorphous silica had disappeared, pH then reached 10.8 due to equilibrium with the C-S-H gel (Ca/Si=0.9) that was generated by dissolution of chalcedony. After 100,000 years, amorphous silica had disappeared up to a distance of 10 m from the disposal facility and pH reached 10.8.

The results show that the high-pH plume would be significantly affected by the flow rate in the natural barrier. In the case where the hydraulic conductivity of the natural barrier was  $1 \times 10^{-8} \text{ m s}^{-1}$ , the highly alkaline components in the disposal facility migrated downstream by advection. There was only a limited effect on the upstream side, where the HLW disposal facility would be located. Moreover, since the true flow velocity in the crystalline bedrock case was large, the effect on the upstream side was also small in the case where the hydraulic conductivity was  $1 \times 10^{-9} \text{ m s}^{-1}$ . In the case where the true flow velocity in the natural barrier was small, the alkali component migrated upstream by diffusion of the alkali components and the groundwater pH became higher than the initial value in the region between 100 – 200 m from the TRU waste repository.

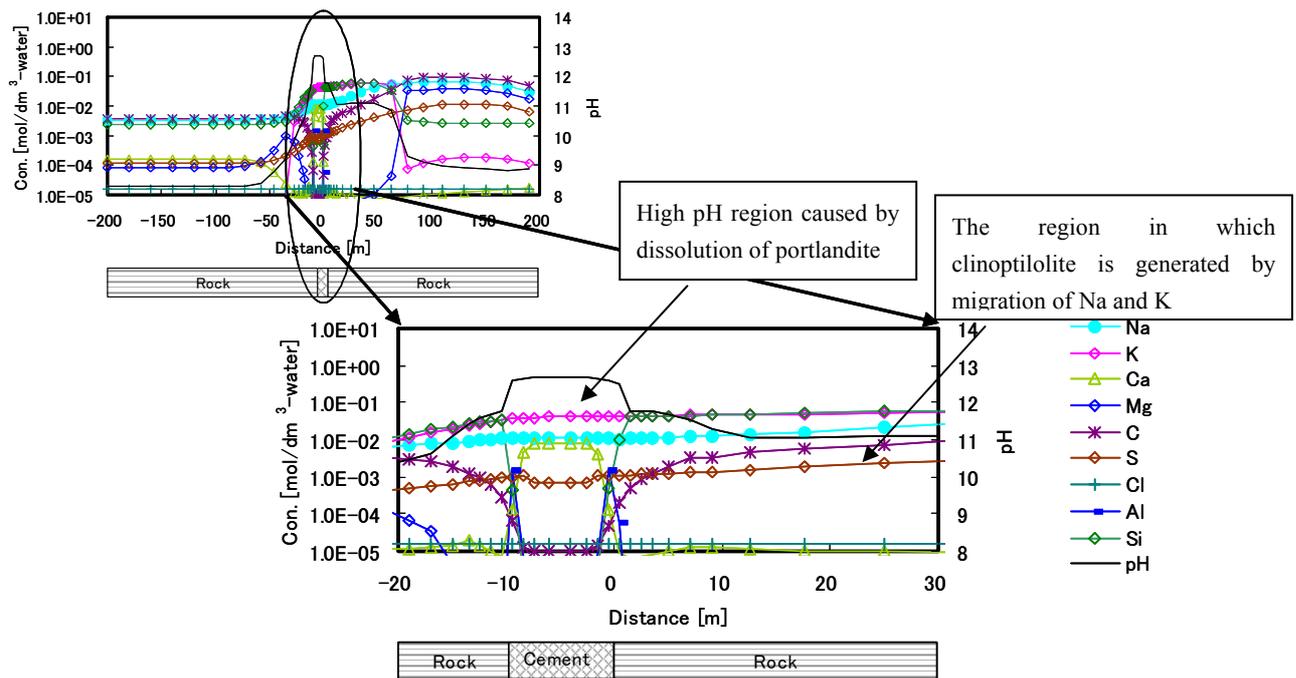


Figure 6.2.2.5-2 Distribution of solute concentrations in the liquid phase after 10,000 years (freshwater type groundwater, crystalline rock)

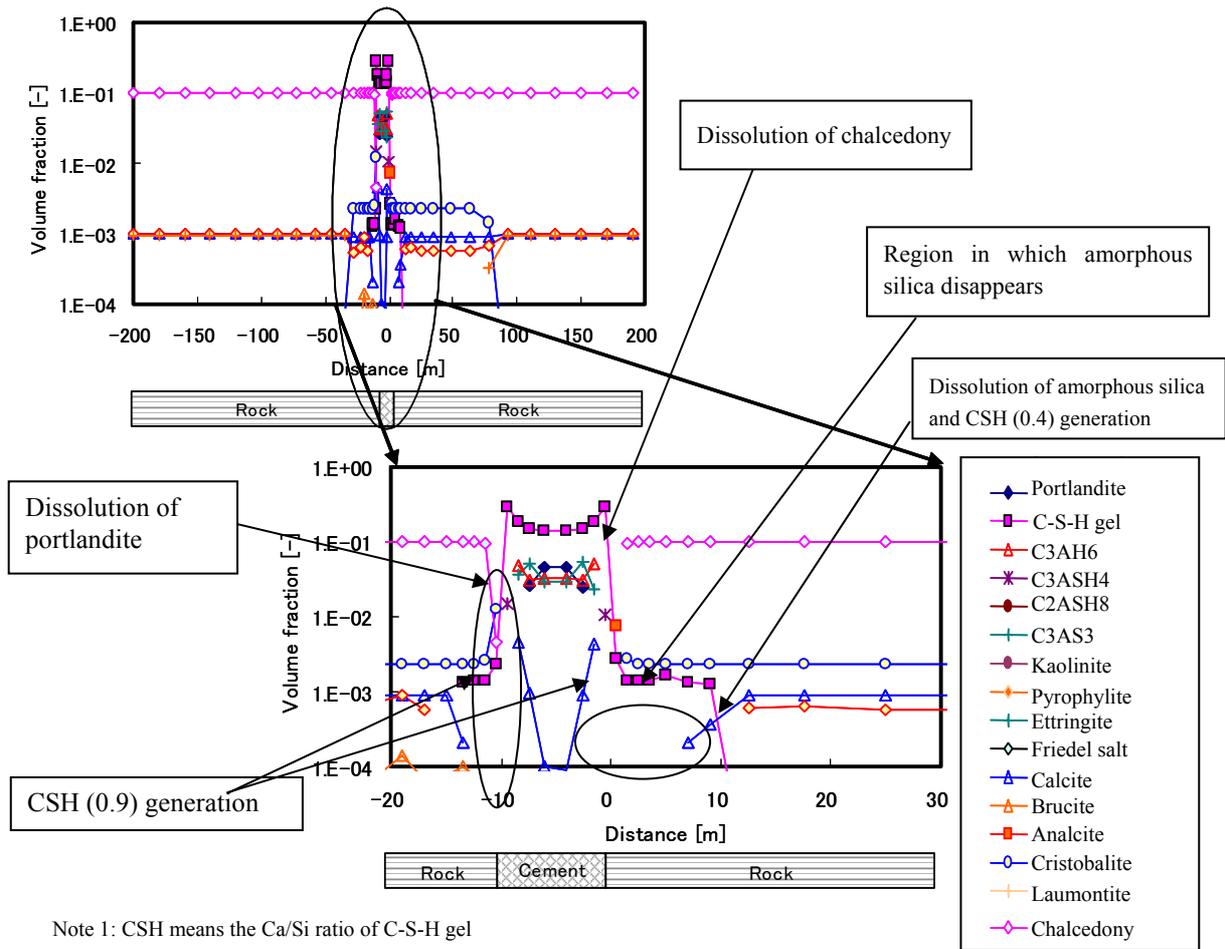


Figure 6.2.2.5-3 Spatial variations in the volumetric fractions of solid phases after 10,000 years (fresh water type groundwater, crystalline rock)

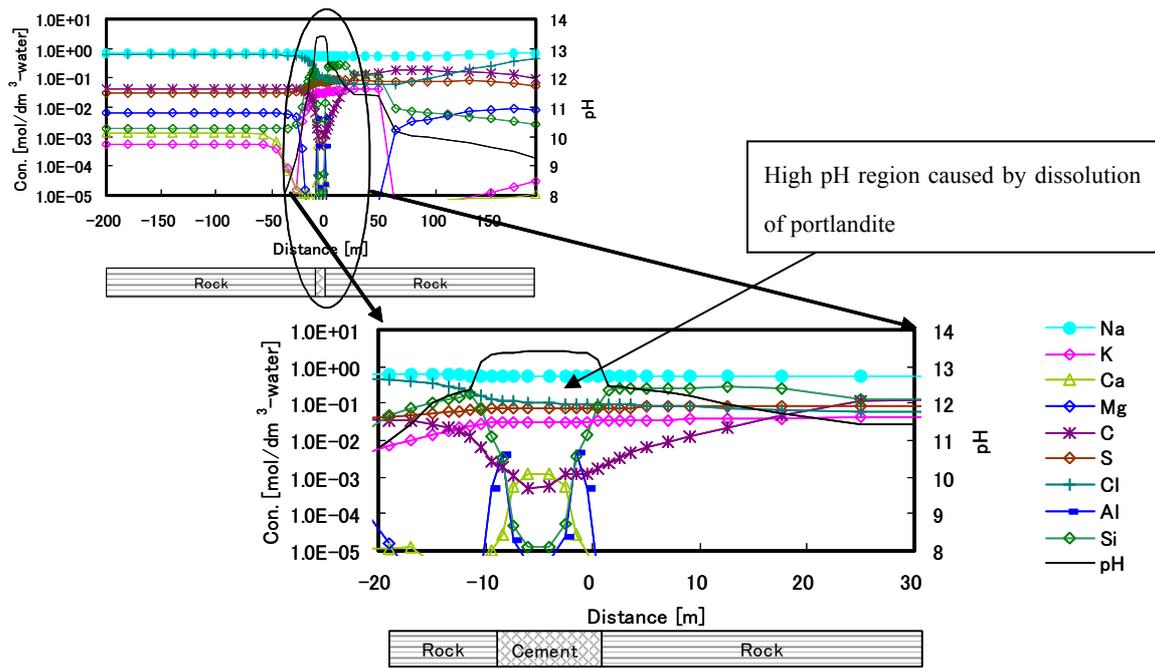


Figure 6.2.2.5-4 Distribution of solute concentrations in the liquid phase after 10,000 years  
(seawater type groundwater, crystalline rock)

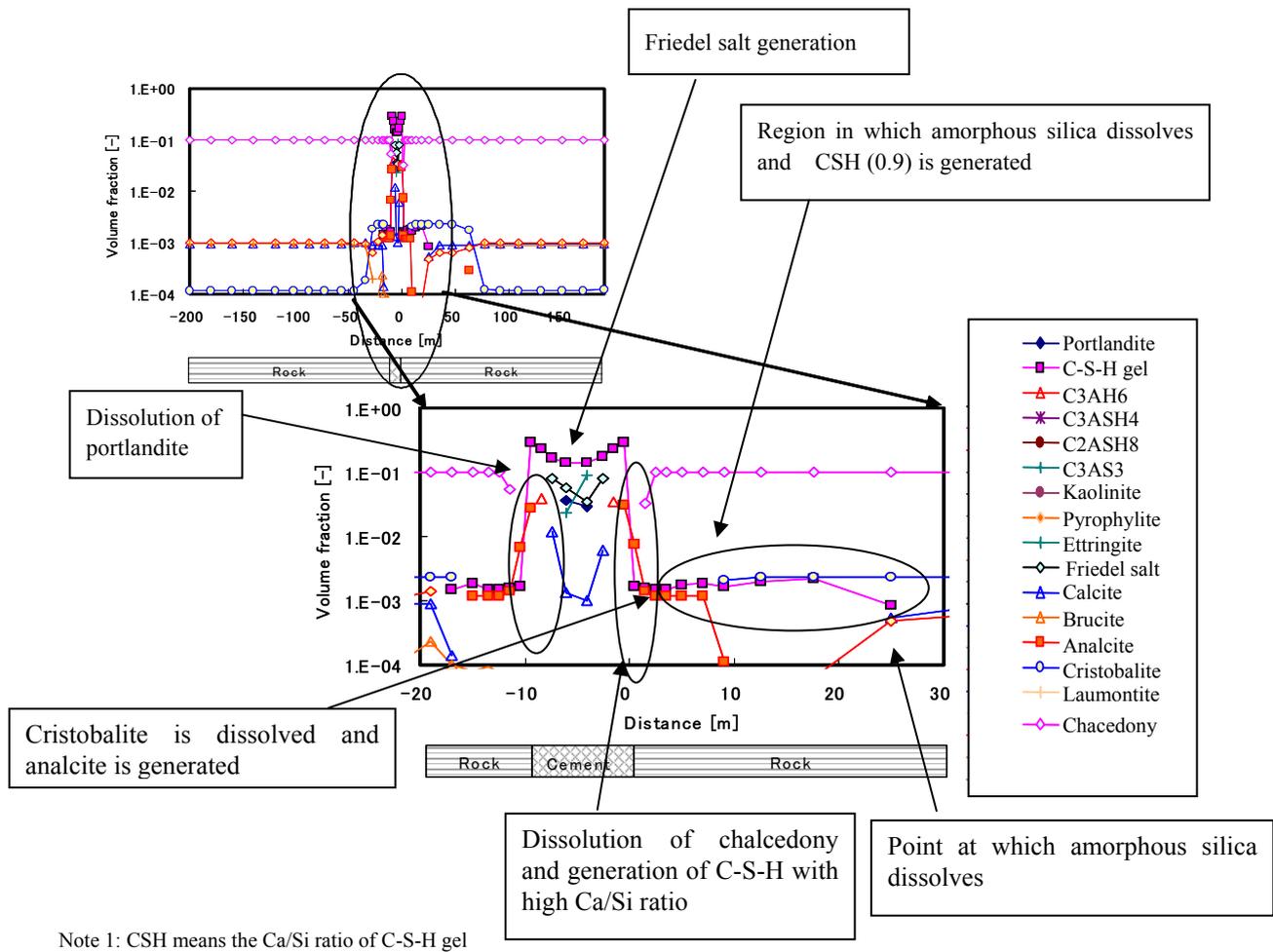


Figure 6.2.2.5-Spatial variations in the volumetric fractions of solid phases after 10,000 years (seawater type groundwater, crystalline rock)

