JAERI-Research
94-028

SAFETY AND SENSITIVITY ANALYSES OF A GENERIC GEOLOGIC DISPOSAL SYSTEM FOR HIGH-LEVEL RADIOACTIVE WASTE

November 1994

Hideo KIMURA, Tomoyuki TAKAHASHI, Shigeki SHIMA
and Hideo MATSUZURU

日本原子力研究所
Japan Atomic Energy Research Institute

本レポートは，日本原子力研究所が不定期に公刊している研究報告書です。

入手の問合わせは，日本原子力研究所技術情報部情報資料課（〒319-11 茨城県那珂郡東海村）あて，お申し越しください。なお，このほかに財団法人原子力弘済会資料センター（〒319-11 茨城県那珂郡東海村日本原子力研究所内）で複写による実費頒布をおこなっております。

This report is issued irregularly.

Inquiries about availability of the reports should be addressed to Information Division, Department of Technical Information, Japan Atomic Energy Research Institute, Tokai-mura, Naka-gun, Ibaraki-ken 319-11, Japan.

編集兼発行　日本原子力研究所
印　　刷　いばらき印刷㈱

JAERI-Research 94-028

Safety and Sensitivity Analyses of a Generic Geologic Disposal System for High-level Radioactive Waste

Hideo KIMURA, Tomoyuki TAKAHASHI, Shigeki SHIMA
and Hideo MATSUZURU

Department of Environmental Safety Research
Tokai Research Establishment
Japan Atomic Energy Research Institute
Tokai-mura, Naka-gun, Ibaraki-ken

(Received October 3, 1994)

This report describes safety and sensitivity analyses of a generic geologic disposal system for HLW, using a GSRW code and an automated sensitivity analysis methodology based on the Differential Algebra. An exposure scenario considered here is based on a normal evolution scenario which excludes events attributable to probabilistic alterations in the environment.

The results of sensitivity analyses indicate that parameters related to a homogeneous rock surrounding a disposal facility have higher sensitivities to the output analyzed here than those of a fractured zone and engineered barriers. The sensitivity analysis methodology provides technical information which might be bases for the optimization of design of the disposal facility.

Safety analyses were performed on the reference disposal system which involve HLW in amounts corresponding to 16,000 MTU of spent fuels. The individual dose equivalent due to the exposure pathway ingesting drinking water was calculated using both the conservative and realistic values of geochemical parameters. In both cases, the commmitted dose equivalent evaluated here is the order of 10^{-7}Sv, and thus geologic disposal of HLW may be feasible if the disposal conditions assumed here remain unchanged throughout the periods assessed here.

Keywords: Radioactive Waste Disposal, High-level Radioactive Waste, Safety Assessment, Normal Evolution Scenario, Source Term, Geologic Media, Radionuclide Migration, Solubility, Groundwater, Geochemistry, Hydrology, Dose Equivalent, Automated Sensitivity Analysis

高レベル放射性廃棄物地層処分の一般的安全評価及び感度解析

日本原子力研究所東海研究所環境安全研究部
木村　英雄・高橋　知之・島　　茂樹・松鶴　秀夫

（1994年10月3日受理）

高レベル放射性廃棄物地層処分の一般的安全評価及び感度解析をＧＳＲＷコード及びＤＡ手法に基づく自動感度解析手法を用いて行った。本手法で用いられた評価シナリオは，処分システムの性能が確率論的事象によって影響を受けないと仮定した通常シナリオに基づいている。感度解析の結果、処分場周辺の均質岩体に関するパラメータは，亀裂帯及び人工バリアに関するパラメータよりも、高い感度を有することがわかった。またこの感度解析手法により、処分場施設設計の最適化の基礎となる技術的な知見を得ることが出来る。

使用済核燃料16,000ＭＴＵに相当する高レベル廃棄物が埋設された概念処分場について安全評価を行った。評価は保守的及び現実的な地球化学パラメータを用いて行い、飲料水摂取による個人被曝線量当量を算出した。その結果個人線量当量は，両ケースとも10^{-7}Svのオーダーであり、本評価で想定した処分条件が評価期間内で変化しないと仮定すれば、高レベル廃棄物の地層処分は可能であることがわかった。

東海研究所：〒319－11　茨城県那珂郡東海村白方字白根２－４

目　　次

Contents

1. INTRODUCTION

The safety of radioactive waste disposal is one of the essential subjects of all countries engaged in nuclear energy production. In particular, high-level radioactive waste (HLW) which contains a significant amount of long-lived radionuclides will be disposed of into deep geologic disposal system consisting of a series of engineered and natural barriers, so as to isolate it effectively and to ensure radiation protection of man and his environment even in the far future.

The Japan Atomic Energy Research Institute (JAERI) is developing a safety assessment methodology for geologic disposal of HLW, in accordance with "The Long-Term Yearly Program for Safety Study on High-Level Radioactive Wastes (1991 - 1995)" established by the Nuclear Safety Commission of Japan. Objectives of the development of the methodology are;

- to demonstrate the feasibility of HLW disposal,
- to provide information on which to establish safety criteria,
- to extract the important pathways and parameters,
- to identify the major uncertainties involved in the assessment,
- to assign a priority of future research, and
- to establish the methodology for licensing procedures.

We have developed an integrated computer code system GSRW (Generic Safety assessment code for geologic disposal of Radioactive Waste)[1], that is based on a normal evolution scenario, as a deterministic safety assessment methodology. The GSRW, in which modular type of a source term models, geosphere models and a biosphere model are interlinked, intends to evaluate radiological consequences to an individual or a population due to radionuclides released from geologic radioactive waste repositories in a deep stable rock mass. A series of barriers provided with a disposal system will prevent the release of radionuclides from a repository, and retard the subsequent transportation of them in the geosphere.

The safety assessment of the repository is based on the analysis of the transport of radionuclides by groundwater in and through the repository and the geosphere, and the behavior of radionuclides in the biosphere. In the GSRW, these computations are based on simplified models which incorporate only the major mechanisms and components which will be potentially involved in the disposal system. This simplified and integrated methodology thus developed makes a large number of computer executions and automated multi-parametric sensitivity analyses by using Differential Algebra method more practical, simplifies interpretation of analytical results, and permits safety assessment aiming mainly at the establishment of safety criteria, in particular where a candidate disposal site is not decided yet.

This report summarizes a deterministic safety assessment methodology based on a normal evolution scenario, and involves the description of the scenario assumed here, the structure of the safety assessment methodology developed, and the methodology of an automated sensitivity analysis. The report also summarizes the results of sensitivity analyses and safety assessment for geologic

disposal of HLW using a computer code system GSRW, and also the discussion of a feasibility of the geologic disposal.

The demonstration of the safety of geologic disposal of HLW is a major scientific and engineering target in a waste management community, given that increasingly large amounts of such the waste including spent nuclear fuels are stored on the surface due to the operation of a nuclear fuel cycle over the past two decades. It might be therefore urgent to evaluate, through safety assessment, a feasibility of geologic disposal which is considered as one of the most promising method to isolate the waste safely. This climate is clearly reflected in the current efforts in the safety or performance assessment field. In the past decades, reports on safety analyses or performance assessments on spent nuclear fuels and/or HLW disposal have been published by SKB (Swedish Nuclear Fuel and Waste Management Co., Sweden) [2), (3)], NAGRA (Nationale Genossenschaft fur die Lagerung radioaktiver Abfalle)[4], SKI (Swedish Nuclear Power Inspectorate, Sweden)[5], TVO (Teollisuuden Voima Oy, Finland)[6], PNC (Power Reactor and Nuclear Fuel Development Corporation, Japan)[7], and UPDATING 1990[8]. Swedish and Finnish assessments were performed on so-called reference or potential sites, however the characteristics of such postulated sites used were determined by taking into account of those of a realistic site which was characterized to some extent by extensive site investigations. They used a deterministic approach based on possible transport pathways from a source to the biosphere, that were estimated by groundwater flow analyses using field data thus obtained. In Japan, such a site investigation has not been conducted on a specific candidate site at this moment in order to characterize the potential or reference site for geologic disposal. This situation is reflected to some extent in the assessment of PNC, placing emphasis on the performance evaluation of engineered barriers whose performance does not necessarily depend on site specific geological characteristics.

To conduct the assessment for a postulated disposal system which is not characterized by a specific engineered design and site conditions, it should deal with a wide variety of possible conditions upon which safety of the disposal system is governed. The diversity of possible conditions makes it practically difficult to implement safety analysis, even if it is not theoretically impossible. One of practical measures to overcome this difficulty might be a sensitivity analysis. The automated sensitivity analysis methodology based on the Differential Algebra[9] permits the analysis for inter-linked models involving a number of parameters, giving a multi-parametric sensitivity.

This report deals with safety assessment of a postulated geologic disposal system for HLW, assuming both conservative and reasonably realistic conditions which are defined by published data. Based on the results of sensitivity analysis, the report also indicates the relative importance of both engineered and natural barriers involved in the disposal system considered here, and discusses a generic feasibility of geologic disposal of HLW.

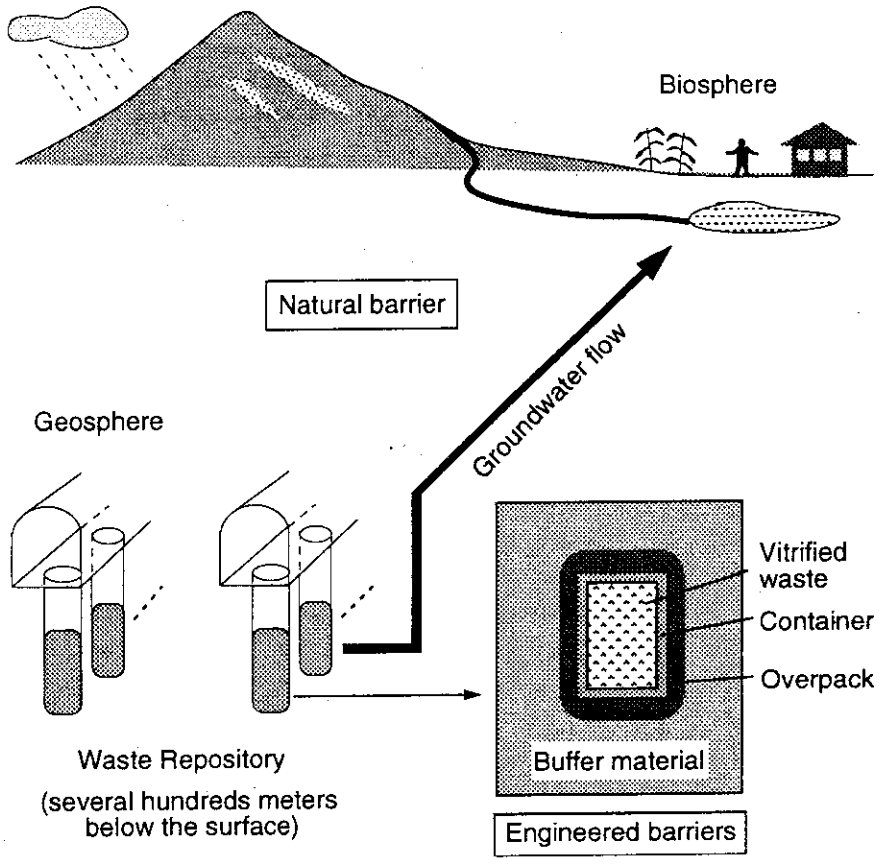

Fig. 1 Multi-barrier concept for the HLW disposal

2. ASSESSMENT SCENARIO

The safety of the disposal system depends on the performance of engineered barriers (a waste matrix, a container, a buffer zone, and concrete structures if necessary) to confine radionuclides in the repository, and the ability of natural barriers to retard and dilute radionuclides during the transport in the geosphere and the biosphere. In the GSRW, the safety of the repository is assumed to be controlled by this multi-barrier concept, as shown in Fig. 1. The concept of the disposal system assumed here consists of;

- the container (overpack made of a carbon steal) acts as a barrier to protect a vitrified waste contained therein from contact with groundwater during a certain initial period of the time after the closure of the repository, and then determines the initiation of radionuclide release,
- the vitrified matrix confines radionuclides, however, once the container failed significantly the matrix will be subjected to a dissolution process by groundwater, which is controlled by the dissolution rate of the matrix and also by the solubility of the elements involved,
- the buffer zone (bentonite), in which radionuclides are transported by diffusion and retention mechanisms, retards the release of radionuclides from the repository,
- the geosphere, where most portions of the radionuclides released from the repository are effectively retained and delayed by various retention mechanisms, retards the transport of the radionuclides

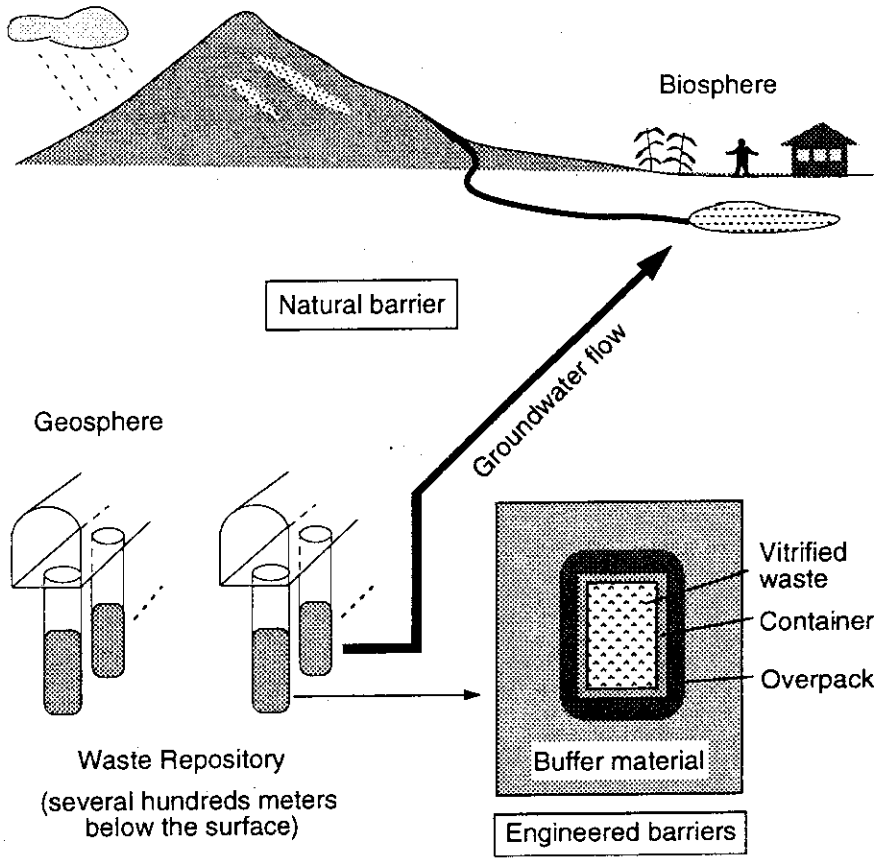

Fig. 1 Multi-barrier concept for the HLW disposal

2. ASSESSMENT SCENARIO

The safety of the disposal system depends on the performance of engineered barriers (a waste matrix, a container, a buffer zone, and concrete structures if necessary) to confine radionuclides in the repository, and the ability of natural barriers to retard and dilute radionuclides during the transport in the geosphere and the biosphere. In the GSRW, the safety of the repository is assumed to be controlled by this multi-barrier concept, as shown in Fig. 1. The concept of the disposal system assumed here consists of;

- the container (overpack made of a carbon steal) acts as a barrier to protect a vitrified waste contained therein from contact with groundwater during a certain initial period of the time after the closure of the repository, and then determines the initiation of radionuclide release,
- the vitrified matrix confines radionuclides, however, once the container failed significantly the matrix will be subjected to a dissolution process by groundwater, which is controlled by the dissolution rate of the matrix and also by the solubility of the elements involved,
- the buffer zone (bentonite), in which radionuclides are transported by diffusion and retention mechanisms, retards the release of radionuclides from the repository,
- the geosphere, where most portions of the radionuclides released from the repository are effectively retained and delayed by various retention mechanisms, retards the transport of the radionuclides

to the biosphere, and

- the radionuclides finally entered into the biosphere will be diluted with a large volume of groundwater in adjacent aquifers, and with surface water bodies such as ocean, river, lake and so on, before coming into contact with man.

This concept which is reflected in the modeling scheme adopted in the GSRW may realize the isolation of most of the radionuclides in the waste during a long time, allow the reduction of radionuclide concentrations at the biosphere, and finally maintain radiological consequences to individuals or populations at an acceptable level, even in the case where the ability of some of barriers would be reduced.

The scenario considered here is based on a groundwater migration scenario, assuming that the performance of the disposal system is not affected by probabilistic events as is the case of normal evolution scenarios [10]. It is assumed in the scenarios that all of the components involved in the repository are resaturated eventually with groundwater, after the closure of the repository. Degradation, with groundwater thus contacted, of the components occurs, which results in corrosion of the container and then dissolution of the vitrified matrix. These processes lead to the release of radionuclides into the buffer zone. Since the hydraulic conductivity in the zone may be negligibly low, the transport in the zone is mainly controlled by the diffusion mechanism. The subsequent transport in the geosphere is governed by groundwater flow through fractured porous media. The control processes of the transport are the advection, the dispersion including molecular diffusion and mechanical dispersion, the retention with mineral components of a rock, and the radioactive decay. Radionuclides entered into adjacent aquifers are diluted with a large volume of groundwater and further by surface water bodies.

3. SAFETY ASSESSMENT METHODOLOGY

A basic consideration adopted in the development of the GSRW code was to provide a flexible and practically used analytical tool for the assessment of long-term radiological consequences from repositories. The code has been designed, therefore, to incorporate as many as the major mechanisms which may significantly affect the consequences, among all of the potential mechanisms that will take place in a disposal system. The mechanisms which are identified to be important from safety assessment point of view are modeled to the extent possible, at a level of detail appropriate to the accuracy with which it is considered to describe them, taking into account the current availability of data required to execute the code. This approach will provide the code which will give analytical results being as accurate as is possible without unnecessary impracticality and complexity. In addition, the GSRW has been constructed in a modular form. It is therefore possible to develop alternative versions of individual modules, corresponding to the increase and enhancement in scientific knowledge required for understanding of a disposal system, and then substitute them for the current version incorporated in the code system.

The GSRW is composed of three interlinked models, a source term, a geosphere and a biosphere

to the biosphere, and

- the radionuclides finally entered into the biosphere will be diluted with a large volume of groundwater in adjacent aquifers, and with surface water bodies such as ocean, river, lake and so on, before coming into contact with man.

This concept which is reflected in the modeling scheme adopted in the GSRW may realize the isolation of most of the radionuclides in the waste during a long time, allow the reduction of radionuclide concentrations at the biosphere, and finally maintain radiological consequences to individuals or populations at an acceptable level, even in the case where the ability of some of barriers would be reduced.

The scenario considered here is based on a groundwater migration scenario, assuming that the performance of the disposal system is not affected by probabilistic events as is the case of normal evolution scenarios [10]. It is assumed in the scenarios that all of the components involved in the repository are resaturated eventually with groundwater, after the closure of the repository. Degradation, with groundwater thus contacted, of the components occurs, which results in corrosion of the container and then dissolution of the vitrified matrix. These processes lead to the release of radionuclides into the buffer zone. Since the hydraulic conductivity in the zone may be negligibly low, the transport in the zone is mainly controlled by the diffusion mechanism. The subsequent transport in the geosphere is governed by groundwater flow through fractured porous media. The control processes of the transport are the advection, the dispersion including molecular diffusion and mechanical dispersion, the retention with mineral components of a rock, and the radioactive decay. Radionuclides entered into adjacent aquifers are diluted with a large volume of groundwater and further by surface water bodies.

3. SAFETY ASSESSMENT METHODOLOGY

A basic consideration adopted in the development of the GSRW code was to provide a flexible and practically used analytical tool for the assessment of long-term radiological consequences from repositories. The code has been designed, therefore, to incorporate as many as the major mechanisms which may significantly affect the consequences, among all of the potential mechanisms that will take place in a disposal system. The mechanisms which are identified to be important from safety assessment point of view are modeled to the extent possible, at a level of detail appropriate to the accuracy with which it is considered to describe them, taking into account the current availability of data required to execute the code. This approach will provide the code which will give analytical results being as accurate as is possible without unnecessary impracticality and complexity. In addition, the GSRW has been constructed in a modular form. It is therefore possible to develop alternative versions of individual modules, corresponding to the increase and enhancement in scientific knowledge required for understanding of a disposal system, and then substitute them for the current version incorporated in the code system.

The GSRW is composed of three interlinked models, a source term, a geosphere and a biosphere

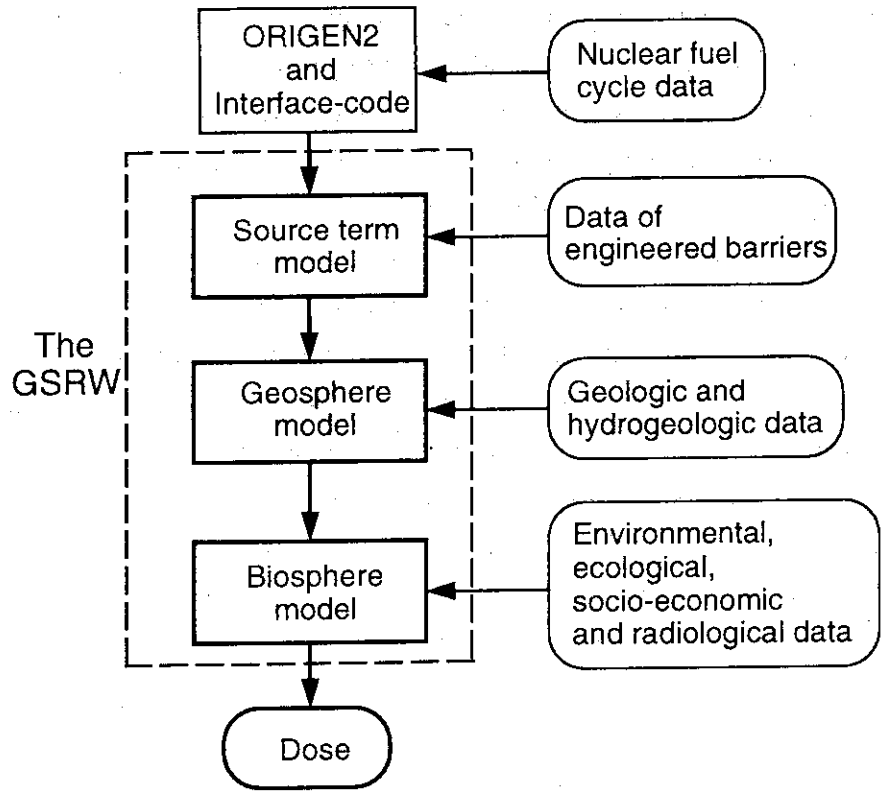

Fig. 2 Sub-models in the GSRW code

model, as illustrated in Fig. 2. The ORIGEN2[11] and interface codes are used to evaluate the inventory of radionuclides in HLW as a function of time. The first model evaluates a source term from a disposal facility which consists mainly of a vitrified waste form, a metallic container and a buffer material. Two kinds of source term models are provided: Model-1 which simulates the dissolution of silicate component of glass and assumes that the radionuclides in the vitrified wastes are released in proportion to the leaching rate of silicate component, and Model-2 which assumes that the concentration of a radionuclide is limited by the solubility of its specific chemical form at the interface between the buffer and the waste. The second model analyzes the transport of radionuclides in the geosphere, which is based on analytical solutions or numerical solutions of a mass transport equation involving an one-dimensional advection, a three or one-dimensional dispersion, a linear sorption and a decay chain. The third model assesses the transport of radionuclides in the biosphere and the resulting radiological consequences to the man, which is based on a dynamic compartment model for the biosphere and a dose factor method for dose calculations.