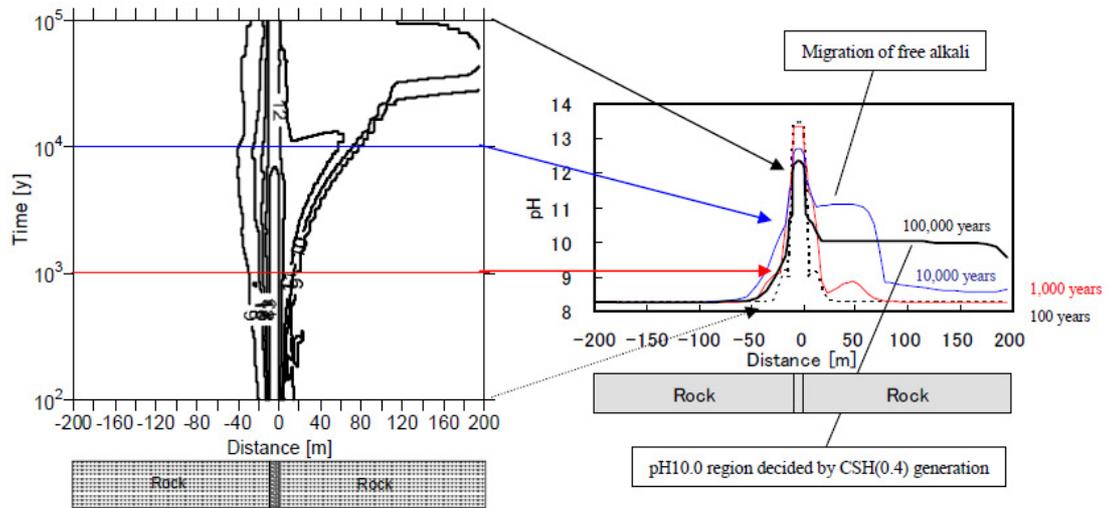


Figure 6.2.2.5-6 Variation in pH ( $1 \times 10^{-9} \text{ m s}^{-1}$ , freshwater groundwater, crystalline rock)

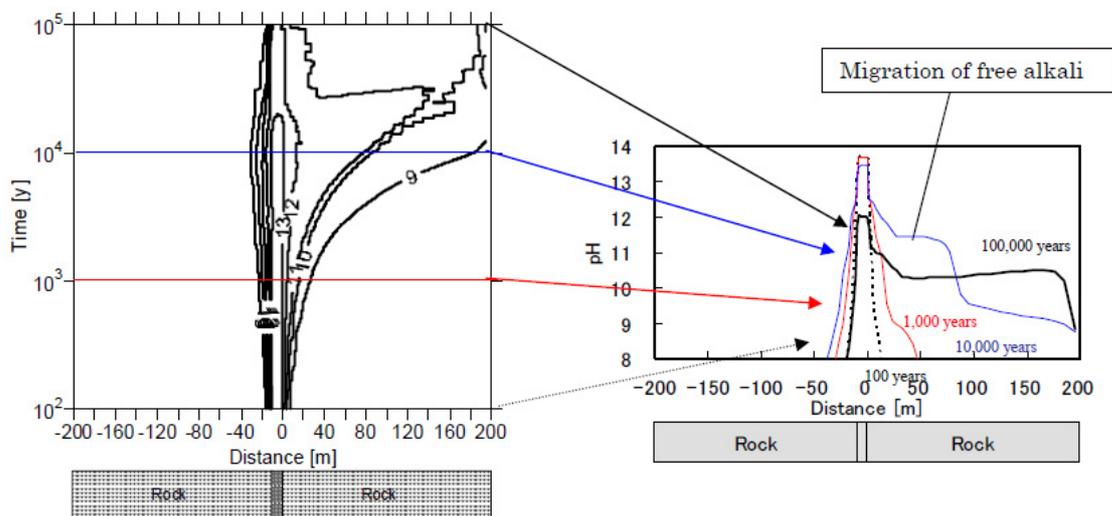
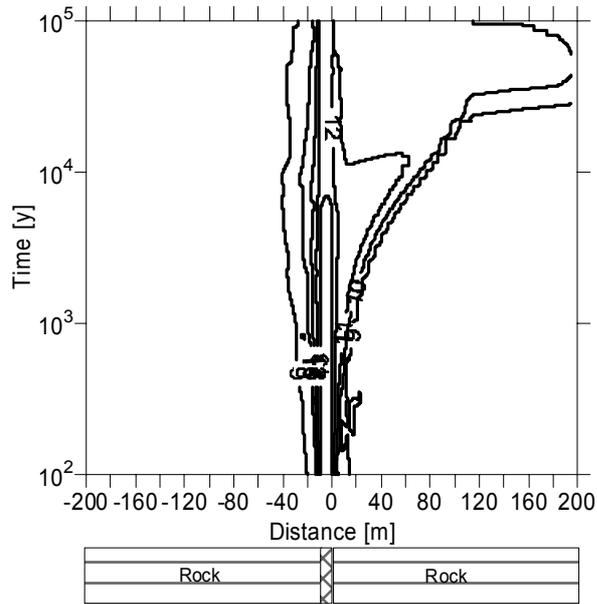
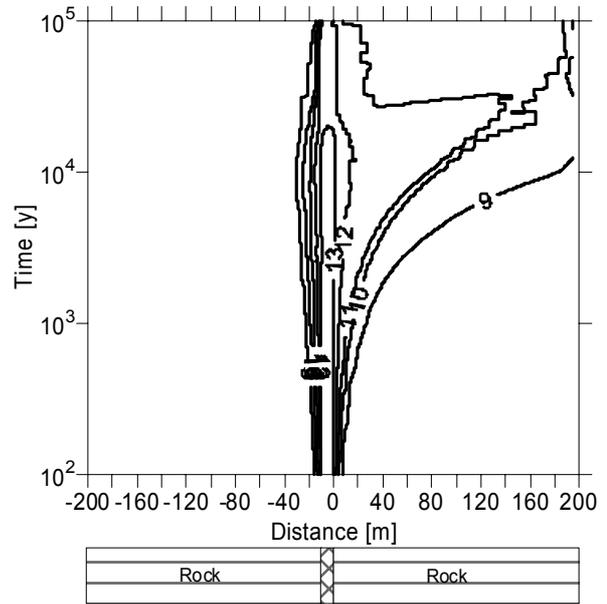


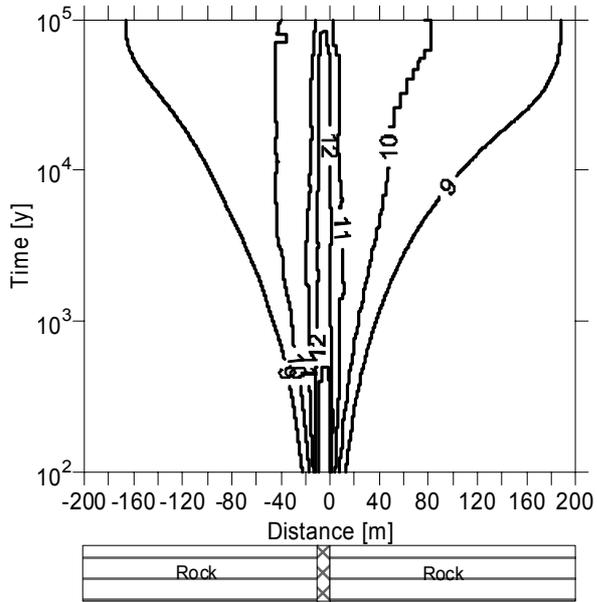
Figure 6.2.2.5-7 Variation in pH ( $1 \times 10^{-9} \text{ m s}^{-1}$ , seawater type groundwater, crystalline rock)



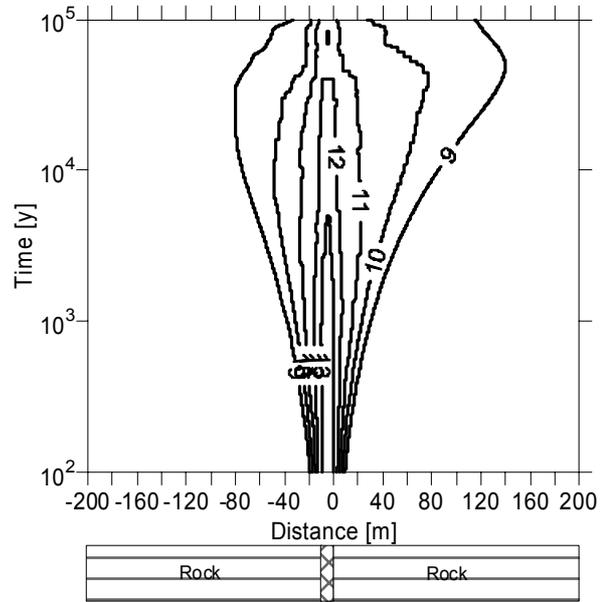
$1 \times 10^{-9} \text{ m s}^{-1}$ , crystalline rock, freshwater type groundwater



$1 \times 10^{-9} \text{ m s}^{-1}$ , crystalline rock, seawater type ground water



$1 \times 10^{-9} \text{ m s}^{-1}$ , sedimentary rock, Fresh water type groundwater



$1 \times 10^{-9} \text{ m s}^{-1}$ , sedimentary rock, seawater type ground water

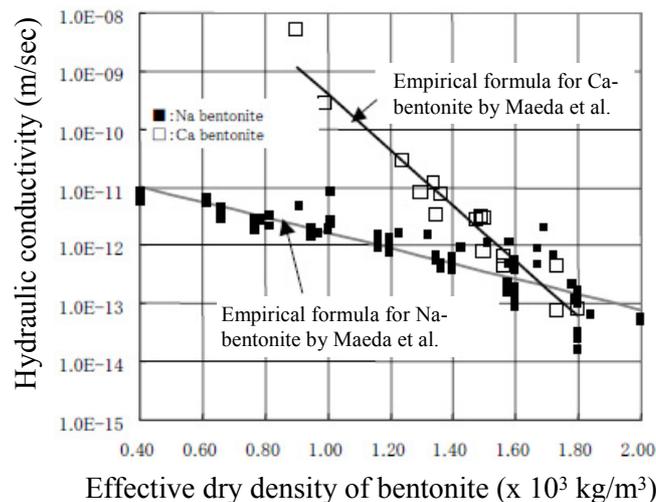
Figure 6.2.2.5-8 Temporal and spatial variations in pH

## (2) Summary of current knowledge concerning the effects of alkaline components

### a. Effects of alkaline components on engineered barrier performance

Among the components of cementitious material that are released into the surrounding groundwater are large quantities of  $\text{Ca}^{2+}$ . This  $\text{Ca}^{2+}$  replaces  $\text{Na}^+$  in smectite, which is converted into Ca-type smectite. Laboratory experiments have revealed that, in a high-pH environment, smectite, which is the main component of bentonite, is dissolved and a C-S-H gel and zeolites are generated as secondary minerals. This result is supported by natural analogue research.

The swelling capability of Ca-type bentonite is less than that of Na-type bentonite. As shown in Figure 6.2.2.5-9, if the bentonite has an effective dry density greater than 1.6 - 1.7  $\text{Mg/m}^3$ , the hydraulic conductivity would be similar, whether it is Ca-type or Na-type. However, if the effective dry density of the bentonite is below 1.6 - 1.7  $\text{Mg/m}^3$ , then its hydraulic conductivity will be greater if it is Ca-type rather than Na-type. The hydraulic conductivity of Ca-bentonite reaches about 1 order of magnitude higher than that of Na-bentonite at a dry density of 1.36  $\text{Mg/m}^3$  (dry density of the bentonite mixed with 30% sand is 1.6  $\text{Mg/m}^3$ ). However, as shown in Figure 6.2.2.5-10, if the dry density of the Ca-bentonite is in the range of 1.2 - 1.8  $\text{Mg/m}^3$ , the effective diffusion coefficient of the bentonite is similar to that of Na-bentonite.



Based on PNC TN8410 98-021 (Maeda et al., 1998)

Figure 6.2.2.5-9 Results of hydraulic conductivity tests on bentonite

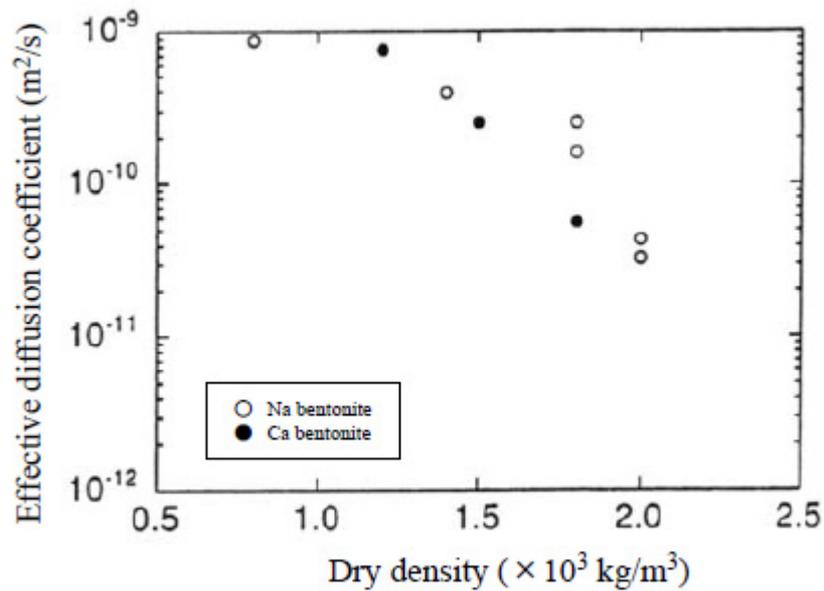


Figure 6.2.2.5-10 Diffusion coefficient of bentonite as a function of dry density (Mihara, 2000)

The alteration of smectite is observed to be pH- and temperature-dependent. Based on a report of Kuroki et al. (2000), and as shown in Table 6.2.2.5-7, the dissolution rate of smectite is slow when the temperature is below 100°C and the pH is 11. In this case, generation of secondary minerals was not observed in alteration experiments carried out over 2 years. This finding is consistent with the result of Bradbury and Baeyens (1997) who considered the stability limit of clay minerals to be at pH 11.