3.1 Source term model

The source term in the GSRW is defined as the time-dependent rate of radionuclide release from the boundary of a disposal unit (buffer - rock interface). The disposal unit is assumed to be composed of vitrified waste, container and buffer material in this model as shown in Fig. 1. Releases of radionuclides from the vitrified wastes will be initiated by failures of the containers.

The degradations of the containers are mainly induced by corrosion of a metal which contacts with groundwater. In this model, the following simplified approach is adopted : the effect of container-degradation is taken into account as the container failure time which is equal to an delay time to the onset of radionuclide releases. The containers are assumed to fail completely, and provide no function to prevent radionuclide releases after the failure time.

The vitrified waste comes into contact with groundwater after the failure of container, and the radionuclides will be released from the waste. The releases of radionuclides from the waste are simulated by the following two different models:1) the radionuclides (with higher solubilities than silicate component) in the waste are assumed to be released in proportion to the leaching rate of silicate component which is a main component of the waste glass (Model-1), 2) the concentration of the radionuclides (with lower solubilities than silicate component) is assumed to be controlled by a solubility-limit of its chemical form at the interface between the waste glass and buffer material, and the release rate of the radionuclides at the interface between buffer material and rock depends on a diffusive flux through the buffer material (Model-2).

3.1.1 Model-1(congruent dissolution model)

In this model, a finite cylinder of vitrified waste is approximated by a sphere or a prolate spheroid in order to give compact analytical solutions. Radionuclide migration through the buffer material is modeled as a one-dimensional diffusive transport, and the shape of waste form only influences the dissolution rate of silicate component from the waste. The boundary of the waste form and buffer material is restricted by the prescribed nuclide flux which is equal to the product of the dissolution rate of silicate component and the existing ratio of the nuclide, and the concentration of the nuclide at another boundary is assumed to be zero because the advection should be dominant in the rock region.

Exact analytical solutions for the time-dependent rate of mass transfer from waste forms of various shapes were given by Chambre' et al.[12],[13]. The steady-state solution for a low-solubility long-lived species i, assuming constant saturation concentration N_i^{sat}(kg/m^3) in the liquid at the waste surface and assuming that the waste form is surrounded by a porous medium, is

$$f_i = \frac{\beta D_e N_i^{sat}}{n_i} \qquad (1)$$

where f_i : the fractional dissolution rate of species i (1/s),

β : a geometrical factor that can be calculated from the waste form dimensions (1/m^2), (= 3/r^2 for the spherical waste of radius r),

D_e : the effective diffusion coefficient in pore water (m^2/s), and

n_i : the concentration of the species i in the waste form (kg/m^3).

Here we assume that at any time t the dissolution rate can be approximated by the steady-state solution, so that the dissolution rate m(t) (kg/s) of silicate component from the spherical waste is given as

$$m(t) = f_s M(t) = f_s n_s \frac{4}{3}\pi r(t)^3 = 4\pi r(t) D_e N_s^{sat} \tag{2}$$

where suffix s means silicate component, and M(t) is the mass of silicate component at time t in the waste glass (kg). Then, the leach time T which is defined as the time interval between the beginning of dissolution and the completion of dissolution of the waste form is given as follows

$$T = \frac{\rho r_0^2}{2 D_e N_s^{sat}}, \tag{3}$$

where ρ : the waste form density (kg/m^3), and

r_0 : the initial radius of the spherical waste (m),

and the time dependence of the sphere radius r is

$$r(t) = \sqrt{r_0^2 - \frac{2 D_e N_s^{sat}}{\rho} t}. \tag{4}$$

The leaching rate $S_i(t)$ of radionuclide i from the waste glass is given as the product of the dissolution rate m(t) of silicate component and the existing ratio $R_i(t)$, which is given by the Bateman's equation, of the radionuclide in the vitrified waste.

$$S_i(t) = m(t) R_i(t) \tag{5}$$

The leaching rate $S_i(t)$ of radionuclide i is used as a prescribed nuclide flux at the interface between the waste form and buffer material. The chain transport equation considering diffusion and instantaneous linear sorption in the buffer material is

$$K_i^B \frac{\partial N_i}{\partial t} = D_{f,i} \frac{\partial^2 N_i}{\partial x^2} - \lambda_i K_i^B N_i + \lambda_{i-1} K_{i-1}^B N_{i-1} \tag{6}$$

where K_i^B : the retardation factor of radionuclide i in the bentonite (-),

N_i : the concentration of radionuclide i in the bentonite porewater (atom/m^3),

$D_{f,i}$: the pore diffusivity (m^2/s), and
λ_i : the decay constant of radionuclide i (1/s).

The boundary conditions are respectively

$$-\varepsilon D_{f,i}\left.\frac{\partial N_i}{\partial x}\right|_{x=0} = S_i(t)/S \tag{7}$$

at the boundary of the waste form and buffer material, and

$$N_i(L,t) = 0 \tag{8}$$

at the boundary of the buffer material and rock (x=L), where e is the porosity of the buffer material (-), S is the surface area of the buffer material at the innerside (m^2) and L is the thickness of the buffer material. Prior to the time t = 0 the nuclide has zero concentration in the buffer material. The analytical solution of eq. (6) for given initial and boundary conditions was derived for a three-member decay chain, to calculate the nuclide fluxes at the boundary of the buffer material and rock, giving the source term of the geosphere model.

3.1.2 Model-2 (solubility-limit model)

A finite cylinder of the vitrified waste is approximated by a sphere in this model to obtain analytical solutions of a decay chain transport in the buffer material. Radionuclide migration through the buffer material is modeled as a one-dimensional diffusive transport in the spherical coordinates. The concentration of the radionuclide is assumed to be the solubility-limit of its specific chemical form on the surface of the waste form, and to be zero at another boundary because the advection is dominant in the rock region. The release rate of the radionuclide from the buffer material depends on the diffusive flux through the buffer material.

The diffusive transport equation in the buffer material is given as follows;

$$K_i^B\frac{\partial N_i}{\partial t} = D_{f,i}\frac{1}{r}\frac{\partial^2}{\partial r^2}(rN_i) - \lambda_i K_i^B N_i + \lambda_{i-1}K_{i-1}^B N_{i-1}. \tag{9}$$

The boundary conditions are respectively

$$N_i(R_0,t) = N_i^{sat}(t) \tag{10}$$

on the surface of the spherical waste form ($r = R_0$), and

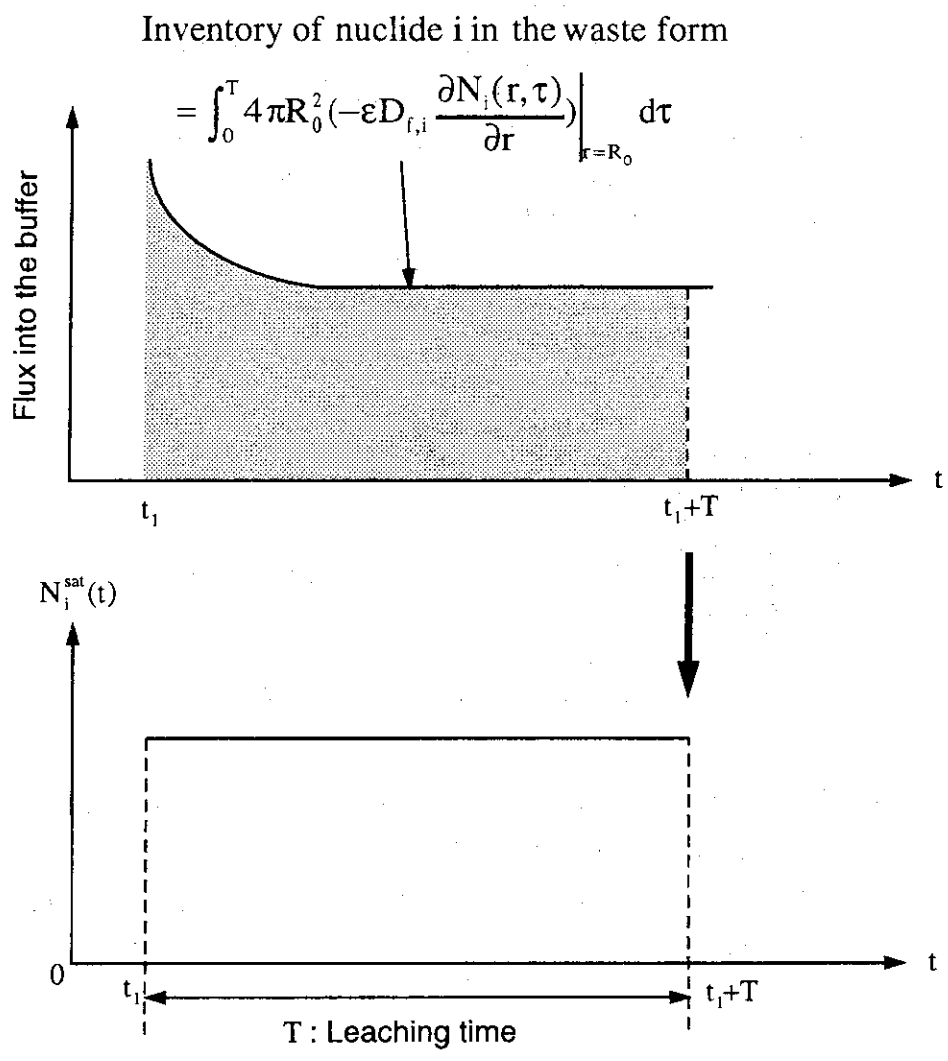

Fig. 3 Estimation of the leaching time in the Model-2 simulation

$$N_i(R_1, t) = 0 \qquad (11)$$

at the boundary of the buffer material and rock ($r = R_1$), where R_0 is the radius of the spherical waste (m), is the solubility-limit of radionuclide i at time t (kg/m^3) and R_1 is a sum of R_0 and the thickness of the buffer material L (m) . The diffusing nuclide has zero initial concentration in the buffer material. Thé analytical solutions of eq. (9) have been obtained for a three-member decay chain, and the nuclide fluxes at the boundary of the buffer material and rock are calculated for the source term of the geosphere model. In this model, the solubility-limit of radionuclide i is given as a time-dependent variable, so as to correspond the situation where it is appropriate to take account of the change of chemical environment in the near field, e.g., due to a large scale tectonic events. The leaching time T is determined by assuming that the integration of the nuclide flux from zero to T with respect to time at the boundary of the waste glass and buffer material is equal to the total

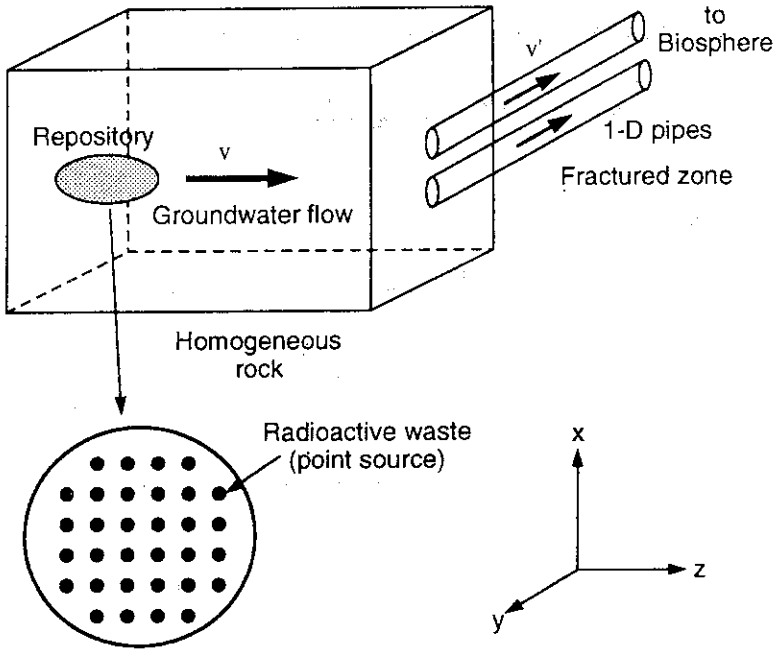

Fig. 4 Conceptual model of the geosphere

inventory of the nuclide in the waste form as shown in Fig. 3. The analytical solution of the nuclide flux on the surface of the waste form has a term of Dirac δ-function like $\delta(r-R_0)$, however it will be neglected by assuming that the integration of the nuclide flux can be done at the neighborhood of the surface of the waste form.

3.2 Geosphere model

The radioactive wastes will be disposed of into a deep stable rock mass which has no major fracture as shown in Fig. 4. Accordingly we can assume that the rock mass is homogeneous, while the scale of a stable rock mass is depend on the specific site, and radionuclide transport in the rock mass is simulated by a one-dimensional advection and three-dimensional dispersion model. Effect of the transverse dispersion might be important for a radionuclide source of finite extent. Waste packages will be emplaced in the waste repository leaving appropriate spaces between the waste packages. The waste packages in the repository can be treated as an array of point sources because the size of the waste package is sufficiently small as compared with the size of the rock mass. The strength of the point source is given by the source term model.

There are several pathways in which radionuclides migrate through the rock mass and transport to the biosphere. Each pathway is probably composed of several fractured zones, and radionuclide transport in it can be simulated by the one-dimensional advection and dispersion model. In this methodology, several pathways to the biosphere can be taken into account for a simulation of radionuclide transport.

In this way, the geosphere model in the GSRW is divided into two parts: 1) one-dimensional advection and three-dimensional dispersion model which describes radionuclide transport around the repository (homogeneous rock mass), 2) one-dimensional advection and dispersion model which

describes radionuclide transport in the far field (pathway from the rock mass to the biosphere). Both geosphere models are connected with each other by a concentration or flux boundary condition as specifying by input data.

3.2.1 One-dimensional advection and three-dimensional dispersion model

A uniform one-dimensional flow field is assumed in this model in order to obtain analytical solutions. We first consider the time-space-dependent concentration of a radionuclide released from a point source in an infinite porous medium, with transverse dispersion. Transport of a chain of decaying solute species is described by the following equation considering equilibrium sorption.

$$K_i^G \frac{\partial N_i^G}{\partial t} = D_x \frac{\partial^2 N_i^G}{\partial x^2} + D_y \frac{\partial^2 N_i^G}{\partial y^2} + D_z \frac{\partial^2 N_i^G}{\partial z^2} - v_z \frac{\partial N_i^G}{\partial z} - \lambda_i K_i^G N_i^G + \lambda_{i-1} K_{i-1}^G N_{i-1}^G + S_i \tag{12}$$

$$D_x = \alpha_x v_z, \qquad D_y = \alpha_y v_z, \qquad D_z = \alpha_z v_z \tag{13}$$

where K_i^G : the retardation factor of radionuclide i in the geosphere (-),

N_i^G : the concentration of radionuclide i in the groundwater(atom/m^3) ,

D_x : the dispersion coefficient along x-direction (m^2/s),

D_y : the dispersion coefficient along y-direction (m^2/s),

D_z : the dispersion coefficient along z-direction (m^2/s),

v_z : the groundwater velocity (m/s),

S_i : the volumetric source term of radionuclide i (atom/m^{3}s),

α_x : the dispersion length along x-direction (m),

α_y : the dispersion length along y-direction (m), and

α_z : the dispersion length along z-direction (m).

The initial and boundary conditions to be solved are

$$N_i^G(x, y, z, 0) = 0 \qquad \text{for any x, y, z,} \tag{14}$$

$$N_i^G(\pm\infty, y, z, t) = N_i^G(x, \pm\infty, z, t) = N_i^G(x, y, \pm\infty, t) = 0 \tag{15}$$

for an infinite medium. Recursive analytical solutions of eq. (12) have been obtained for a decay-chain by Pigford et al.[14]. In this model, analytical solutions for the band release of a four-member decay chain are involved. A release pattern from a point source is obtained by dividing them into

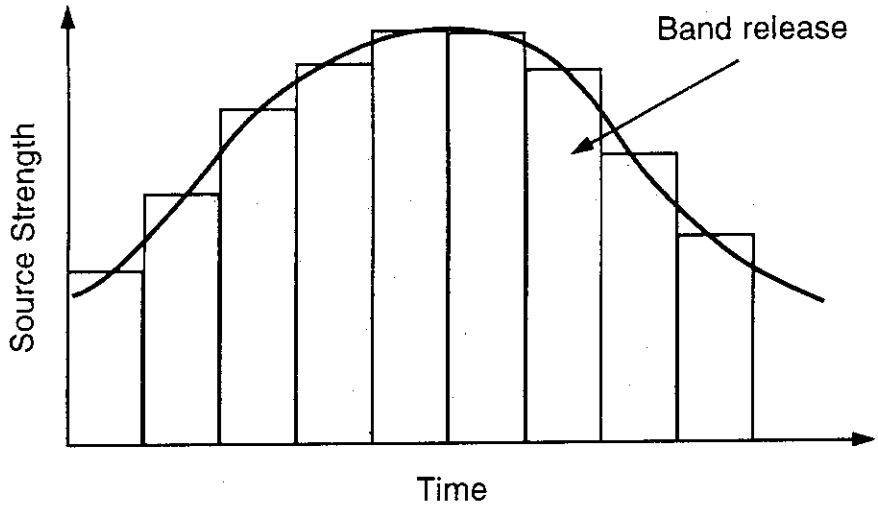

Fig. 5 Treatment of a generalized release pattern from a point source

a series of band releases (this concept does not involve radioactive decay in the source and differs from that defined by Pigford et al.) with different source strengths as shown in Fig. 5.

Now consider a finite array of such point sources emplaced in the infinite medium. If we assume that the nuclide flux from an upstream point source has no effect on the release of nuclide from a downstream point source, the concentration of the array of point sources can be obtained by the superposition of the analytical solutions. This assumption may give a conservative assessment, because such the effect causes the decrease of release rate of nuclide from the downstream sources.

3.2.2 One-dimensional advection and dispersion model

This model is solved by two methods : analytical solutions assuming a homogeneous porous medium, and numerical solutions considering multiple layers. Both methods take account of an equilibrium sorption and decay chain in addition to the advection and dispersion in the porous medium. The governing equation in this case is similar to eq. (12), where the dispersion terms for x and y-directions are neglected. The analytical solutions for the one-dimensional advection and dispersion model are almost the same as those of eq. (12)[15]. The numerical solutions for the one-dimensional multiple layers are obtained by the ordinary finite-difference method based on the implicit method.

3.3 Biosphere model

Once radionuclides emerge from the geosphere, they enter into the environment and cause radiological consequences to the public through a variety of exposure pathways. The long-term safety assessment for the future generation may require consideration of the alteration in the biosphere associated with the cycling of ice age [16]. Due to the large uncertainty about future environmental conditions, the basis of model structures for times far into the future should not depend on detailed environmental features reflecting from a specific site, taking into account that a candidate site for

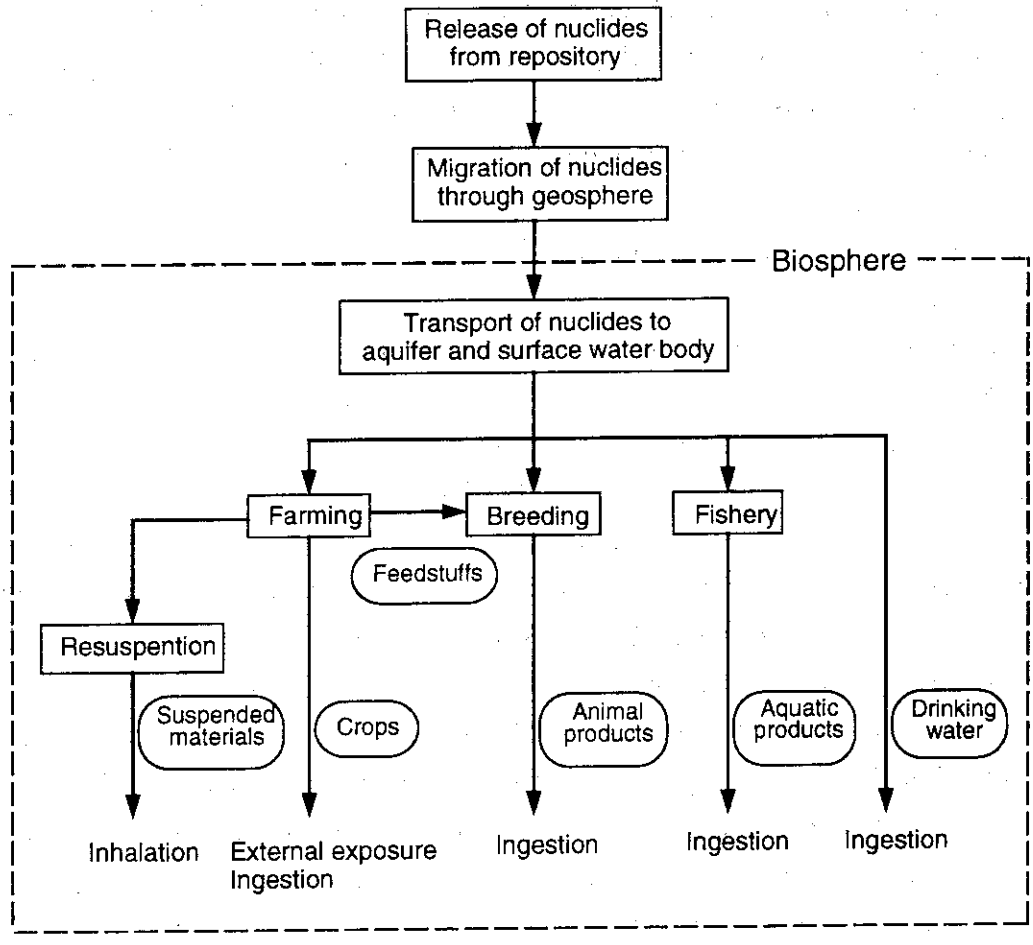

Fig. 6 Transport and exposure pathways involved in the groundwater scenario

HLW disposal is not decided yet in Japan. The basic philosophy adopted in the development of the GSRW is to evaluate the consequences due to radionuclide release under environmental conditions not too far differed from those currently and commonly experienced in Japan.