In Sweden, based primarily on present knowledge obtained in the ECOCLAY project, it is considered that from the point of view buffer and backfill stability, the pH of the groundwater should be less than 11 (SKB, 2004a; SKB, 2004b). As shown in Figure 6.2.2.5-11, Huertas et al. (2001) have shown that the dependence of smectite dissolution rate on pH has a gradient of -0.34. Sato et al. (2003) obtained gradients of 0.21 - -0.25.

These results show a decrease in the dissolution rate with decreasing pH. However, the dissolution rate of smectite is not only affected by pH, but is also regulated by temperature and surface area. Hence, the range of pH effects cannot be discussed simply in isolation.

Table 6.2.2.5-7 Alteration of bentonite in alkaline solution (Kuroki et al., 2000)

Immersion liquid	Ca(OH) <sub>2</sub>					
	10.5		11.5		12.5	
Mineral	Bentonite	Generated minerals	Bentonite	Generated minerals	Bentonite	Generated minerals
50°C	Ca-type formed, but mostly remaining at 720 days	Not observed until 720 days	Ca-type formed, but mostly remaining at 720 days	Not observed until 720 days	Mostly lost after 180 days	30 - 180 days: CAH and C-S-H 360 days: CAH 720 days: CASH
80°C	Ca-type formed, but mostly remaining at 720 days	Not observed until 720 days	Ca-type formed, slightly decreased at 720 days	180 - 360 days: mixed-layer mineral After 720 days: C-S-H and CASH	Mostly lost after 90 days	30 - 360 days: C-S-H and CASH 720 days: mixed-layer mineral
100°C	-----	-----	Ca-type formed, but mostly remaining until 14 days	Not observed until 14 days	Mostly lost after 7 days	7 - 30 days: C-S-H and CASH

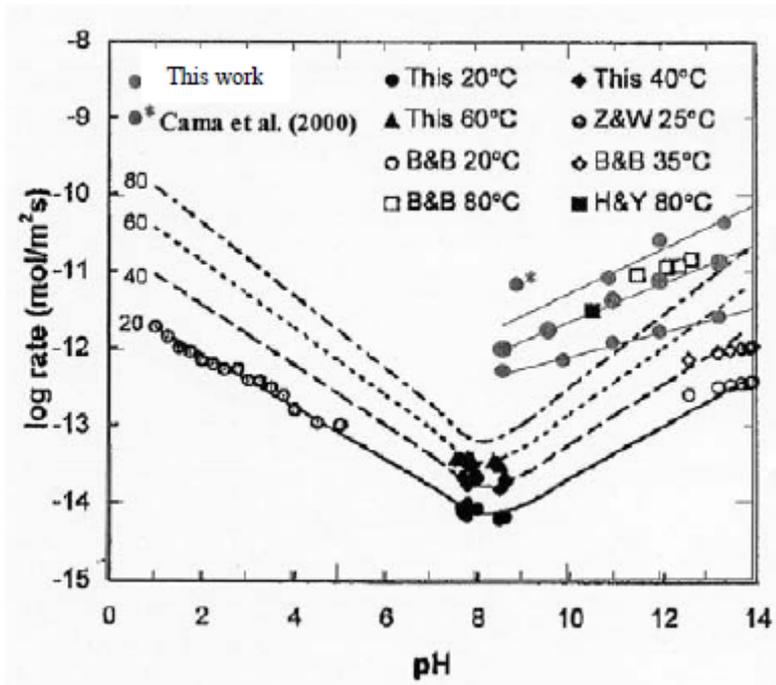


Figure 6.2.2.5-11 pH dependence of smectite dissolution rate (Sato et al., 2003)

As shown in Figure 6.2.5-12, if pH of the groundwater around the bentonite is below 12.5, the carbon steel which is used as the overpack for vitrified waste will not be passivated and corrosion will occur across the whole area (Taniguchi et al., 2002).

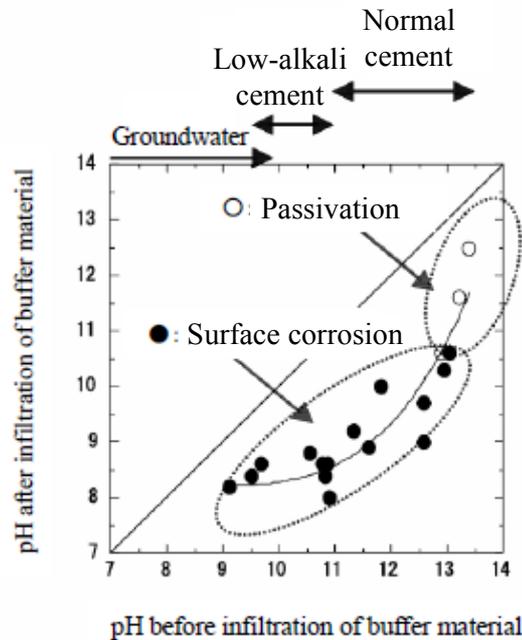


Figure 6.2.5-12 Conditions for passivation of carbon steel in buffer material

**b. Effects of alkaline components on the natural barrier**

The cementitious material that will be used in large quantities in a TRU waste disposal facility will be gradually dissolved by the groundwater. A highly alkaline solution will then expand into the surrounding host rock. Since the pH of this plume will be maintained above 12.5 on the long term (several 1,000 to several 100,000 years), the minerals in the host rock through which the high-pH groundwater migrates will be dissolved and secondary minerals will be generated. The groundwater chemistry will also be changed.

As a result of these processes, it is assumed that the pore structure and hydraulic conductivity of the host rock will vary and that the sorption distribution coefficient and solubility of radionuclides will be changed.

In Bradbury and Baeyens (1997), C-S-H, CASH and zeolites, etc. are considered to be the generated phases. However, these researchers reported that the system is very complex and that it changes temporally and spatially. Additionally, Nakazawa et al. (2004) reported that the dissolution of crystalline minerals was not observed during their experimental investigation of the alteration of basement rock by alkaline solutions. However, they reported that amorphous minerals were dissolved and that the Si released reacted with Ca from the cement to form C-S-H gel.

### **(3) Summary**

The pH variations in the natural barrier caused by a high-pH groundwater plume during a period of 100,000 years were evaluated for distances of 200 m upstream and downstream from the TRU waste disposal facility. The results showed that the high-pH plume could strongly affect the true flow velocity in the natural barrier. However, only a limited effect was revealed on the upstream side of the TRU waste disposal facility, where the HLW facility would be constructed. In the case where the true flow velocity in the natural barrier is small, migration by diffusion would become significant. In this case, alkaline components from the TRU waste facility could migrate in the upstream direction and cause the pH there to become higher than that of the initial groundwater. However, even in the high-pH case, the pH was found to be only about 9 to 10 across a wide area. The pH values of < 11 that would be needed to affect the buffer material of the HLW facility significantly (see (2) above) would occur close to the TRU waste facility. Consequently, at distances of more than a few tens of meters in the upstream direction, the effects of the alkaline plume are considered to be small.

Even in the case where long-term geological variations are assumed to cause groundwater flow to change so that flow from the TRU waste facility is in the direction of the HLW facility, a high pH due to Ca migration would occur only within a few tens of meters from the TRU waste facility.

Among the cases specified in this evaluation were ones in which a short-term high pH was assumed to be caused by the migration of Na and K from cement. However, it is considered that there are many uncertainties connected with the specification of the secondary minerals that form under these conditions.

From these results, it can be stated that, even when a change in flow direction is considered, if the TRU waste and HLW disposal facilities are separated by more than 100 m, any effects of the high-pH plume on the HLW facility will be insignificant.

## **6.2.3 Evaluation of co-location disposal**

In this section, based on the previous evaluation results, the concept of co-location disposal is described and evaluated.

### **(1) Reciprocal influences**

The potential interaction between the TRU waste and HLW disposal facilities caused by transport of heat, organic material, nitrate and a high-pH plume are evaluated. Considering the approaches adopted in foreign countries, the measures for avoiding these influences are as follows:

- ① ensuring a sufficient separation distance
- ② designing an appropriate facility layout (considering groundwater flow direction, fractures in host

rock)

- ③ adopting suitable engineered measures (e.g. specifying plugs)

By combining ①, ② and ③, it is possible to evaluate the effects of heat generation by the waste, components included in the waste (organic material, nitrate) and alkaline components from cementitious materials. Based on the previous analyses of interaction, disposal safety can be ensured by maintaining a separation distance of several 100 m between the TRU waste and HLW disposal facilities. It is also considered that changes in waste treatment methods and the specifications of the engineered barriers may also be effective in reducing interaction (cf. Sections 7.4 and 7.5).

## **(2) Evaluation of the disposal facility concept and layout**

Based on the above evaluation results, an outline conceptual illustration of co-location disposal of TRU waste and HLW has been prepared. The basic concept is as follows.

- ① Based on analytical results, in the case of general geological conditions, influences of the TRU waste disposal facility on the HLW repository are mostly prevented by ensuring a separation distance of 300 m. In foreign countries, the separation distance between HLW and TRU waste is specified to be between several 100 m and 500 m.
- ② Considering the above, the separation distance between the TRU waste and HLW disposal facilities is specified to be several 100 m. Plugs are also to be emplaced in the main tunnel and all the connecting tunnels in both disposal facilities. These engineering measures are performed to prevent the tunnels from becoming flow paths.
- ③ The TRU waste and HLW disposal facilities are constructed so as to prevent the various operations in one facility from interfering with those in the other facility.
- ④ In order to decrease the effects on the HLW facility of organic material, nitrate and high-pH pore water from the TRU waste disposal facility, the two facilities are constructed parallel to the groundwater flow direction.
- ⑤ Based on the above evaluation results, the co-location disposal of TRU waste and HLW is outlined in Figure 6.2.3-1. The right-hand side of this figure is an illustration of co-location disposal in which both facilities are constructed parallel to the groundwater flow direction and are separated by a distance of several 100 m. The left-hand figure shows a horizontal plane through the TRU waste disposal facility. Three-dimensional views of both the TRU waste and HLW facilities are shown in Figure 6.2.3-2. To ensure the required separation distance, it is possible to distribute the two facilities in three dimensions.

### **6.2.4 Summary**

By ensuring a separation distance of several 100 m between TRU waste and HLW disposal facilities, any interaction can generally be prevented. However, since the analysis of interaction contains numerous

conservative assumptions and there are uncertainties in evaluating fractured host rock, a detailed analysis should be performed under realistic geological conditions.

### **6.2.5 Future issues**

#### ① Evaluation of interaction

Although thermal and chemical (organic material, nitrate and high-pH plume) influences are evaluated in this report, it is necessary in future to give further consideration to mechanical and hydrogeological influences.

#### ② Assessment and evaluation of interaction

This report presents investigations and evaluations of interaction based on present knowledge, such as that obtained from the international literature. The individual issues for evaluation are basically the same as those described in Section 4.4. To evaluate the effects on the HLW disposal facility, it is necessary to develop an improved understanding of the effects of a high-pH plume on barrier materials in the HLW disposal facility. To this end, additional experimental research is necessary. Since a wide range of basement rocks are considered on an assessment scale, it is necessary to include site inhomogeneity in future evaluations.

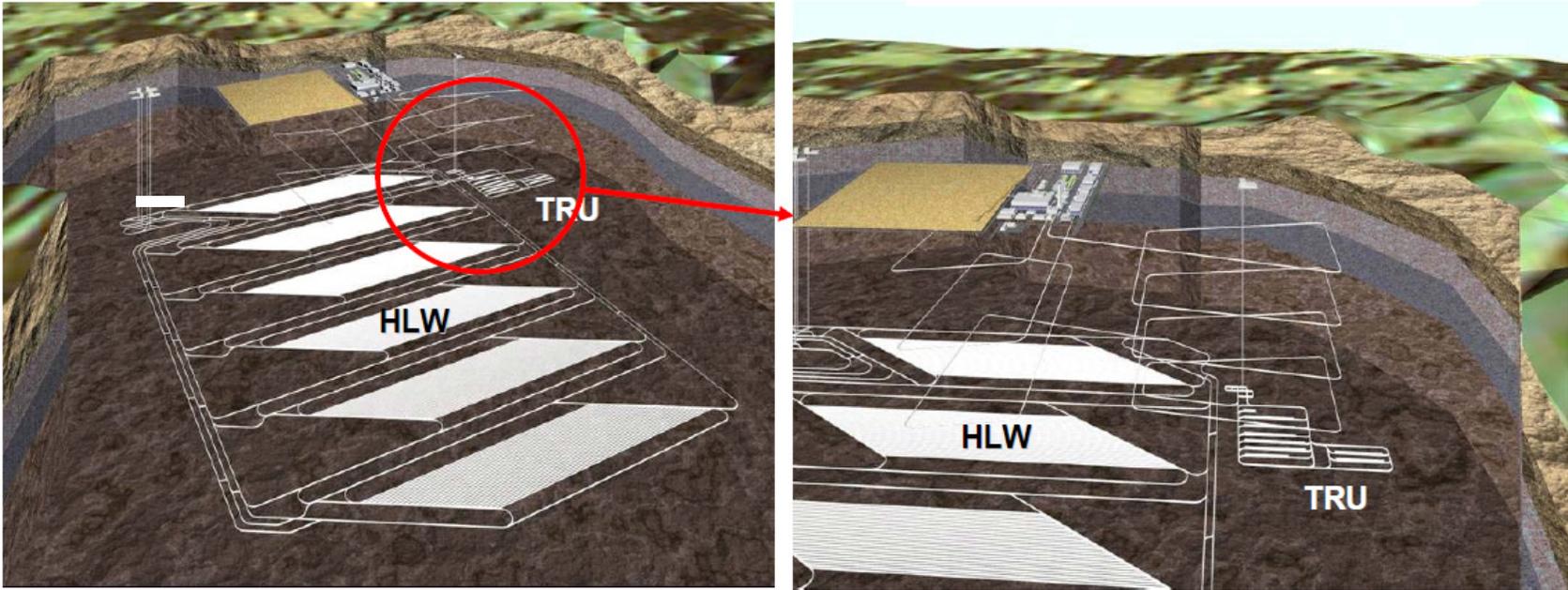
#### ③ Optimization of system design

It is necessary to evaluate in detail how uncertainties in the geological environment and data affect spatio-temporal variations in influences. Additionally, based on this evaluation, consideration should be given to optimizing aspects of the disposal concept, such as the separation distance, arrangement of the waste and engineering measures.



The layout of co-located disposal

Vicinity of the TRU waste disposal area



6-51

Figure 6.2.3-2 Co-location disposal concept of TRU waste and HLW

## **6.3 Effects of waste classification by $\alpha$ -emitting nuclide concentration at intermediate-depth disposal**

### **6.3.1 Need to evaluate the classification concept for intermediate-depth disposal**

As described previously, TRU waste has various characteristics and the concentrations of radionuclides that it contains vary over a wide range (Fig. 2.3-1). Hence, it is appropriate to classify the waste appropriately into waste that is suitable for each of the current disposal concepts (shallow disposal, intermediate-depth disposal and geological disposal). An upper concentration limit for  $\alpha$ -emitting nuclides may be appropriate to waste classification for each disposal concept.

The upper concentration limit for waste from nuclear reactors is specified in a report by the Atomic Energy Agency of Japan (AEC, 2000). However, limits for TRU waste generated from reprocessing facilities have not yet been decided.

An upper concentration limit for all  $\alpha$ -emitting nuclides of 1 GBq/t is used for concrete vault disposal. Intermediate-depth disposal is considered in a report by the Atomic Energy Commission of Japan (AEC, 2000) to be appropriate for waste that contains  $\alpha$ -emitting nuclides at concentrations in excess of this value (1 GBq/t).

Since the amount of waste appropriate for each disposal concept will change depending on the classification values chosen, in this evaluation the upper concentration limits for  $\alpha$ -emitting nuclides are estimated in order to construct an optimum disposal system. In this way, each disposal concept is used in the most effective way possible. Since the concentrations of  $\alpha$ -emitting nuclides will significantly affect the quantities of waste for intermediate-depth disposal, the effect of the classification on the safety of this type of disposal is also evaluated.

In the calculation of  $\alpha$ -emitting nuclide concentrations to be used as classification values, safety is ensured by requiring that the dose arising from the waste disposed of according to each concept is below the regulatory limit. However, in future these doses must be considered further by all relevant organizations, taking into account risk theory and the concept of geological disposal at intermediate depth.

### **6.3.2 Upper concentration limit**

The upper concentration limit for radionuclides in TRU waste has been derived using the same

method as that adopted for radioactive waste from nuclear reactors. The upper concentration limits of important nuclides are estimated from dose conversion factors in ICRP Pub.72 and nuclide compositions of TRU waste. The evaluation results are shown in Table 6.3.2-1.

As shown in this table, the nuclides to be assessed are influenced by the characteristics of the facility generating the waste. However, the calculated upper concentration limits are not significantly different from the values for waste from nuclear reactor facilities. In the case of intermediate-depth disposal, the values are also not significantly different from those estimated by the Japan Atomic Energy Research Institute.

Table 6.3.2-1 Estimated upper limits of nuclide concentrations in TRU waste

(Unit: Bq/t)

Nuclide	Intermediate-depth disposal			Concrete vault disposal		Trench disposal	
	TRU waste	(Ref)* <sup>1</sup> Power plant waste	(Ref)* <sup>2</sup> TRU waste (JAERI calculation)	TRU waste	(Ref)* <sup>1</sup> Power plant waste	TRU waste	(Ref)* <sup>1</sup> Power plant waste
H-3	—	—	—	—	—	—	3.0×10 <sup>9</sup>
C-14	—	5.2×10 <sup>14</sup>	5.2×10 <sup>14</sup>	3.4×10 <sup>10</sup>	3.7×10 <sup>10</sup>	—	1.1×10 <sup>8</sup>
Cl-36	9.3×10 <sup>10</sup>	1.0×10 <sup>11</sup>	9.7×10 <sup>10</sup>	—	—	—	—
Co-60	—	—	—	4.1×10 <sup>13</sup>	1.1×10 <sup>13</sup>	6.9×10 <sup>9</sup>	8.1×10 <sup>9</sup>
Ni-63	—	—	—	8.9×10 <sup>11</sup>	1.1×10 <sup>12</sup>	7.2×10 <sup>9</sup>	7.2×10 <sup>9</sup>
Sr-90	—	—	—	6.5×10 <sup>10</sup>	7.4×10 <sup>10</sup>	6.0×10 <sup>6</sup>	4.7×10 <sup>6</sup>
Nb-94	—	—	—	1.1×10 <sup>10</sup>	—	—	—
Tc-99	4.4×10 <sup>11</sup>	8.2×10 <sup>11</sup>	4.6×10 <sup>11</sup>	—	—	—	—
I-129	2.2×10 <sup>10</sup>	—	2.3×10 <sup>10</sup>	1.4×10 <sup>8</sup>	—	—	—
Cs-137	—	—	—	1.0×10 <sup>12</sup>	1.1×10 <sup>12</sup>	9.7×10 <sup>7</sup>	1.0×10 <sup>8</sup>
α- emitting nuclide	1.6×10 <sup>14</sup> (Pu-238) 1.8×10 <sup>14</sup> (Am-241)	1.3×10 <sup>10</sup> (Np-237)	2.6×10 <sup>14</sup> (Am-241)	2.5×10 <sup>9</sup> (Am-241)	1.1×10 <sup>9</sup> (Am-241)	7.2×10 <sup>7</sup> (Am-241)	1.7×10 <sup>7</sup> (Am-241)

\*1 : safety regulation of low-level solid waste land disposal in Japan (AEC, 2000)

\*2 : JAERI, 2004

### 6.3.3 Upper limit of α-emitting nuclide concentration

As described before, the possibility of intermediate-depth disposal for TRU waste which is over the target classification value (1GBq/t) is described in an expert report (NSC, 2000). In this section, the limit appropriate for such waste is calculated using several concepts by Nuclear Safety Commission of Japan (NSC, 2004), as follows:

- ① The limiting total concentration of α-emitting nuclides (below the dose regulation value) after the institutional control period is taken to be the maximum dose that would arise as a consequence of the unlikely event that a repository at intermediate depth would be affected by human intrusion (drilling).

- ② The limiting concentration of  $\alpha$ -emitting nuclides is such that, during the period within which the geology of the disposal facility at intermediate depth is stable and society is unchanged, the concentration of  $\alpha$ -emitting nuclides within the facility decreases by radioactive decay to below the limit appropriate for shallow concrete vault disposal (1 GBq/t).
- ③ The limiting concentration of  $\alpha$ -emitting nuclides is such that the temporal change of potential impacts (= radioactivity/annual intake limit) on the long term is the same as the temporal change in the potential impacts of the low-level waste targeted for intermediate-depth disposal.

To evaluate the drilling scenario (①), an preliminary risk assessment was performed since there is no established method for estimating risk taking into account quantitative probabilities. The probability of occurrence, which is given by drilling frequency (about  $2 \times 10^{-8}/\text{m}^2/\text{year}$  at below 50 m) (JAERI, 2004), and the area allocated for intermediate-depth disposal (about  $300 \text{ m} \times 500 \text{ m}$ ) is estimated to be about  $3 \times 10^{-4}/\text{year}$ , i.e. the probability is in the order  $10^{-3}/\text{year}$ .

If the probability of likely phenomena is estimated to be in the order  $10^0/\text{year}$ , then the relative probability of low-frequency phenomena is estimated to be about  $10^{-3}/10^0=10^{-3}$ . If the dose standard for likely phenomena is  $10 \mu\text{Sv}/\text{y}$ , the target dose for the low-frequency event is 1000 times larger and becomes  $10 \text{ mSv}/\text{y}$ , taking into account the above probability.

If the dose limit of  $10 \text{ mSv}/\text{occurrence}$  is applied to the drilling scenario for the TRU waste disposal facility, the  $\alpha$ -emitting nuclide concentration value for classification purposes is estimated to be about 200 GBq/t. In this report, the target dose ( $10 \text{ mSv}/\text{y}$ ) for low-frequency events is consistent with the general reference level in ICRP Pub.81. Moreover, this value is an intermediate dose limit, between those appropriate for the public ( $1 \text{ mSv}/\text{y}$ ) and workers in the nuclear industry ( $50 \text{ mSv}/\text{y}$  and  $100 \text{ mSv}/5\text{y}$ ).

The estimated upper limit in ② above depends on the specified period for which the geology of the disposal facility is stable and there are no significant changes in social circumstances. This period is the time available for the  $\alpha$ -emitting nuclides present initially to decay so that their total concentration is less than the limit for shallow concrete vault disposal (1 GBq/t). Generally, the period during which there will be no significant geological or environmental changes is considered to be several tens of thousands of years. The total concentration of  $\alpha$ -emitting nuclides in the main waste is estimated to decrease to about 1/20 of the original value after 10,000 years and to about 1/100 of the original value after 50,000 years (cf. Figure 6.3.3-1). The limiting initial concentrations of  $\alpha$ -emitting nuclides that would decrease to 1 GBq/t after these periods are estimated to be about

20 GBq/t and 100 GBq/t, respectively. Thus, according to this concept, the  $\alpha$ -emitting nuclide concentration to be used for classification is considered to lie between several tens of GBq/t and 100 GBq/t.

The limiting  $\alpha$ -emitting nuclide concentration in the case of ③ is estimated to be a maximum of about 20 GBq/t, based on a report published by the Atomic Energy Commission of Japan (AEC, 2000).

As mentioned above, the upper limit of  $\alpha$ -emitting nuclides concentration which can classify radioactive waste between intermediate depth disposal and geological disposal is estimated to be about 20 GBq/t to 200 GBq/t. In this report, the  $\alpha$ -emitting nuclide concentration that will be used for classification purposes in later discussions is specified to be an intermediate value of 100 GBq/t.

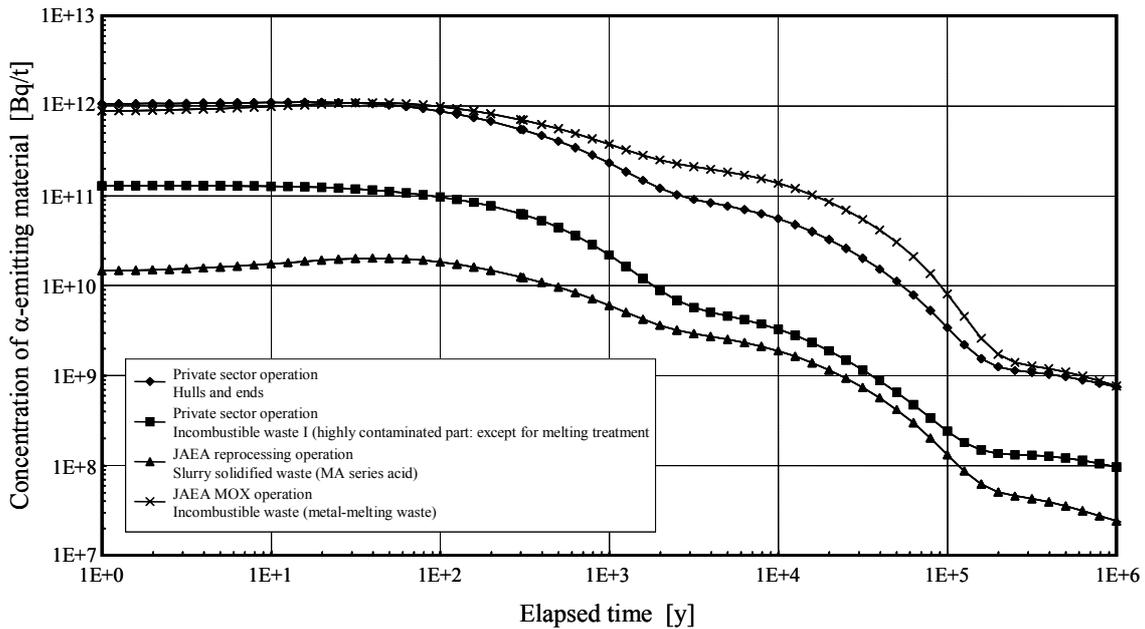


Figure 6.3.3-1 Concentrations of  $\alpha$ -emitting nuclides in the main waste

### 6.3.4 Amounts of waste generated

Based on the concepts discussed in the previous section, the quantities of waste that will be appropriate for intermediate-depth disposal and geological disposal are shown in Table 6.3.4-1. A comparison with amounts of waste generated as described in Section 2.4.2 is shown in Table 6.3.4-2. Additionally, the radioactivity are compared in Table 6.3.4-3.