The biosphere model combined with a dosimetry sub-model, which is based on a time-dependent compartment model, simulates the transport of radionuclides through the biosphere. The model evaluates the internal doses in a term of the committed dose equivalent resulting from both inhalation of contaminated suspended materials and ingestion of contaminated food and drinking water, and the external doses in a term of the dose equivalent due to the immersion in gamma radiation fields. As shown in Fig. 6, the exposure pathways assumed here include:

- internal exposure from inhalation of suspended materials,
- internal exposure from ingestion of food (farm products, aquatic products) and drinking water, and
- external exposure from radionuclides deposited on the surface.

A dosimetry sub-model evaluates the committed dose equivalent (internal exposures) and the dose equivalent (external exposures) for an individual. In addition, the sub-model calculate "postulated" population doses (the committed dose equivalent for a population) assuming that the

annually accumulated radioactivity passing through a vertical section at a certain distance from the repository is all ingested by a local population. This index would be useful particularly for a case where a large number of sensitivity analysis are performed to extract the important pathways and parameters and to identify the relative importance of various barriers involved in a disposal system.

4. SENSITIVITY ANALYSIS METHODOLOGY

The sensitivity and uncertainty analysis are essential to identify the important parameters which dominate the radiological consequence due to the geological disposal. In the past, analytical methods based on standard statistical method and direct numerical method based on perturbation method have been used for sensitivity analysis. For large-scale problem treats many parameters, however, these methods have computational difficulties. Oblow et al.[17], [18], [19] developed an automated procedure for sensitivity analysis using computer calculus. The procedure is embodied in a FORTRAN precompiler called GRESS, which automatically processes computer models adding derivative-taking to the normal calculated results. We also developed the automated procedure for performing large-scale sensitivity studies based on the use of computer tools. The procedure is composed of a FORTRAN precompiler called SANA and a preprocessor called PRESANA. The major differences of both procedures can be seen at the following points. One is the difference of computational methods to obtain the derivatives of parameters. GRESS computes the derivatives analytically, while SANA computes the derivatives using the Differential Algebra (DA) method which was developed by M. Berz [9]. Differential algebras are related to the theories of nonstandard analysis, formal power series and automated differentiation. Another difference is an addition of preprocessor computer tool PRESANA to help the translation of FORTRAN codes. In order to precompile FORTRAN codes, preprocessing operations are necessary and require not a few man powers. The reduction of man power is accomplished by the preprocessor PRESANA.

4.1 Differential Algebra method

The DA-method was first applied for the description of beam dynamics. Any order derivatives of parameters can be obtained by this method according to the power and the memory of computer system. Parameters related to sensitivity study in a given FORTRAN program are treated as vector variables (DA variables) in the DA-method. Computation of high order (> 2) derivatives requires large memory and "Supercomputing" technique.

A brief description of DA-method is given here with the help of a simple example (0-th and first order derivatives). Consider the vector space R^2 of ordered pairs (a_0, a_1), $a_0, a_1 \in R$ in which an addition and a scalar multiplication are defined in the usual way:

$$(a_0, a_1) + (b_0, b_1) = (a_0 + b_0,\ a_1 + b_1) \tag{16}$$

annually accumulated radioactivity passing through a vertical section at a certain distance from the repository is all ingested by a local population. This index would be useful particularly for a case where a large number of sensitivity analysis are performed to extract the important pathways and parameters and to identify the relative importance of various barriers involved in a disposal system.

4. SENSITIVITY ANALYSIS METHODOLOGY

The sensitivity and uncertainty analysis are essential to identify the important parameters which dominate the radiological consequence due to the geological disposal. In the past, analytical methods based on standard statistical method and direct numerical method based on perturbation method have been used for sensitivity analysis. For large-scale problem treats many parameters, however, these methods have computational difficulties. Oblow et al.[17], [18], [19] developed an automated procedure for sensitivity analysis using computer calculus. The procedure is embodied in a FORTRAN precompiler called GRESS, which automatically processes computer models adding derivative-taking to the normal calculated results. We also developed the automated procedure for performing large-scale sensitivity studies based on the use of computer tools. The procedure is composed of a FORTRAN precompiler called SANA and a preprocessor called PRESANA. The major differences of both procedures can be seen at the following points. One is the difference of computational methods to obtain the derivatives of parameters. GRESS computes the derivatives analytically, while SANA computes the derivatives using the Differential Algebra (DA) method which was developed by M. Berz [9]. Differential algebras are related to the theories of nonstandard analysis, formal power series and automated differentiation. Another difference is an addition of preprocessor computer tool PRESANA to help the translation of FORTRAN codes. In order to precompile FORTRAN codes, preprocessing operations are necessary and require not a few man powers. The reduction of man power is accomplished by the preprocessor PRESANA.

4.1 Differential Algebra method

The DA-method was first applied for the description of beam dynamics. Any order derivatives of parameters can be obtained by this method according to the power and the memory of computer system. Parameters related to sensitivity study in a given FORTRAN program are treated as vector variables (DA variables) in the DA-method. Computation of high order (>2) derivatives requires large memory and "Supercomputing" technique.

A brief description of DA-method is given here with the help of a simple example (0-th and first order derivatives). Consider the vector space R^2 of ordered pairs (a_0, a_1), $a_0, a_1 \in R$ in which an addition and a scalar multiplication are defined in the usual way:

$$(a_0, a_1) + (b_0, b_1) = (a_0 + b_0, a_1 + b_1) \tag{16}$$

$$t \cdot (a_0, a_1) = (ta_0, ta_1) \tag{17}$$

for $a_0, a_1, b_0, b_1 \in R$. Components a_0 and b_0 denote 0-th order derivatives (real part), and components a_1 and b_1 denote first order derivatives (differential part). Besides the above addition and scalar multiplication a multiplication and a quotient between vectors are introduced in the following way:

$$(a_0, a_1) \cdot (b_0, b_1) = (a_0 \cdot b_0, a_0 \cdot b_1 + a_1 \cdot b_0) \tag{18}$$

$$\frac{(a_0, a_1)}{(b_0, b_1)} = \left(\frac{a_0}{b_0}, \frac{a_1}{b_0} - \frac{a_0 \cdot b_1}{b_0^2}\right) \quad (\text{if } b_0 \neq 0) \tag{19}$$

As an example, consider the following function:

$$f(x) = \frac{1}{x + 1/x} \tag{20}$$

The derivative of the function is

$$f'(x) = \frac{1/x^2 - 1}{(x + 1/x)^2}. \tag{21}$$

Suppose we are interested in the value of the function and its derivative at $x=2$. We obtain

$$f(2) = \frac{2}{5}, \quad f'(2) = -\frac{3}{25}. \tag{22}$$

Now take the definition of the function f in the equation (20) and evaluate it at $(2,1)$. Because the function f is composed of the basic function x, and its derivative at $x=2$ is 1. We obtain:

$$\begin{aligned} f[(2,1)] &= \frac{1}{(2,1) + (1,0)/(2,1)} = \frac{1}{(2,1) + (1/2, -1/4)} \\ &= \frac{1}{(5/2, 3/4)} = (2/5, -\frac{3}{4} \Big/ \frac{25}{4}) \\ &= (\frac{2}{5}, -\frac{3}{25}) \end{aligned} \tag{23}$$

As we can see, after the evaluation of the function that the first component is just the value of the function at $x=2$, whereas the second component is the derivative of the function at $x=2$. For a

function including higher order derivatives, the derivatives of the function are obtained in a similar way.

4.2 SANA Precompiler and PRESANA Preprocessor

We assume that a FORTRAN program is computing a vector quantity **F**. This vector quantity will depend on a set of input parameters denoted by **p**. The basic problem in any sensitivity study is to find the rate of change (d**F**/d**p**) in the result F arising from changes in input model parameters **p**. The output of the program will be calculated by the rules of arithmetics, FORTRAN intrinsic functions and/or user subroutines. Therefore, if the FORTRAN program and intrinsic functions are expressed in the rules of arithmetics defined by the Differential Algebra, we can compute the derivatives of **F** about **p**. The purpose of the SANA precompiler is to help us writing such FORTRAN program. The SANA code reads the FORTRAN source code text, and replaces any differentiable operation by an operation involving the computation of the original quantity and the derivatives of that quantity with respect to the chosen independent parameters p. In practice this is achieved by replacing the original FORTRAN statement by a series of CALL statements to DA-library. The DA-library is composed of the subroutines and the FORTRAN intrinsic functions which perform various DA operations. It is necessary to precompile the FORTRAN code by the SANA code that user assigns the names and the array sizes of DA variables and DA statements need DA operations in the original FORTRAN source. This work could be done by the preprocessor PRESANA. The following preprocessing works are sequentially performed by the PRESANA code:

- translation of logical IF statement to block IF statement search of the main program
- preservation of information about the variables appeared in the subroutines
- preservation of COMMON information
- pursuit how the DA variables propagate in the original FORTRAN program
- pick up non-DA variables need not differentiation (defined as RI variables ; scalar variables)
- various checking work about type, variables and statement preparation of main program for the DA-program (sensitivity analysis program)
- generate DA-commands specifying the names and the array size of DA and RI variables, and remove type statements and array declarators of DA variables in the original program
- generate DA-commands in the assignment statements including the DA variables
- convert the DA variables appeared in the WRITE and IF statements to the RI variables
- specification of input parameters for sensitivity analysis
- specification of output variables for sensitivity analysis

4.3 Verification of DA-method

As an actual example of this process, consider a small program shown in Fig. 7. This simple FORTRAN program calculates the following function

```
*PRE-NO-NV NO=1,NV=2
      PROGRAM MAINP
      IMPLICIT DOUBLE PRECISION(A-H,O-Z)
      COMMON /AAA/ AA1,AA2
      DIMENSION BB1(10),BB2(10),CC1(10)
      READ(5,*)AA10,AA2,AA30
*PRE-INPUT 1 AA1=AA10 'INPUT PARAMETER AA1'
*PRE-INPUT 2 AA3=AA30 'INPUT PARAMETER AA3'
      DO 100 I=1,10
       BB1(I)=I*AA1 + AA3
       BB2(I)=I*AA2
  100 CONTINUE
      CALL CALC(BB1,BB2,CC1)
      DO 200 I=1,10
*PRE-OUTPUT(6) CC1(I) 'RESULT CC1(#)'
  200 CONTINUE
      STOP
      END
C
      SUBROUTINE CALC(XX,YY,ZZ)
      IMPLICIT DOUBLE PRECISION(A-H,O-Z)
      COMMON /AAA/ AA1,AA2
      DIMENSION XX(10),YY(10),ZZ(10),WW(10)
      DO 100 I=1,10
       WW(I)=XX(I)*XX(I)
       IF (WW(I) .GT. 100.D0) WW(I)=100.D0
       ZZ(I)=WW(I)+SIN(YY(I))*AA2
  100 CONTINUE
      RETURN
      END
```

← assign No. of DA-parameters (NV) and degree of derivative (NO)