In the case where the  $\alpha$ -emitting nuclide concentration used for waste classification is 100 Gq/t, the amount of waste that is suitable for intermediate-depth disposal increases by about 14,000 m<sup>3</sup> and the amount of waste for geological disposal decreases by about 14,000 m<sup>3</sup>. The quantities of fission products or actinides in waste that is suitable for intermediate-depth disposal increase by 1 order of magnitude due to the change in the  $\alpha$ -emitting nuclide concentration used for classification. However, the total amount of radioactive material for geological disposal is hardly changed.

Table 6.3.4-1 Amounts of waste generated in each disposal category  
 (upper limit of  $\alpha$ -emitting nuclide concentration: 100 GBq/t)

(Unit: m<sup>3</sup>)

Waste classification	Concrete vault disposal	intermediate -depth disposal	geological disposal	Total
Private sector reprocessing/ MOX operating waste	24,667	20,540	6,165	51,372
Private sector waste reprocessing and dismantling	36,647	7,181	431	44,260
Private sector MOX dismantling waste	491	1,338	175	2,004
JAEA reprocessing waste	10,388	7,607	291	18,286
JAEA MOX dismantling waste	0	400	907	1,307
JAEA waste reprocessing and dismantling	7,238	1,070	236	8,544
JAEA MOX dismantling waste	0	1,159	886	2,045
Returned low-level Waste (COGEMA)	0	0	937	937
Returned low-level Waste (BNGS)	9,000	0	2,520	11,520
Total	88,431	39,295	12,548	140,274

Table 6.3.4-2 Comparison between amounts of waste generated in each disposal category

(Unit: m<sup>3</sup>)

Waste classification	intermediate-depth disposal		geological disposal	
	$\alpha$ : 1 G Bq/t	$\alpha$ : 100 GBq/t	$\alpha$ : 1 GBq/t	$\alpha$ : 100 GBq/t
Private sector reprocessing/ MOX operating waste	13,276	20,540	13,429	6,165
Private sector waste reprocessing and dismantling	6,974	7,181	639	431
Private sector MOX dismantling waste	1,250	1,338	263	175
JAEA reprocessing waste	3,410	7,607	4,488	291
JAEA MOX dismantling waste	14	400	1,293	907
JAEA waste reprocessing and dismantling	256	1,070	1,051	236
JAEA MOX dismantling waste	25	1,159	2,020	886
Returned low-level waste (COGEMA)	0	0	937	937
Returned low-level Waste (BNGS)	0	0	2,520	2,520
Total	25,205	39,295	26,640	12,548

Table 6.3.4-3 Radioactivity in each disposal category (Bq)

Nuclide	intermediate-depth disposal		geological disposal	
	$\alpha$ : 1 GBq/t	$\alpha$ : 100 GBq/t	$\alpha$ : 1 GBq/t	$\alpha$ : 100 GBq/t
H-3	$4.5 \times 10^{14}$	$5.5 \times 10^{14}$	$7.1 \times 10^{17}$	$7.1 \times 10^{17}$
C-14	$3.4 \times 10^{14}$	$3.4 \times 10^{14}$	$5.7 \times 10^{14}$	$5.6 \times 10^{14}$
Cl-36	$5.6 \times 10^{12}$	$5.9 \times 10^{12}$	$8.8 \times 10^{12}$	$8.6 \times 10^{12}$
Co-60	$6.5 \times 10^{17}$	$6.5 \times 10^{17}$	$1.6 \times 10^{18}$	$1.6 \times 10^{18}$
Ni-59	$1.4 \times 10^{15}$	$1.4 \times 10^{15}$	$7.2 \times 10^{15}$	$7.2 \times 10^{15}$
Ni-63	$2.1 \times 10^{17}$	$2.1 \times 10^{17}$	$1.2 \times 10^{18}$	$1.2 \times 10^{18}$
Se-79	$8.6 \times 10^{10}$	$9.1 \times 10^{10}$	$2.7 \times 10^{12}$	$2.7 \times 10^{12}$
Sr-90	$2.2 \times 10^{15}$	$5.0 \times 10^{15}$	$5.9 \times 10^{17}$	$5.4 \times 10^{17}$
Zr-93	$1.5 \times 10^{14}$	$1.5 \times 10^{14}$	$3.1 \times 10^{14}$	$3.0 \times 10^{14}$
Nb-94	$1.2 \times 10^{14}$	$1.2 \times 10^{14}$	$2.6 \times 10^{15}$	$2.6 \times 10^{15}$
Mo-93	$4.5 \times 10^{12}$	$4.5 \times 10^{12}$	$5.2 \times 10^{13}$	$5.2 \times 10^{13}$
Tc-99	$8.0 \times 10^{11}$	$1.8 \times 10^{13}$	$6.6 \times 10^{14}$	$6.4 \times 10^{14}$
Pd-107	$3.3 \times 10^9$	$4.2 \times 10^9$	$5.5 \times 10^{11}$	$5.4 \times 10^{11}$
Ag-108m	$3.8 \times 10^{12}$	$3.8 \times 10^{12}$	$2.1 \times 10^{12}$	$2.1 \times 10^{12}$
Sn-126	$2.3 \times 10^{10}$	$1.0 \times 10^{12}$	$6.1 \times 10^{12}$	$5.0 \times 10^{12}$
I-129	$1.5 \times 10^{11}$	$1.0 \times 10^{12}$	$5.2 \times 10^{13}$	$5.1 \times 10^{13}$
Cs-135	$1.4 \times 10^{10}$	$3.2 \times 10^{11}$	$3.4 \times 10^{12}$	$3.1 \times 10^{12}$
Cs-137	$3.0 \times 10^{15}$	$7.0 \times 10^{16}$	$7.6 \times 10^{17}$	$7.0 \times 10^{17}$
Pu-241	$3.2 \times 10^{15}$	$1.2 \times 10^{16}$	$5.7 \times 10^{17}$	$5.6 \times 10^{17}$
Am-242m	$2.4 \times 10^{11}$	$2.0 \times 10^{13}$	$6.2 \times 10^{13}$	$4.2 \times 10^{13}$
$\beta\gamma$ total	$4.6 \times 10^{18}$	$4.8 \times 10^{18}$	$1.7 \times 10^{19}$	$1.7 \times 10^{19}$
U-233	$5.0 \times 10^9$	$5.0 \times 10^9$	$4.7 \times 10^9$	$4.7 \times 10^9$
U-234	$1.7 \times 10^9$	$4.8 \times 10^{10}$	$1.7 \times 10^{12}$	$1.6 \times 10^{12}$
U-235	$6.9 \times 10^7$	$1.6 \times 10^{10}$	$1.1 \times 10^{11}$	$1.1 \times 10^{11}$
U-236	$1.0 \times 10^9$	$1.2 \times 10^{10}$	$1.2 \times 10^{12}$	$1.2 \times 10^{12}$
U-238	$9.2 \times 10^8$	$1.2 \times 10^{10}$	$1.1 \times 10^{12}$	$1.1 \times 10^{12}$
Np-237	$1.3 \times 10^9$	$5.7 \times 10^{10}$	$2.4 \times 10^{12}$	$2.3 \times 10^{12}$
Pu-238	$1.1 \times 10^{13}$	$1.7 \times 10^{14}$	$1.7 \times 10^{16}$	$1.7 \times 10^{16}$
Pu-239	$1.1 \times 10^{12}$	$3.2 \times 10^{13}$	$2.4 \times 10^{15}$	$2.4 \times 10^{15}$
Pu-240	$1.7 \times 10^{12}$	$4.0 \times 10^{13}$	$3.2 \times 10^{15}$	$3.2 \times 10^{15}$
Pu-242	$6.6 \times 10^9$	$1.1 \times 10^{11}$	$1.1 \times 10^{13}$	$1.1 \times 10^{13}$
Am-241	$3.0 \times 10^{12}$	$1.1 \times 10^{14}$	$6.1 \times 10^{15}$	$6.1 \times 10^{15}$
Am-243	$6.7 \times 10^{10}$	$1.7 \times 10^{12}$	$2.1 \times 10^{14}$	$2.1 \times 10^{14}$
Cm-244	$7.8 \times 10^{12}$	$1.1 \times 10^{14}$	$9.4 \times 10^{15}$	$9.3 \times 10^{15}$
Cm-245	$8.7 \times 10^8$	$2.6 \times 10^{10}$	$9.8 \times 10^{11}$	$9.5 \times 10^{11}$
$\alpha$ total	$2.5 \times 10^{13}$	$4.8 \times 10^{14}$	$3.9 \times 10^{16}$	$3.9 \times 10^{16}$

### 6.3.5 Safety assessment for intermediate-depth disposal

Since the radioactivity and quantities of waste that are suitable for intermediate depth disposal may change, this section evaluates the dose arising from intermediate-depth disposal based on the assessment model and parameters in sections 5.3.3 and 5.3.4.

The dose assessment results obtained when an  $\alpha$ -emitting nuclide concentration of 100 GBq/t was used for waste classification are shown in Table 6.3.5-1 and Figure 6.3.5-1. Additionally, a comparison of dose assessment results for each classification value is shown in Table 6.3.5-2.

By changing the  $\alpha$ -emitting nuclide concentration used for waste classification, the radioactivity contributed by  $\alpha$ -emitting nuclides and fission products increases by between 1 and 2 orders of magnitude. However, the main nuclide to dominate dose rate in the case of the river water use pathway is C-14. Therefore, the effect on the dose is considered to be small since the contribution of  $\alpha$ -emitting nuclides and fission products is small.

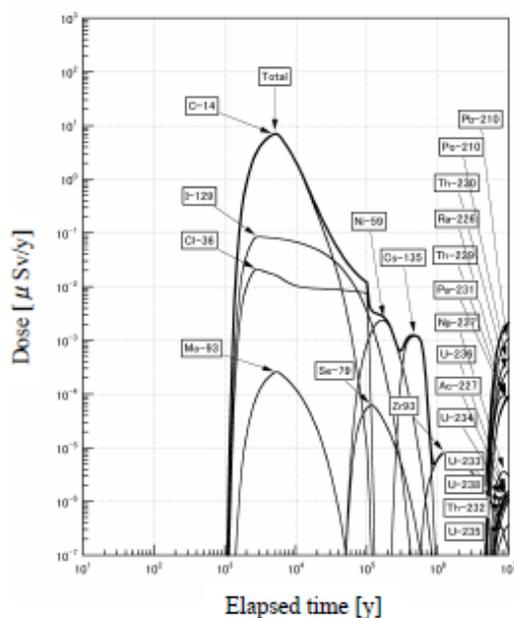
In the case of the river shore construction pathway and river shore residential pathway, daughter nuclides of  $\alpha$ -emitting particles become the main nuclides contributing to the dose. Hence, the dose increases by 1 order of magnitude when the  $\alpha$ -emitting nuclide concentration used for waste classification is changed. However, the dose is smaller than that obtained in the case of the river water use pathway.

Table 6.3.5-1 Result of dose assessment for intermediate-depth disposal  
(upper limit of  $\alpha$ -emitting nuclide concentration: 100 GBq/t)

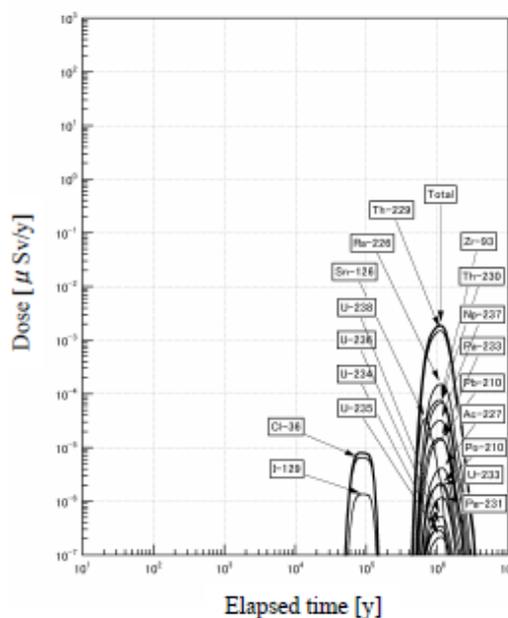
Evaluation pathway		Dose ( $\mu$ Sv/y)	Time (y)	Main nuclide
river water use	Base case	$9.3 \times 10^{-2}$	$8.6 \times 10^4$	I-129, Tc-99
	Alternative case	6.9	$5.0 \times 10^3$	C-14
River shore construction	Base case	$1.9 \times 10^{-3}$	$1.1 \times 10^6$	Th-229
River shore residential	Base case	$4.0 \times 10^{-3}$	$1.1 \times 10^6$	Th-229, Ra-226
River shore agriculture	Base case	$1.1 \times 10^{-1}$	$7.9 \times 10^4$	Cl-36

Table 6.3.5-2 Dose comparison for intermediate-depth disposal

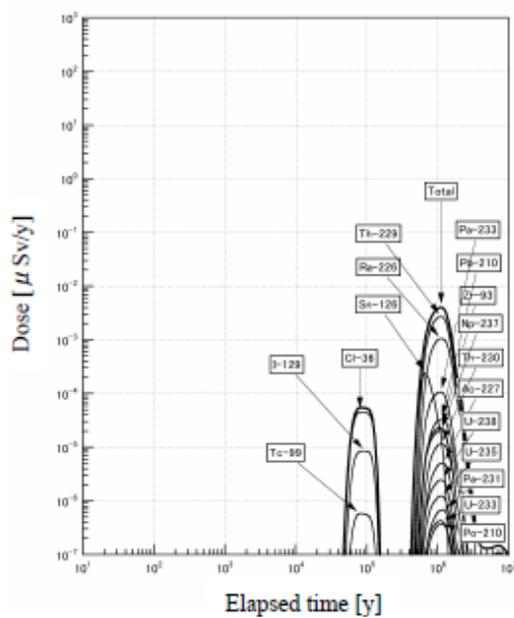
Evaluation pathway		Dose ( $\mu\text{Sv/y}$ )	
		$\alpha$ : 1 GBq/t	$\alpha$ : 100 GBq/t
river water use	Base case	$2.3 \times 10^{-2}$	$9.3 \times 10^{-2}$
	Alternative case	6.8	6.9
River shore construction	Base case	$3.1 \times 10^{-4}$	$1.9 \times 10^{-3}$
River shore residential	Base case	$5.0 \times 10^{-4}$	$4.0 \times 10^{-3}$
River shore agriculture	Base case	$6.3 \times 10^{-2}$	$1.1 \times 10^{-1}$



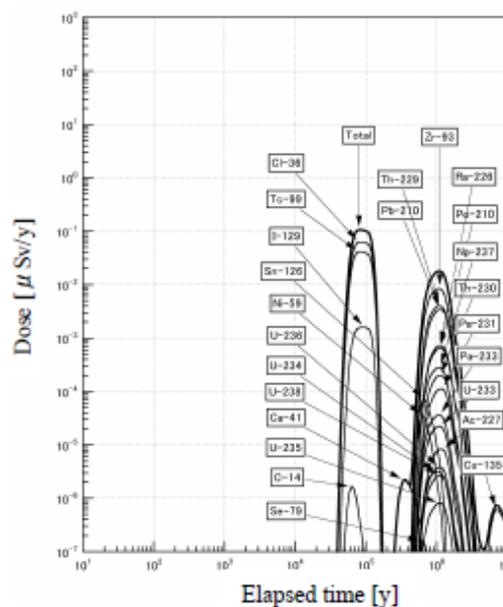
Channel for river water use (Alternative case)



Channel for river water use (Base case)



River bank residential route (Base case)



River bank agricultural route (Base case)

Figure 6.3.5-1 Results of dose assessment for intermediate-depth disposal (100 GBq/t)

## **6.3.6 Safety assessment of geological disposal**

When upper limit of  $\alpha$ -emitting nuclides is 100 GBq/t, the amount of waste intended for geological disposal decreases as described in Section 6.3.4. However, since there is expected to be only a small change in radioactivity of I-129 and C-14, which are the main nuclides that contribute to long-term exposure dose, the dose will be almost the same as that of the evaluation result presented in Chapter 4.

## **6.3.7 Summary**

If the upper limit of  $\alpha$ -emitting nuclides concentration used to classify waste for intermediate-depth disposal is increased to 100 GBq/t, the quantity of suitable waste would also increase. However, an safety assessment has shown that, despite this increase, the dose would remain sufficiently below the target dose caused by general phenomena affecting concrete vault disposal (10  $\mu$ Sv/y) that the safety of intermediate-depth disposal could be ensured.

The amount of waste suitable for geological disposal would decrease. However, the dose would be almost the same as that calculated in Chapter 4, since the amount of radioactivity contributed by the main nuclides present would change by only a small amount.

## **6.4 Effect of return method on waste returned from overseas**

### **6.4.1 Summary of returned radioactive waste**

#### **6.4.1.1 Summary of waste**

The electric power utilities have entrusted the reprocessing of some spent fuel to BNGS and COGEMA. The amounts of radioactive waste that are returned to Japan following reprocessing (returned radioactive waste) will be equivalent to the amounts of waste that were generated by the electric power utilities.

International seawater transportation from the U.K. and France to Japan is required over a long period to return the waste. To reduce the number of times that waste must be shipped and the size of waste storage facilities that are required, the possibility for reducing the quantity of returned waste has been evaluated.

The classifications of the returned waste are based on the radioactive waste management policies in

the U.K. and France and are described below.

- UK: The waste is returned in a single form, as vitrified high-level waste (single return method).
- France: The method for returning the waste depends on its category (high-level waste, low-level waste [hulls and ends, low-level liquid waste, miscellaneous waste])

At present, the timing of waste return, the return methods and the returned amounts are being coordinated by the electrical power utilities.

### **(1) Waste returned from BNGS**

It is now proposed to substitute the wide variety of wastes that are produced by BNGS with a quantity of vitrified high-level waste that would give an equivalent radiation effect. This substituted waste would be returned to Japan. This approach is termed the single return method. The original wastes and substituted wastes have different nuclide compositions and contain different quantities of nuclides. Consequently, the equivalent quantities of waste are determined by comparing how a person would be affected by ingesting nuclides that had migrated from each waste type during a fixed reference period (Kosako, 2004).

The returned vitrified waste that has been substituted for other reprocessing wastes (according to the single return method) would have the same specifications as that of the vitrified waste that has been returned from BNGS from the beginning. Consequently, it is possible to store and dispose of the substituted waste safely.

All the vitrified waste that is planned to be returned has the same specification and will be packaged in containers with the same characteristics. Additionally, the quantity of waste packages is substantially reduced to about 150 from about 10,500. Consequently, treatment, transport and storage will be optimized.

### **(2) Waste returned from COGEMA**

The liquid waste produced by reprocessing spent fuel is classified into high-level waste, the main components of which are fission products, and low-level waste, which is generated by reprocessing plant operations.

It was initially planned to return the low-level waste in the form of 1,100 containers of bituminized waste solid. However, one of the return methods now being considered is to substitute the bituminized waste with about 28 containers of waste referred to as solidified low-level liquid waste.

The solidified low-level liquid waste is produced by vitrification of the waste fluid produced during washing activities in the reprocessing plant. This vitrified waste is stored in the same kind of package (canister) as other vitrified waste. COGEMA has already stopped operations at the UP2 reprocessing plant and plans to solidify the low-level liquid waste produced while washing part of this facility. COGEMA is presently developing technologies to carry out the vitrification.

The characteristics of the solidified low-level liquid waste, the inventory and an assessment of safety are described in the next section.

According to this newly planned method for returning waste from COGEMA, the vitrified waste will all be stored in identical packages. The quantity of the waste can be reduced to about 28 solidified low-level liquid waste containers, in contrast to the 1,100 containers that would be required for solid bitumen waste. The transport and storage of the waste is thus optimized.

#### **6.4.1.2 Waste treatment and waste packages**

For the cases where waste substitution is proposed, Table 6.4.1.2-1 compares the treatment methods, solidification methods and waste packages to be used for the waste that will be returned.

#### **6.4.1.3 Quantities of waste generated**

Assuming the numbers of waste packages to be returned, the estimated quantities of waste generated are summarized in Table 6.4.1.3-1. The cumulative volume of waste packages produced is estimated to be between about 140,000 m<sup>3</sup> and about 130,000 m<sup>3</sup> (Table 6.4.1.3-2) in the case where there has been volume reduction and solidification of the waste. The calculated quantities of waste packages in this report might change depending on the extent to which the actual operating conditions of the facilities differ from the assumed operating conditions.

Table 6.4.1.2-1 Assumed waste treatment and waste packages (returned waste)

	Waste type	At present for the initially planned method of return			At the time of return by the newly proposed method		
		Treatment method	Package	Solidification method	Treatment method	Package	Solidification method
TRU waste	Miscellaneous solid (BNGS)	Compression	1500L container	Cement solidification	—	—	—
	MEB cladding + barium carbonate slurry solidified in cement (BNGS)	—	500L drum	Cement solidification	—	—	—
	Solidified cement (hulls and ends) (BNGS)	—	500L drum	Cement solidification	—	—	—
	Magnox waste solidified in cement (BNGS)	—	500L drum	Cement solidification	—	—	—
	Centrifuge cake solidified in cement (BNGS)	—	500L drum	Cement solidification	—	—	—
	Solidified material storage container (COGEMA)	Compression	Canister	—	Compression	Canister	—
	Bituminized waste (COGEMA)	Chemical treatment	230L drum	Bitumen solidification	—	—	—
	Solidified low-level liquid waste (COGEMA)	—	—	—	—	Canister	Vitrification
High-level radioactive waste	Vitrified waste (BNGS)	Evaporation and concentration	Canister	Vitrification	Evaporation and concentration	Canister	Vitrification
	Vitrified waste (COGEMA)	Evaporation and concentration	Canister	Vitrification	Evaporation and concentration	Canister	Vitrification

Table 6.4.1.3-1 Estimated quantities of waste generated (returned waste including high-level waste)

Waste type		Total quantity generated			
		At present for the initially planned method of return		At the time of return by the newly proposed method	
		(number)	(m <sup>3</sup> )	(number)	(m <sup>3</sup> )
TRU waste	Miscellaneous solid (BNGS)	3,300	4,950	—	—
	Magnox miscellaneous solid (BNGS)	2,700	4,050	—	—
	MEB cladding + barium carbonate slurry solidified in cement (BNGS)	250	140	—	—
	Solidified cement (hulls and ends: PWR) (BNGS)	1,035	580	—	—
	Solidified cement (hulls and ends: BWR) (BNGS)	1,035	580	—	—
	Magnox waste solidified in cement (BNGS)	900	504	—	—
	Centrifuge cake solidified in cement: PWR (BNGS)	630	353	—	—
	Centrifuge cake solidified in cement: BWR (BNGS)	650	364	—	—
	Solidified material storage container (COGEMA)	3,600	684	3,600	684
	Bituminized waste (COGEMA)	1,100	253	—	—
	Solidified low-level liquid waste (COGEMA)	—	—	28	5
	Subtotal		15,200	12,457	3,628
High-level waste	High-level glass solid (BNGS)	850	160	1,000	190
	High-level glass solid (COGEMA)	1,350	260	1,350	260
	Subtotal		2,200	420	2,350
Total		17,400	12,877	5,978	1,139

Table 6.4.1.3-2 Comparison between estimated quantities of TRU waste generated

Waste type	Total quantity generated (m <sup>3</sup> )			
	At present for the initially planned method of return		At the time of return by the newly proposed method	
Private sector reprocessing and operating waste	50,450	97,635	50,450	97,635
Private sector MOX operating waste	922		922	
Private sector waste reprocessing and dismantling waste	44,259		44,259	
Private sector MOX dismantling waste	2,004		2,004	
JAEA reprocessing and operating waste	18,286	30,182	18,286	30,182
JAEA MOX operating waste	1,307		1,307	
JAEA reprocessing and dismantling waste	8,544		8,544	
JAEA MOX dismantling waste	2,045		2,045	
Returned low-level waste (BNGS)	11,520	12,457	0	689
Returned low-level waste (COGEMA)	937		689	
Total	140,274	140,274	128,506	128,506

## 6.4.1.4 Properties of waste packages

### (1) Concentrations of radionuclides in waste

The concentrations of the radionuclides in the wastes that are actually returned to Japan are calculated to be the same as the concentrations of radionuclides in the wastes that could be returned.

The concentrations of radionuclides in the different types of wastes are shown in Table 6.4.1.4-1.

Table 6.4.1.4-1 Concentrations of radionuclides in each type of waste (returned waste) (1/2)

(Unit: Bq/t)

Nuclide	At present for the initially planned method of return							
	BNGS							
	Magnox: miscellaneous solids	Magnox: solidified in cement	MEB cladding+barium carbonate slurry solidified in cement	Centrifuge cake slurry solidified in cement (PWR)	Centrifuge cake slurry solidified in cement (BWR)	Solidified cement (hulls and ends: PWR)	Solidified cement (hulls and ends: BWR)	Miscellaneous solids (BNGS)
H-3	—	—	$2.0 \times 10^8$	$3.8 \times 10^9$	$2.4 \times 10^9$	$4.4 \times 10^{12}$	$2.8 \times 10^{12}$	$6.3 \times 10^6$
C-14	—	—	$8.8 \times 10^{10}$	$3.3 \times 10^6$	$2.1 \times 10^6$	$1.4 \times 10^{10}$	$6.7 \times 10^9$	$3.1 \times 10^4$
Cl-36	—	—	$1.3 \times 10^2$	$3.6 \times 10^4$	$2.3 \times 10^4$	$6.8 \times 10^4$	$4.5 \times 10^4$	—
Co-60	—	$2.3 \times 10^{12}$	$1.7 \times 10^{11}$	—	—	$2.5 \times 10^{13}$	$6.1 \times 10^{12}$	$1.3 \times 10^6$
Ni-59	—	—	—	—	—	$1.3 \times 10^{11}$	$1.8 \times 10^{10}$	—
Ni-63	—	—	—	—	—	$1.7 \times 10^{13}$	$2.3 \times 10^{12}$	—
Sr-90	—	$3.8 \times 10^{12}$	$3.0 \times 10^9$	$8.4 \times 10^{11}$	$5.4 \times 10^{11}$	$1.1 \times 10^{12}$	$7.2 \times 10^{11}$	—
Zr-93	—	—	—	—	—	$5.5 \times 10^9$	$4.3 \times 10^9$	—
Nb-94	—	—	—	—	—	$5.8 \times 10^{10}$	$1.1 \times 10^9$	—
Tc-99	—	—	$8.5 \times 10^5$	$2.1 \times 10^8$	$1.3 \times 10^8$	$2.7 \times 10^8$	$1.7 \times 10^8$	—
I-129	—	—	$3.2 \times 10^8$	$3.9 \times 10^5$	$2.4 \times 10^5$	$4.7 \times 10^5$	$3.0 \times 10^5$	$3.1 \times 10^4$
Cs-135	—	—	$1.9 \times 10^4$	$5.3 \times 10^6$	$3.4 \times 10^6$	$3.6 \times 10^7$	$2.7 \times 10^7$	—
Cs-137	—	$5.8 \times 10^{12}$	$4.3 \times 10^9$	$7.2 \times 10^{12}$	$6.8 \times 10^{12}$	$7.7 \times 10^{12}$	$5.8 \times 10^{12}$	—
Pu-241	—	$6.3 \times 10^{12}$	$4.2 \times 10^9$	$7.6 \times 10^{12}$	$5.4 \times 10^{12}$	$1.3 \times 10^{12}$	$7.1 \times 10^{11}$	—
By total	$3.0 \times 10^8$	$6.2 \times 10^{13}$	$2.8 \times 10^{11}$	$3.5 \times 10^{13}$	$3.1 \times 10^{13}$	$9.7 \times 10^{13}$	$4.1 \times 10^{13}$	$1.0 \times 10^8$
U-234	—	—	$6.2 \times 10^4$	$2.0 \times 10^7$	$1.4 \times 10^7$	$4.6 \times 10^7$	$3.4 \times 10^7$	—
U-235	—	—	$1.0 \times 10^3$	$2.9 \times 10^5$	$2.6 \times 10^5$	$1.2 \times 10^6$	$9.9 \times 10^5$	—
U-236	—	—	$1.4 \times 10^4$	$3.9 \times 10^6$	$2.5 \times 10^6$	$5.7 \times 10^6$	$3.7 \times 10^6$	—
U-238	—	—	$1.7 \times 10^4$	$4.6 \times 10^6$	$4.5 \times 10^6$	$1.0 \times 10^7$	$1.0 \times 10^7$	—
Np-237	—	—	$1.4 \times 10^4$	$3.8 \times 10^6$	$1.9 \times 10^6$	$4.3 \times 10^6$	$2.2 \times 10^6$	—
Pu-238	—	$3.5 \times 10^{10}$	$1.2 \times 10^8$	$2.1 \times 10^{11}$	$9.8 \times 10^{10}$	$3.0 \times 10^{10}$	$1.2 \times 10^{10}$	—
Pu-239	—	$5.0 \times 10^{10}$	$1.8 \times 10^7$	$3.1 \times 10^{10}$	$3.0 \times 10^{10}$	$8.4 \times 10^9$	$6.4 \times 10^9$	—
Pu-240	—	$7.4 \times 10^{10}$	$2.8 \times 10^7$	$4.9 \times 10^{10}$	$4.7 \times 10^{10}$	$9.0 \times 10^9$	$6.8 \times 10^9$	—
Pu-242	—	—	—	$1.6 \times 10^8$	$1.0 \times 10^8$	—	—	—
Am-241	—	$3.7 \times 10^{10}$	$9.5 \times 10^7$	$2.6 \times 10^{10}$	$1.5 \times 10^{10}$	$2.8 \times 10^{10}$	$1.6 \times 10^{10}$	—
Am-243	—	—	$7.8 \times 10^5$	$2.2 \times 10^8$	$7.7 \times 10^7$	$1.9 \times 10^8$	$7.2 \times 10^7$	—
Cm-244	—	$5.5 \times 10^9$	$5.9 \times 10^7$	$1.7 \times 10^{10}$	$4.2 \times 10^9$	$1.4 \times 10^{10}$	$3.8 \times 10^9$	—
$\alpha$ total	$1.1 \times 10^7$	$2.6 \times 10^{11}$	$3.2 \times 10^8$	$3.3 \times 10^{11}$	$2.0 \times 10^{11}$	$9.1 \times 10^{10}$	$4.5 \times 10^{10}$	$3.8 \times 10^6$