← assign input parameters for sensitivity analysis

← assign output parameter

Fig. 7 Program list of example FORTRAN code

```
*PRE-NO-NV NO=1,NV=2
*      MAIN PROGRAM OF DA
       IMPLICIT DOUBLE PRECISION(A-H,O-Z)
       INTEGER NODA, NVDA
       COMMON /DATFLG/ DATFLG
       INTEGER DATFLG
       COMMON /DANONV/ NODA, NVDA
       DATFLG = 0
       NODA =   1
       NVDA =   2
       CALL DAINI( NODA, NVDA, 1 )
       CALL DAINIT
       CALL MAINP
       STOP
       END
       SUBROUTINE DAINIT
       IMPLICIT DOUBLE PRECISION(A-H,O-Z)
       INTEGER NODA, NVDA
       COMMON /DATFLG/ DATFLG
       INTEGER DATFLG
       COMMON /DANONV/ NODA, NVDA
       DOUBLE PRECISION AA2
       COMMON/AAA/AA1,AA2
*SAN   B D ;
*SAN   D V DA COM AA1 NODA NVDA ;
*SAN   E D ;
       IF (DATFLG.EQ.1) WRITE(6,'(A)')  '*START DAINIT'
       RETURN
       END
       SUBROUTINE MAINP
       IMPLICIT DOUBLE PRECISION(A-H,O-Z)
       COMMON /AAA/ AA1,AA2
       DIMENSION BB2(10)
*      DIMENSION BB1(10),BB2(10),CC1(10)
       INTEGER  NODA, NVDA
       COMMON /DATFLG/ DATFLG
       INTEGER DATFLG
       COMMON /DANONV/ NODA, NVDA
*SAN   B D ;
*SAN   D V DA EXT AA1 NODA NVDA ;
*SAN   D V IN INT I ;
*SAN   D V DA INT BB1 NODA NVDA 10 ;
*SAN   D V DA INT AA3 NODA NVDA ;
*SAN   D V DA INT CC1 NODA NVDA 10 ;
*SAN   E D ;
       IF (DATFLG.EQ.1) WRITE(6,'(A)')  '*START MAINP '
       READ(5,*)AA10,AA2,AA30
       CALL DAVAR( AA1, AA10, 1 )
*PRE-INPUT 1 AA1=AA10 'INPUT PARAMETER AA1'
       CALL DAVAR( AA3, AA30, 2 )
*PRE-INPUT 2 AA3=AA30 'INPUT PARAMETER AA3'
       DO 100 I=1,10
*DA    BB1(I)=I*AA1+AA3 ;
*       BB1(I)=I*AA1 + AA3
        BB2(I)=I*AA2
  100  CONTINUE
       CALL CALC(BB1,BB2,CC1)
       DO 200 I=1,10
       WRITE( 6, * ) ' RESULT CC1(#)'
       CALL DAPRI( CC1(I), 6 )
*PRE-OUTPUT(6) CC1(I) 'RESULT CC1(#)'
  200  CONTINUE
       RETURN
*      STOP
       END
C
       SUBROUTINE CALC(XX,YY,ZZ)
       IMPLICIT DOUBLE PRECISION(A-H,O-Z)
       COMMON /AAA/ AA1,AA2
       DIMENSION YY(10)
*      DIMENSION XX(10),YY(10),ZZ(10),WW(10)
       INTEGER  NODA, NVDA
       COMMON /DATFLG/ DATFLG
       INTEGER DATFLG
       COMMON /DANONV/ NODA, NVDA
*SAN   B D ;
*SAN   D V DA EXT AA1 NODA NVDA ;
*SAN   D V RE EXT AA2 ;
*SAN   D V DA EXT XX NODA NVDA 10 ;
*SAN   D V RE EXT YY 10 ;
*SAN   D V DA EXT ZZ NODA NVDA 10 ;
*SAN   D V IN INT I ;
*SAN   D V DA INT WW NODA NVDA 10 ;
*SAN   E D ;
       IF (DATFLG.EQ.1) WRITE(6,'(A)')  '*START CALC  '
       DO 100 I=1,10
*DA    WW(I)=XX(I)*XX(I) ;
*       WW(I)=XX(I)*XX(I)
       IF(DARE(WW(I)).GT.100.D0)THEN
*      IF( WW(I).GT.100.D0 ) THEN
*DA    WW(I)=100.D0 ;
*          WW(I)=100.D0
       ENDIF
*       IF (WW(I) .GT. 100.D0) WW(I)=100.D0
*DA    ZZ(I)=WW(I)+SIN(YY(I))*AA2 ;
*       ZZ(I)=WW(I)+SIN(YY(I))*AA2
   100 CONTINUE
       RETURN
       END
```

Fig. 8 Program list of Pre-DA code generated by the PRESANA code

Table1 The results of sensitivity analysis code generated by the DA-method

x=1, y=2, z=0.2			
i	$f_i(x,y,z)$	$\partial f_i/\partial x$	$\partial f_i/\partial y$
1	.9040D+01	.6000D+01	.6000D+01
2	.1608D+02	.1600D+02	.8000D+01
3	.2511D+02	.3000D+02	.1000D+02
4	.3614D+02	.4800D+02	.1200D+02
5	.4917D+02	.7000D+02	.1400D+02
6	.6419D+02	.9600D+02	.1600D+02
7	.8120D+02	.1260D+03	.1800D+02
8	.1002D+03	.1600D+03	.2000D+02
9	.1002D+03	.0000D+00	.0000D+00
10	.1002D+03	.0000D+00	.0000D+00

$$f_i(x,y,z)=\begin{cases}(ix+y)^2+z\cdot\sin(iz) & \text{for } (ix+y)^2\le 100\\ 100+z\cdot\sin(iz) & \text{for } (ix+y)^2>100\end{cases} \qquad (24)$$

for integer i=1~10 and arbitrary input values of x (AA1), y (AA3) and z (AA2). In this source program, information to be necessary for the PRESANA code is given by the statement typed '*PRE' in the first 4 columns. This information contains specifications of input and output parameters for sensitivity analysis. In this case, parameters AA1 and AA3 are chosen as the input parameters, and CC1(I) (f_i) is the output parameter. Output list of the PRESANA code is shown in Fig. 8. In this output list, the statements typed '*SAN' in the first 4 columns specify the names and the array sizes of DA and RI variables. The statements typed '*DA' in the first 3 columns mean that these statements include the DA operations. The target DA program (sensitivity analysis code) is obtained by precompiling this source program using the SANA code. All these procedures are illustrated in Fig. 9.

The results of the sensitivity analysis code generated from these procedures are shown in Table 1. The derivative values obtained from the DA-method coincide with the exact solution of $\partial f_i/\partial x$ and $\partial f_i/\partial y$. Especially, zero sensitivities are correctly calculated for . This confirms that the present automated method for the sensitivity analysis has enough accuracies and applicabilities.

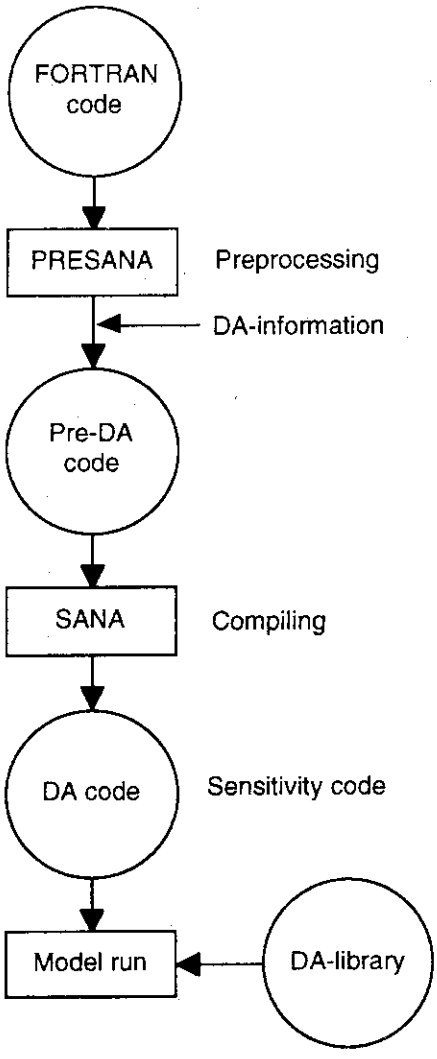

Fig. 9 Flow chart of the DA-method

5. BASIC ASSUMPTIONS AND INPUT DATA

This assessment of the potential radiological consequences of geologic disposal of HLW is based on a generic approach rather than site specific one, reflecting from the current situation mentioned above. However, it might be required to define the disposal system to some extent so as to enable the analysis of the transport of radionuclides in geologic formations and the biosphere following entry of groundwater into a repository and the release of them to the surrounding strata. A crystalline bedrock is now considered to be one of potential geologic strata for geologic disposal of HLW in Japan. We thus assume that a repository will be constructed in a deep and stable granite bedrock at one of depths ranging from 500 m to 1000 m. The main feature of a disposal system postulated here is a multi-barrier concept involving a container/over pack surrounded with a bentonite buffer to reduce water flow rate around the container. High-level radioactive wastes are, before disposal, stored for 30 - 50 years with the purpose of decreasing the heat generation rate to an acceptable level. According to the disposal concept proposed by the PNC, a waste container is further encapsulated in a carbon steel over-pack with a thickness of 30 cm, in order to protect the waste from coming into contact with groundwater over at least 1000 years. During this period, a large portion of heat generating radionuclides might be decayed out into insignificant levels. Consequently, the effect of thermal load to the migration of radionuclides, i.e., a buoyancy effect, will be negligible at the time initiating the migration. This study does not deal with the coupled thermo-hydro-mechanical analysis, because it requires detailed site specific data.

5.1 Inventory and source term

By the end of 1993, there were 46 nuclear power plants in operation at 17 sites in Japan, holding a total electrical capacity of ca. 37,000 MWe. Almost all of the plants are the light water type, including 24 BWR and 20 PWR. All the spent nuclear fuels leaving the plants will be reprocessed to recover fissile Pu and U. This assessment uses the following assumptions to calculate the inventory of radionuclides in HLW:

- the spent fuels from BWR are only taken into account,
- the fuel contains 3% of fissile ^{235}U,
- the burnup of the fuel is 27,500 MWD/MTU, and the specific power is 25.9 MW/MTU,
- the irradiation time is 5 years,
- the vitrified waste production is 0.12 m^3/MTU, and
- the transfer rate of U, Pu and I from the fuel into the vitrified waste is equally 1%.

Several studies reported different values of the transfer rates of U and Pu (PAGIS[10], PNC[7], NAGRA[4] etc.) from each other, reflecting from the fact that the rate depends on a design and an operation experience of the reprocessing plant. This study, therefore, used the normalized transfer rate (1%) for U and Pu, which is considered to be enough conservative. This approach also may enable the conversion of the results of the analysis to those corresponding to the realistic case.

Here, the vitrified waste is assumed to contain ^{129}I which is one of the most hazardous nuclides

Table 2 The calculated inventory of major radionuclides in 0.16 m^3 HLW glass

Nuclide	No. of atoms	Activities (Bq)
Se-79	4.9682E+22	1.678E+10
Zr-93	5.2852E+24	7.732E+10
Nb-94	5.0859E+18	5.582E+6
Tc-99	5.3363E+24	5.525E+11
Pd-107	1.2451E+24	3.904E+9
Sn-126	1.3365E+23	1.467E+10
I-129	9.0089E+21	1.236E+7
Cs-135	1.9418E+24	2.030E+10
Cf-252	8.0224E+4	6.670E-4
Cm-248	3.2465E+16	2.096E+3
Pu-244	3.2611E+17	8.729E+1
Pu-240	8.7480E+22	2.927E+11
Np-236	7.5536E+17	1.382E+5
U-236	1.3268E+23	1.245E+8
Th-232	6.2262E+18	9.762E+0
Cm-245	1.0754E+21	2.777E+9
Pu-241	1.8249E+18	2.782E+9
Am-241	1.7035E+23	8.655E+12
U-237	1.6306E+15	1.935E+9
Np-237	1.8478E+24	1.895E+10
Pa-233	6.9579E+11	2.067E+5
U-233	5.1606E+20	7.124E+7
Th-229	1.0186E+18	3.063E+6
Cm-250	8.5196E+7	2.527E-4
Cm-246	9.5383E+19	4.380E+8
Am-242m	1.5168E+19	2.361E+9
Am-242	1.9686E+14	2.361E+9
Pu-242	1.2390E+22	7.233E+8
U-238	3.2052E+25	1.574E+8
Cm-242	1.4962E+14	7.363E+6
Pu-238	1.1590E+18	2.901E+8
U-234	3.2490E+21	2.911E+8
Th-230	8.2069E+18	2.389E+6
Ra-226	3.0435E+16	4.175E+5
Cm-247	8.1976E+17	1.139E+3
Am-243	1.4915E+23	4.442E+11
Np-239	5.8312E+2	1.991E-3
Pu-239	1.7080E+23	1.556E+11
U-235	3.2307E+23	1.007E+7
Pa-231	3.9786E+17	2.662E+5

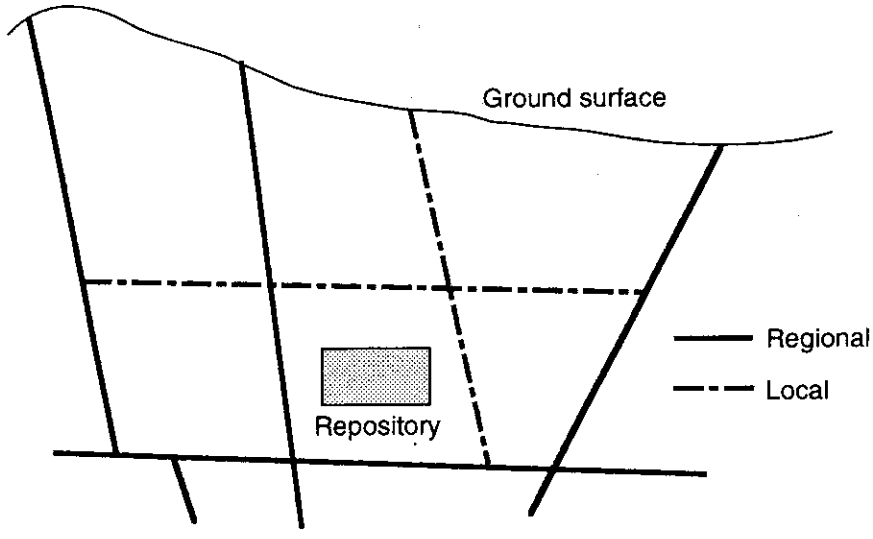

Fig. 10 Location of a potential HLW repository and fractured zones

from a radiological point of view, although a large portion of radioactive iodines in the spent fuel will be release as a gaseous state at the process dissolving the fuel during reprocessing. This assumption also permits the evaluation of radiological consequences due to the case where ^{129}I removed from off-gases during reprocessing is packaged as a solid and disposed in the same repository. The inventory of radionuclides in HLW was calculated by using ORIGEN2 and I/F codes based on the assumptions mentioned above. This approach assumes that all the waste is generated simultaneously, whereas it would be produced over several decades. This simplification, however, will have an insignificant effect on the results of the assessment, taking into account of the time scale involved in the assessment. The calculated inventory of radionuclides in 0.16 m^3 waste glass in one container is listed in Table 2. The assessment made here, however, dealt with long-lived radionuclides that might have radiological significance in the long-term safety analysis.

The source term, i.e., the fluxes of radionuclides released from the waste glass into groundwater at the interface between the buffer material and the surrounding rock, was calculated using Model-2 (solubility-limit model), while only for ^{129}I and ^{135}Cs, Model-1 (congruent dissolution model) was used because of their relatively high solubilities. This calculation assumes for simplification that the release of radionuclides initiates 1000 years after the closure of the repository, according to the disposal system concept proposed by the PNC[7]. However, this assumption does not mean that there is not entry of groundwater into the repository, but implies only that no significant leaching of the waste glass takes place until this time.

The thickness of the buffer material is assumed to be 0.5 m as a standard value, and it is subjected to sensitivity analyses by varying the value ranging from 0.1 m to 1.5 m. The diffusion coefficient in the buffer material is assumed to be 1 x 10^{-10} m^2/s regardless of kinds of nuclides, taking into account of the reported values[20], and the porosity of the buffer is 0.3. The solubility and retardation factor used here will be discussed in the following section (5.3 Geochemistry).

Table 3 The hydraulic conductivity and porosity of granite in Japan

rock type	hydraulic conductivity (m/s)	porosity (-)
fractured zone	1E-7 ~ 1E-6	0.1 ~ 0.25
rock mass	1E-12 ~ 1E-9	0.003 ~ 0.05

5.2 Hydrology

The rock is described in terms of two major hydraulic units; fractured zones and rock mass. The fractured zones are classified into regional fractured zones and local fractured zones. It might be reasonable to assume that the repository is constructed in a stable rock mass surrounded by local fracture zones which further connect with regional fracture zones, as shown in Fig. 10, in order to avoid the occurrence of a short circuit of groundwater from the repository to the biosphere. In principle, flow paths from the repository to the biosphere should be identified by detailed groundwater flow analyses based on characterization data of a specific site to be analyzed. This kind of analyses give a groundwater flow vector and a possible migration length in a geologic system, and require extensive site specific data which are still beyond the current data availability in Japan.

This report assumes a typical hydraulic condition of the geologic system in Japan. However, very little is known about the hydrology of deep geologic system. In view of the lack of information and also the uncertainties about present and future hydrology, the approach used in this analysis is to consider the variability of major model parameters concerned with the hydrology. The groundwater flow velocity v (real velocity, m/s) is given as follows;

$$v = \frac{k}{\phi} I \tag{24}$$

where k : hydraulic conductivity (m/s),

ϕ : flow porosity (-),

I : hydraulic gradient (-).

The hydraulic conductivity and porosity of granite in Japan are shown in Table 3 [21]. Although the hydraulic gradient depends on a specific hydrological system to be analyzed, we assume here the value of 0.01 taking account of the value (0.005) used by SKI[5], because the repository will be constructed in a region of low hydraulic gradient. The groundwater flow velocity is correspondingly in the rage 0.00006 to 0.1 m/y in the rock mass, and in the rage 0.1 to 3 m/y in the fractured zones regardless of local and regional fractures. This analysis uses 0.1 and 1 m/y as the conservative flow velocity in the rock mass and the fractured zones, respectively. The migration length of radionuclides from the repository to the biosphere depends on site specific geological situations. This study arbitrarily assumes that the migration length in the homogeneous rock mass is in the range from

Table 4 Elemental solubility limits in the reducing conditions

Element	Solubility Limits (mol/ℓ)	
	Realistic	Conservative
Se	3E-8	1E-7
Zr	1E-10	1E-9
Nb *	1E-9	1E-5
Tc	2E-9	1E-6
Pd *	2E-7	2E-6
Sn	1E-11	1E-9
I	x	x
Cs	x	x
Th	2E-9	2E-8
Pa	2E-8	1E-5
U	4E-8	1E-5
Np	4E-9	4E-9
Pu	4E-8	3E-5
Am	4E-7	1E-3
SiO_2 **	5E-4	5E-4

x : congruent leaching
* : SKB-91
** : PSI-Bericht Nr. 74

100 to 500 m, depending on possible scales of a stable rock mass, and thus the length is subjected to the sensitivity analysis. The migration length in the fractured zones depends on fracture networks of a specific geological formation to be analyzed, and is thus rather complicated to assume a typical value. However, the preliminary sensitivity analysis which used the same conditions described in this report clearly showed that the consequence depended insignificantly on this parameter. The length is therefore rather arbitrarily assumed to be 1000 m.

5.3 Geochemistry

The geochemistry of geologic environment, especially chemical and electrochemical characteristics of deep groundwater, governs the behavior of radionuclides, i.e., dissolution - precipitation equilibrium, sorption-desorption equilibrium, and so on, which determines the release of radionuclides from the waste glass and the migration of radionuclides in geologic formations. Groundwater in deep geologic formations is characterized by a specific electrochemical condition, i.e.., a reducing condition, because of a relatively low oxygen concentration, and also by a chemical

Table 5 Distribution coefficients for bentonite and granite

Element	Kd-values (m^3/kg)					
	Buffer material (reducing conditions)		Granite [homogeneous-rock] (reducing conditions)		Granite [fractured-zone] (oxidising conditions)	
	Realistic	Conservative	Realistic	Conservative	Realistic	Conservative
Se	0.01	0.01	0.001	0.001	0.001	0.001
Zr	0.1	0.02	4	0.1	4	0.1
Nb	0.1	0.02	4	0.1	4	0.1
Tc	0.05	0	0.005	0	0.0002	0
Pd	0.01	0.002	0.01	0.002	0.005	0.001
Sn	0.1	0.01	0.01	0.001	0.01	0.001
I	0	0	0	0	0	0
Cs	0.02	0.002	0.05	0.005	0.05	0.005
Ra	-	-	0.1	0.005	0.1	0.005
Ac	-	-	-	-	0.5	0.01
Th	1	0.002	5	0.01	5	0.01
Pa	-	-	0.1	0.01	0.01	0.001
U	0.2	0.1	5	0.01	0.01	0.002
Np	1	0.1	5	0.1	0.01	0.001
Pu	1.5	0.1	5	0.5	3	1
Am	2	0.25	5	0.5	5	0.5

Table 6 Assumed porosities and densities of the buffer material and granite rock

material/rock	porosity (-)	density (kg/m^3)
Buffer	0.3	1500
Granite (homogeneous)	0.03	2300
Granite (fractured zone)	0.1	2000

nature, i.e., it involves various kinds of co-existing ions equilibrated with chemical components of the formations in groundwater. The chemical equilibrium which determines the solubility and the sorption behavior of radionuclides is dominated by these characteristics mentioned above which are highly site specific in nature.

The solubility, to be used in this analysis, of radionuclides in natural groundwater is somewhat different from that measured in a solution prepared in a laboratory, because in natural groundwater the dissolution-precipitation process involves complex multicomponent chemical reactions. In order to obtain the solubility of a specific element in a specific geological environment, therefore, it is necessary to use a thermo-chemical computer code such as EQ3/6 [22], PHREEQE [23] and so on. As mentioned before, taking into account of the lack of site specific scientific basis concerned upon which computations are implemented, we used the solubility data, as shown in Table 4, citing from the SKI report[5] and other sources (SKB[3] and PSI[24] reports) for the elements Nb, Pd and Si which are not appeared in the SKI report. The values of the distribution coefficient for granite (homogeneous rock and fractured zones) and bentonite are also cited from the SKI and SKB reports as shown in Table 5. At the oxidizing conditions, Pa tends to form oxyacids which may be unfavorable to adsorb onto granite[25], especially for the case of chemical adsorption mechanisms. This study therefore used lower Kd values for Pa at the oxidizing condition than those appeared in the SKI report which used the same value for Pa at both reducing and oxidizing conditions. In these tables, the word "realistic" value means reference one and "conservative" value implies the upper bound[5]. Table 6 shows assumed porosities and densities of the buffer material and granite rock used for the estimation of the retardation factors [26]. These values are relatively conservative comparing with other reports (SKI, SKB and PNC).

6. SENSITIVITY ANALYSIS

We applied the automated sensitivity analysis methodology (DA-method) to GSRW code to identify the relative importance of parameters which have the possibility to dominate radiological consequences due to geologic disposal of HLW, and to clarify the time-dependent sensitivity behavior of a specific parameter. The analysis was made on one of radiologically important TRU elements (^{237}Np decay chain involving ^{233}U and ^{229}Th), and that of fission products (^{135}Cs), separately. This intends to clarify the difference in sensitivity behavior of Model-1 which is used for ^{135}Cs and Model-2 which is used for ^{237}Np decay chain, and also to see the release and transport behaviors of the decay chain of ^{237}Np in engineered and natural barriers.