Table 6.4.1.4-1 Concentrations of radionuclides in each type of waste (returned waste) (2/2)

(Unit: Bq/t)

Nuclide	At present for the initially planned method of return		At the time of return by the newly proposed method	
	COGEMA			
	Solidified material storage container	Bituminized waste	Solidified material storage container	Solidified low-level liquid waste
H-3	$2.1 \times 10^{13}$	$1.8 \times 10^{10}$	$2.1 \times 10^{13}$	$3.2 \times 10^{11}$
C-14	$2.0 \times 10^{10}$	—	$2.0 \times 10^{10}$	—
Cl-36	—	—	—	—
Co-60	$1.1 \times 10^{14}$	$1.8 \times 10^{11}$	$1.1 \times 10^{14}$	$3.2 \times 10^{12}$
Ni-59	—	—	—	—
Ni-63	$3.9 \times 10^{13}$	—	$3.9 \times 10^{13}$	—
Se-79	$7.9 \times 10^7$	—	$7.9 \times 10^7$	—
Sr-90	$2.0 \times 10^{13}$	$1.2 \times 10^{12}$	$2.0 \times 10^{13}$	$2.1 \times 10^{13}$
Zr-93	$1.2 \times 10^{10}$	—	$1.2 \times 10^{10}$	—
Nb-94	—	—	—	—
Mo-93	—	—	—	—
Tc-99	$3.3 \times 10^9$	$1.8 \times 10^8$	$3.3 \times 10^9$	$3.2 \times 10^9$
I-129	$7.6 \times 10^6$	$1.8 \times 10^8$	$7.6 \times 10^6$	$3.2 \times 10^9$
Cs-135	$1.0 \times 10^8$	—	$1.0 \times 10^8$	—
Cs-137	$2.1 \times 10^{13}$	$2.4 \times 10^{12}$	$2.1 \times 10^{13}$	$4.3 \times 10^{13}$
Pu-241	$1.4 \times 10^{13}$	—	$1.4 \times 10^{13}$	—
Am-242m	—	—	—	—
$\beta\gamma$ total	$3.6 \times 10^{14}$	$2.1 \times 10^{13}$	$3.6 \times 10^{14}$	$3.7 \times 10^{14}$
U-234	—	—	—	—
U-235	—	$2.6 \times 10^6$	—	$4.6 \times 10^7$
U-236	—	—	—	—
U-238	—	$2.6 \times 10^7$	—	$4.6 \times 10^8$
Np-237	$1.0 \times 10^7$	—	$1.0 \times 10^7$	—
Pu-238	$6.7 \times 10^{11}$	$1.3 \times 10^{11}$	$6.7 \times 10^{11}$	$2.3 \times 10^{12}$
Pu-239	$4.3 \times 10^{10}$	$1.5 \times 10^{10}$	$4.3 \times 10^{10}$	$2.6 \times 10^{11}$
Pu-240	$7.4 \times 10^{10}$	$2.0 \times 10^{10}$	$7.4 \times 10^{10}$	$3.6 \times 10^{11}$
Pu-242	$4.1 \times 10^8$	—	$4.1 \times 10^8$	—
Am-241	$2.5 \times 10^{10}$	$3.4 \times 10^{11}$	$2.5 \times 10^{10}$	$6.1 \times 10^{12}$
Am-243	$2.5 \times 10^{10}$	—	$2.5 \times 10^{10}$	—
Cm-244	$1.4 \times 10^{11}$	$1.6 \times 10^{10}$	$1.4 \times 10^{11}$	$2.9 \times 10^{11}$
$\alpha$ total	$9.7 \times 10^{11}$	$5.2 \times 10^{11}$	$9.7 \times 10^{11}$	$9.3 \times 10^{12}$

**(2) Contents of waste**

Here, the radioactive concentrations in the TRU waste are adjusted, depending on the waste volume reduction method proposed for waste to be returned from BNGS and the proposed substitution of solidified low-level liquid waste for bituminized waste to be returned by COGEMA.

The contents of the solidified low-level liquid waste from cleaning fluids used in the facility are described in Section 6.4.1.1. Additionally, all the solidified waste materials are in the form of glass. Here, the quantities of the waste constituents are calculated by considering the treatment method and filling ratio. The variations in the contents of nitrate and organic material are shown in Table 6.4.1.4-2 and Table 6.4.1.4-3. The waste to be returned by the newly proposed methods would contain no nitrate or organic material.

Table 6.4.1.4-2 Amount of nitrate in the waste (returned waste)

Waste type	Disposal classification	At present for the initially planned method of return			At the time of return by the newly proposed method		
		Quantity generated (Number)	Quantity of contents		Quantity generated (Number)	Quantity of contents	
			(kg/number)	(t)		(kg/number)	(t)
MEB cladding + barium carbonate slurry solidified in cement	Geological disposal	250	3	1	—	—	—
Bituminized waste (COGEMA)		1,100	40	44	—	—	—
Solidified low-level liquid waste (COGEMA)		—	—	—	28	0	0
Total		1,350	—	45	28	0	0

Table 6.4.1.4-3 Amount of organic material in the waste (returned waste)

Waste type	Disposal classification	Organic material type	At present for the initially planned method of return			At the time of return by the newly proposed method		
			Quantity generated (Number)	Quantity of contents		Quantity generated (Number)	Quantity of contents	
				(kg/number)	(t)		(kg/number)	(t)
Bituminized waste (COGEMA)	Geological disposal	Bitumen	1,100	140	154	—	—	—
Solidified low-level liquid waste (COGEMA)		—	—	—	—	0	0	0
Total		—	1,100	140	154	0	0	0

### (3) Heat generation of waste

In the solidified low-level liquid waste proposed to be returned from COGEMA, the radioactive concentration in waste package is larger than that in bituminized waste. Since the heat generation rate of the waste depends on the radioactivity present, the heat generation rate is calculated from the radioactivity concentrations shown in (1). The rates of heat generation of the wastes are shown in Table 6.4.1.4-4.

Table 6.4.1.4-4 Heat generation rate of waste (returned waste)

(Unit: W/each)

Waste type	At present for the initially planned method of return	At the time of return by the newly proposed method
Bituminized waste (COGEMA)	0.3	—
Solidified low-level liquid waste (COGEMA)	—	$\leq 90$

### 6.4.1.5 Waste classification

Based on the concept in Section 2.5.1, the waste at the time of return is classified into that suitable for concrete vault disposal, intermediate depth disposal and geological disposal.

The results of the classification are shown in Tables 6.4.1.5-1 and 6.4.1.5-2. The waste that is suitable for concrete vault disposal is decreased by about 9000 m<sup>3</sup> to about 80,000 m<sup>3</sup>. The waste suitable for geological disposal is decreased by about 3,000 m<sup>3</sup>, to about 24,000 m<sup>3</sup> or about 10,000 m<sup>3</sup> in the cases where the  $\alpha$ -emitting nuclide concentrations used for classification are 1 GBq/t or 100 GBq/t respectively. Since no returned low-level waste is considered suitable for intermediate-depth disposal, there is no change in the quantity of waste that is suitable for such disposal.

The radioactivity in each disposal category at the time of implementing the proposed return methods are shown in Table 6.4.1.5-3. These results show that, although the quantities of waste are reduced by the proposed methods, the contribution of radioactivity by the substituted waste is large. Hence, the overall amounts of radioactivity are unchanged.

Table 6.4.1.5-1 Estimated amounts of TRU waste generated in each disposal category  
(upper limit of  $\alpha$ -emitting nuclide concentration for intermediate-depth disposal: 1 GBq/t)

Waste type	Total quantity generated (m <sup>3</sup> )							
	At present for the initially planned method of return				At the time of return by the newly proposed method			
	Concrete vault disposal	Intermediate -depth disposal	Geological disposal	Total	Concrete vault disposal	Disposal at intermediate depth	Geological disposal	Total
Private sector reprocessing / MOX operating waste	24,667	13,276	13,429	51,372	24,667	13,276	13,429	51,372
Private sector reprocessing / MOX dismantling waste	37,138	8,224	902	46,264	37,138	8,224	902	46,264
JAEA reprocessing / MOX operation waste	10,388	3,424	5,781	19,593	10,388	3,424	5,781	19,593
JAEA reprocessing / MOX dismantling waste	7,238	281	3,071	10,589	7,238	281	3,071	10,589
Returned low-level waste (BNGS)	9,000	0	2,520	11,520	0	0	0	0
Returned low-level waste (COGEMA)	0	0	937	937	0	0	689	689
Total	88,431	25,205	26,640	140,274	79,431	25,205	23,872	128,506

Table 6.4.1.5-2 Estimated quantities of TRU waste generated in each disposal category  
(upper limit of  $\alpha$ -emitting nuclide concentration for intermediate-depth disposal: 100 GBq/t)

Waste type	Total quantity generated (m <sup>3</sup> )							
	At present for the initially planned method of return				At the time of return by the newly proposed method			
	Concrete vault disposal	Intermediate -depth disposal	Geological disposal	Total	Concrete vault disposal	Intermediate -depth	Geological disposal	Total
Private sector reprocessing / MOX operating waste	24,667	20,540	6,165	51,372	24,667	20,540	6,165	51,372
Private sector reprocessing / MOX dismantling waste	37,138	8,519	606	46,264	37,138	8,519	606	46,264
JAEA reprocessing / MOX operation waste	10,388	8,007	1,198	19,593	10,388	8,007	1,198	19,593
JAEA reprocessing / MOX dismantling waste	7,238	2,229	1,122	10,589	7,238	2,229	1,122	10,589
Returned low-level waste (BNGS)	9,000	0	2,520	11,520	0	0	0	0
Returned low-level waste (COGEMA)	0	0	937	937	0	0	689	689
Total	88,431	39,295	12,548	140,274	79,431	39,295	9,780	128,506

Table 6.4.1.5-3 Amount of radioactive substances in each disposal classification (Bq)

Nuclide	At s present for the initially planned method of return			At the time of return by the newly proposed method		
	Concrete vault disposal	Geological disposal		Concrete vault disposal	Geological disposal	
		$\alpha$ : 1GBq/t	$\alpha$ :100GBq/t		$\alpha$ : 1GBq/t	$\alpha$ :100GBq/t
H-3	$7.4 \times 10^{13}$	$7.1 \times 10^{17}$	$7.1 \times 10^{17}$	$7.4 \times 10^{13}$	$7.0 \times 10^{17}$	$7.0 \times 10^{17}$
C-14	$1.3 \times 10^{12}$	$5.7 \times 10^{14}$	$5.6 \times 10^{14}$	$1.3 \times 10^{12}$	$5.1 \times 10^{14}$	$5.0 \times 10^{14}$
Cl-36	$7.5 \times 10^8$	$8.8 \times 10^{12}$	$8.6 \times 10^{12}$	$7.5 \times 10^8$	$8.8 \times 10^{12}$	$8.6 \times 10^{12}$
Co-60	$5.9 \times 10^{12}$	$1.6 \times 10^{18}$	$1.6 \times 10^{18}$	$5.9 \times 10^{12}$	$1.5 \times 10^{18}$	$1.5 \times 10^{18}$
Ni-59	$1.8 \times 10^{10}$	$7.2 \times 10^{15}$	$7.2 \times 10^{15}$	$1.8 \times 10^{10}$	$7.0 \times 10^{15}$	$7.0 \times 10^{15}$
Ni-63	$1.6 \times 10^{12}$	$1.2 \times 10^{18}$	$1.2 \times 10^{18}$	$1.6 \times 10^{12}$	$1.2 \times 10^{18}$	$1.2 \times 10^{18}$
Se-79	$1.1 \times 10^9$	$2.7 \times 10^{12}$	$2.7 \times 10^{12}$	$1.1 \times 10^9$	$2.7 \times 10^{12}$	$2.7 \times 10^{12}$
Sr-90	$6.8 \times 10^{14}$	$5.9 \times 10^{17}$	$5.4 \times 10^{17}$	$6.8 \times 10^{14}$	$5.8 \times 10^{17}$	$5.3 \times 10^{17}$
Zr-93	$3.2 \times 10^{10}$	$3.1 \times 10^{14}$	$3.0 \times 10^{14}$	$3.2 \times 10^{10}$	$2.9 \times 10^{14}$	$2.9 \times 10^{14}$
Nb-94	$4.0 \times 10^8$	$2.6 \times 10^{15}$	$2.6 \times 10^{15}$	$4.0 \times 10^8$	$2.5 \times 10^{15}$	$2.5 \times 10^{15}$
Mo-93	$7.5 \times 10^7$	$5.2 \times 10^{13}$	$5.2 \times 10^{13}$	$7.5 \times 10^7$	$5.2 \times 10^{13}$	$5.2 \times 10^{13}$
Tc-99	$2.2 \times 10^{11}$	$6.6 \times 10^{14}$	$6.4 \times 10^{14}$	$2.2 \times 10^{11}$	$6.6 \times 10^{14}$	$6.4 \times 10^{14}$
Pd-107	$2.8 \times 10^8$	$5.5 \times 10^{11}$	$5.4 \times 10^{11}$	$2.8 \times 10^8$	$5.5 \times 10^{11}$	$5.4 \times 10^{11}$
Ag-108m	$1.7 \times 10^8$	$2.1 \times 10^{12}$	$2.1 \times 10^{12}$	$1.7 \times 10^8$	$2.1 \times 10^{12}$	$2.1 \times 10^{12}$
Sn-126	$1.3 \times 10^{10}$	$6.1 \times 10^{12}$	$5.0 \times 10^{12}$	$1.3 \times 10^{10}$	$6.1 \times 10^{12}$	$5.0 \times 10^{12}$
I-129	$6.7 \times 10^9$	$5.2 \times 10^{13}$	$5.1 \times 10^{13}$	$6.7 \times 10^9$	$5.2 \times 10^{13}$	$5.1 \times 10^{13}$
Cs-135	$4.3 \times 10^9$	$3.4 \times 10^{12}$	$3.1 \times 10^{12}$	$4.3 \times 10^9$	$3.3 \times 10^{12}$	$3.0 \times 10^{12}$
Cs-137	$9.5 \times 10^{14}$	$7.6 \times 10^{17}$	$7.0 \times 10^{17}$	$9.5 \times 10^{14}$	$7.3 \times 10^{17}$	$6.6 \times 10^{17}$
Pu-241	$2.9 \times 10^{14}$	$5.7 \times 10^{17}$	$5.6 \times 10^{17}$	$2.9 \times 10^{14}$	$5.5 \times 10^{17}$	$5.4 \times 10^{17}$
Am-242m	$2.1 \times 10^{11}$	$6.2 \times 10^{13}$	$4.2 \times 10^{13}$	$2.1 \times 10^{11}$	$6.1 \times 10^{13}$	$4.1 \times 10^{13}$
$\beta\gamma$ total	$5.0 \times 10^{15}$	$1.7 \times 10^{19}$	$1.7 \times 10^{19}$	$5.0 \times 10^{15}$	$1.6 \times 10^{19}$	$1.6 \times 10^{19}$
U-233	$2.6 \times 10^4$	$4.7 \times 10^9$	$4.7 \times 10^9$	$2.6 \times 10^4$	$4.7 \times 10^9$	$4.7 \times 10^9$
U-234	$4.9 \times 10^8$	$1.7 \times 10^{12}$	$1.6 \times 10^{12}$	$4.9 \times 10^8$	$1.5 \times 10^{12}$	$1.5 \times 10^{12}$
U-235	$4.3 \times 10^7$	$1.1 \times 10^{11}$	$1.1 \times 10^{11}$	$4.3 \times 10^7$	$1.1 \times 10^{11}$	$1.1 \times 10^{11}$
U-236	$7.5 \times 10^8$	$1.2 \times 10^{12}$	$1.2 \times 10^{12}$	$7.5 \times 10^8$	$1.1 \times 10^{12}$	$1.1 \times 10^{12}$
U-238	$6.2 \times 10^8$	$1.1 \times 10^{12}$	$1.1 \times 10^{12}$	$6.2 \times 10^8$	$1.0 \times 10^{12}$	$1.0 \times 10^{12}$
Np-237	$5.3 \times 10^9$	$2.4 \times 10^{12}$	$2.3 \times 10^{12}$	$5.3 \times 10^9$	$2.4 \times 10^{12}$	$2.3 \times 10^{12}$
Pu-238	$7.1 \times 10^{12}$	$1.7 \times 10^{16}$	$1.7 \times 10^{16}$	$7.1 \times 10^{12}$	$1.7 \times 10^{16}$	$1.7 \times 10^{16}$
Pu-239	$6.3 \times 10^{11}$	$2.4 \times 10^{15}$	$2.4 \times 10^{15}$	$6.3 \times 10^{11}$	$2.3 \times 10^{15}$	$2.2 \times 10^{15}$
Pu-240	$9.9 \times 10^{11}$	$3.2 \times 10^{15}$	$3.2 \times 10^{15}$	$9.9 \times 10^{11}$	$3.1 \times 10^{15}$	$3.0 \times 10^{15}$
Pu-242	$4.1 \times 10^9$	$1.1 \times 10^{13}$	$1.1 \times 10^{13}$	$4.1 \times 10^9$	$1.1 \times 10^{13}$	$1.0 \times 10^{13}$
Am-241	$2.4 \times 10^{12}$	$6.1 \times 10^{15}$	$6.1 \times 10^{15}$	$2.4 \times 10^{12}$	$6.0 \times 10^{15}$	$5.8 \times 10^{15}$
Am-243	$5.6 \times 10^{10}$	$2.1 \times 10^{14}$	$2.1 \times 10^{14}$	$5.6 \times 10^{10}$	$2.0 \times 10^{14}$	$2.0 \times 10^{14}$
Cm-244	$5.7 \times 10^{12}$	$9.4 \times 10^{15}$	$9.3 \times 10^{15}$	$5.7 \times 10^{12}$	$9.4 \times 10^{15}$	$9.3 \times 10^{15}$
Cm-245	$7.4 \times 10^8$	$9.8 \times 10^{11}$	$9.5 \times 10^{11}$	$7.4 \times 10^8$	$9.8 \times 10^{11}$	$9.5 \times 10^{11}$
$\alpha$ total	$1.7 \times 10^{13}$	$3.9 \times 10^{16}$	$3.9 \times 10^{16}$	$1.7 \times 10^{13}$	$3.8 \times 10^{16}$	$3.7 \times 10^{16}$

## 6.4.2 Disposal method

Features of the solidified low-level liquid wastes are that they are vitrified, they do not contain nitrates and the quantities are extremely small ( $5 \text{ m}^3$ ). Although the heat generation rate in Table 6.4.1.4-4 is relatively high, this is a conservative value. In reality, the value is actually predicted to be quite low. Moreover, it is expected that I-129 will be volatilized during the vitrification process.

As described previously, since the solidified low-level liquid waste is vitrified in order to ensure the long-term isolation, a high-pH environment should be prevented, as in the case of a disposal facility for vitrified high-level waste. In order to maintain a low hydraulic conductivity in the region of the waste and to reduce the physical effects of the host rock, bentonite should be emplaced around the waste.

The number of waste canisters to be disposed of under these conditions is 28 and is reduced to 7 if the 4 canisters are emplaced in each waste package. If the radioactive concentration and the heat generation rate in the actual waste and the disposal site conditions are clarified, these wastes can be appropriately disposed of using the same disposal technology as that for other TRU waste and HLW.

## 6.4.3 Safety assessment of concrete vault disposal

If the proposed return method is implemented, miscellaneous waste produced by BNGS for concrete vault disposal is decreased by  $9,000 \text{ m}^3$  and the radioactivity are also decreased. However, the radioactivity in this waste is small relative to those in the total waste. Consequently, the total radioactivity shows almost no decrease as shown in Table 6.4.1.5-3. Hence, the dose from concrete vault disposal is almost the same as the evaluation result in Chapter 5.

## 6.4.4 Safety assessment of geological disposal

If the proposed methods are implemented, COGEMA will return  $5 \text{ m}^3$  of solidified low-level liquid waste rather than  $253 \text{ m}^3$  of bituminized waste. However, since the radioactivity are almost the same in the two types of waste, the dose from geological disposal would be almost the same as the assessment result in Chapter 4.

According to the single return method, the quantity of cement solidified wastes of MEB cladding and barium-silica carbonate cement, hulls and ends, magnox and centrifuge cake to be returned by BNGS for geological disposal will be decreased from  $2,520 \text{ m}^3$ . However, since the decrease in the

radioactivity will be trivial, the dose arising from geological disposal will be almost the same as the evaluation result in Chapter 4.

While the quantity of TRU waste for geological disposal will be decreased, the quantity of HLW will be increased. However, since this latter increase will be equal to only about 150 waste canisters, in contrast to a total number of about 40,000, the effect on dose will not be changed.

### 6.4.5 Summary

If the methods proposed for returned waste from the U.K. and France are selected, the requirement for international transport and storage will be decreased, as shown in Figure 6.4.5-1. Additionally, since the quantity of the waste will be decreased, these methods of return will allow the design of the facility to be optimized. The safety of disposal can also be demonstrated more readily.

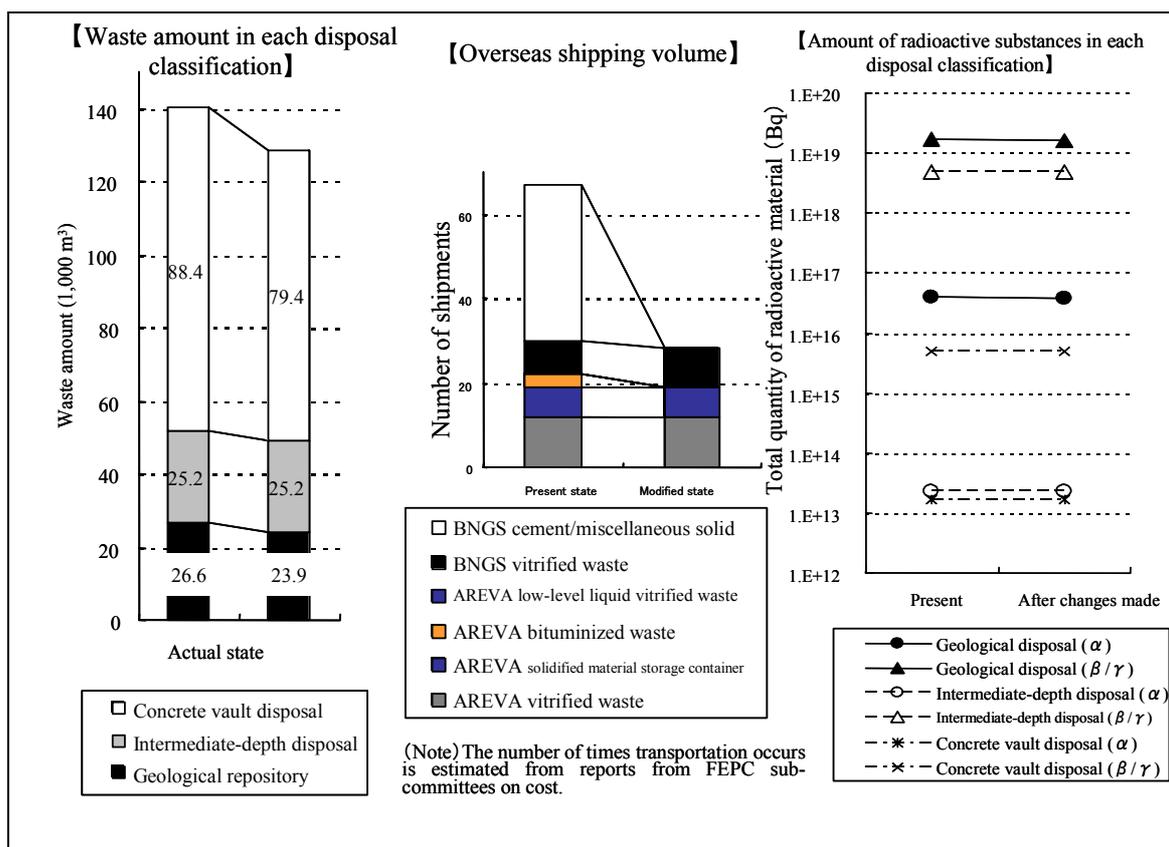


Figure 6.4.5-1 Optimization depending on the selected method for returning waste

#### **6.4.6 Future issues**

Waste will be returned to Japan by BNGS and AREVA. Based on current knowledge, this report has evaluated the optimization and safety of the disposal of this returned waste.

The solidified low-level liquid waste proposed to be returned by AREVA can be disposed of appropriately by the method described in Section 6.4.2.

However, optimization of the disposal method should be considered, taking into account technological developments by AREVA, the heat generation rate, radioactivity concentration and resistance properties for high pH of the solidified low-level liquid waste which is actually produced.

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