Figure 11 shows the conceptualized disposal system assumed here for the sensitivity analysis. The migration length in a homogeneous rock surrounding a disposal facility was assumed arbitrarily to be 300 m, and that in a fractured zone connecting to surface water bodies was 1000 m. Abeline et al. estimated Peclet-numbers (= migration length / dispersion length) of advection-dispersion model to be in a range of 1 to 6 so as to fit measured breakthrough curves obtained by Stripa 3-D Migration experiment [27]. Therefore, the longitudinal dispersion length in the homogeneous rock and the dispersion length in the fractured zone is assumed to be 1/10 of the migration length conservatively, and the transverse dispersion length in the homogeneous rock to be 1/50. The facility was simulated by a simple vertical array of 16 point sources as shown in Fig. 11, each of which involves 3 containers, only to examine the effect of distance between waste packages on the

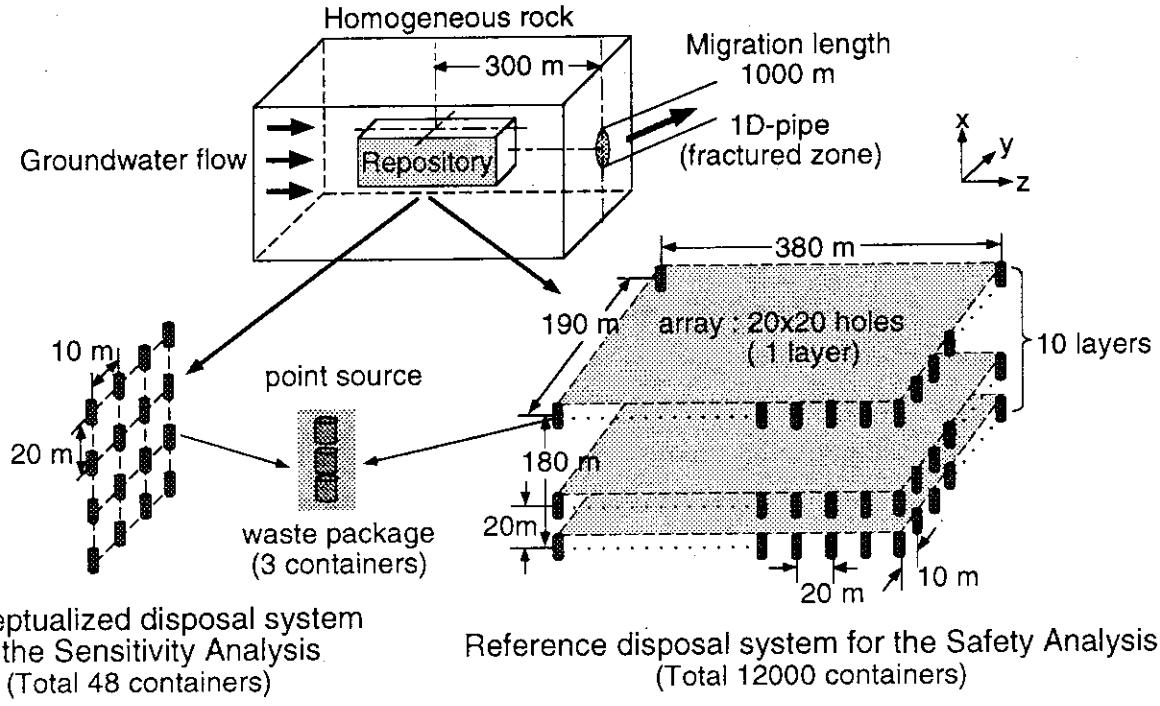

Fig. 11 Conceptualized disposal system for the sensitivity analysis and reference disposal system for the safety analysis

concentration of the nuclides at an outlet boundary of the geosphere. This simple array used here also permits the reduction in computation time.

The concentration of each nuclide at the outlet boundary of the geosphere was chosen as a performance measure in this analysis. The sensitivity of the concentration thus calculated to a specific input parameter is expressed in term of a normalized sensitivity coefficient defined by the ratio of the percent change in the concentration to that in the input parameter value, as a function of time. The following parameters were selected for the sensitivity analysis, based on preliminary analyses:

[Source term model]

- solubility of element i (N_i^{sat})
- thickness of the buffer material (L_B)
- density of the waste form (ρ)
- retardation factor of nuclides in the buffer material (K_i^B)
- diffusion coefficient of nuclide i in the buffer material ($D_{f,i}$)
- radius of the waste form (R_0)

[Geosphere model]

- distance between point sources in the x and y directions (Δx and Δy)
- dispersion lengths of the homogeneous rock (α_x, α_y, α_z)
- groundwater velocity in the homogeneous rock (v_h)
- retardation factor of nuclide i in the homogeneous rock ($K_h^G(i)$)
- migration length in the homogeneous rock (L_h)
- dispersion length in the fractured zone (α_f)
- groundwater velocity in the fractured zone (v_f)
- retardation factor of nuclide in the fractured zone ($K_f^G(i)$)
- migration length in the fractured zone (L_f)

6.1 Sensitivity behavior of source term model

Figure 12 shows the time-dependent concentration profile of ^{135}Cs at the outlet boundary of the geosphere and sensitivity coefficients of parameters, i.e., the diffusion coefficient and the retardation factor of cesium in the buffer material (the transport of cesium in the buffer material), the thickness of the buffer material, the density of the waste form, the effective diffusion coefficient and solubility of SiO_2 (the release of Cs from the waste form). A positive value of the sensitivity coefficient means that the concentration of a nuclide increases in proportion to the increment of the corresponding parameter, and vice verse for a negative value. The sensitivity patterns of the effective diffusion coefficient and the solubility of SiO_2 are completely identical to each other, and are exactly contrary to that of the density of the waste form. These sensitivity behaviors reasonably coincides with those expected from the mathematical model of congruent leaching (Model-1). The parameters

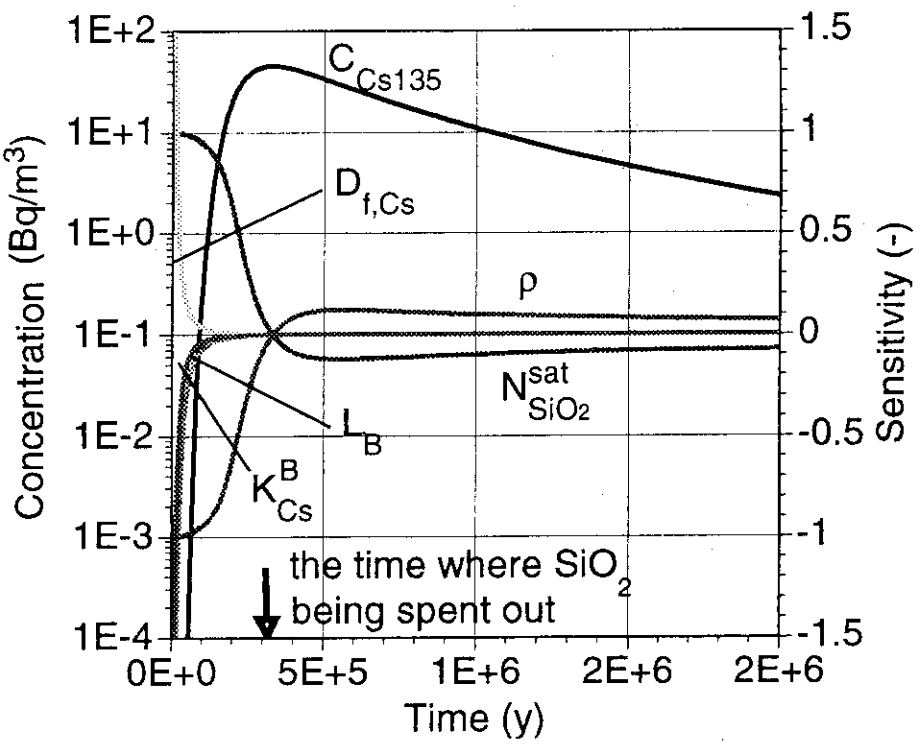

Fig. 12 Time-dependent concentration profile of ^{135}Cs at the outlet boundary of the geosphere and sensitivity coefficients of parameters (Model-1)

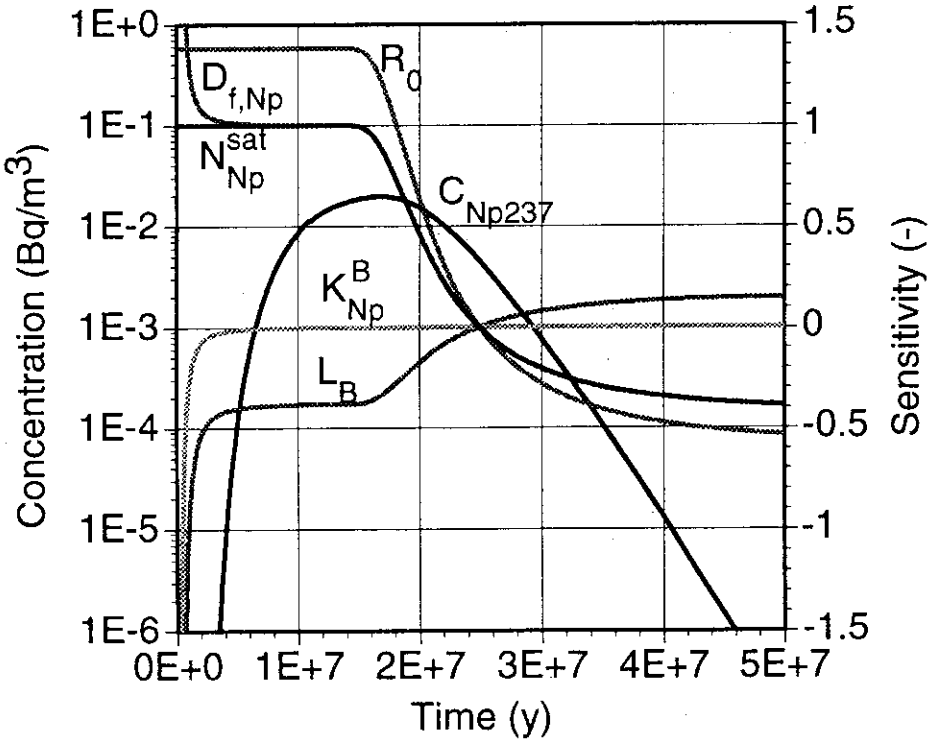

Fig. 13 Time-dependent concentration profile of ^{237}Np at the outlet boundary of the geosphere and sensitivity coefficients of parameters (Model-2)

related to the buffer, i.e., the diffusion coefficient (Cs), the retardation factor and the thickness, have a high sensitivity only at the early period of time giving an extremely low concentration of ^{135}Cs, and an insignificant sensitivity at the time appearing the peak concentration of the nuclide. This means that in the case of Model-1, these parameters do not significantly affect the peak concentration of ^{135}Cs, while slightly alter the time appearing the peak concentration. On the contrary, the parameters which control the release rate of a nuclide from the waste form, i.e., the density of the waste form, and the effective diffusion coefficient and the solubility of SiO_2, have a relatively high sensitivity coefficient during periods of the time reaching the peak concentration. After the spent out of SiO_2 in the waste form, the factor turns to reverse sign values, indicating that, e.g., in the case of the sensitivity factor having positive vales initially, the faster the release rate of the nuclide, the lower the concentration of the nuclide at the longer time, especially after the time giving peak concentration.

Figure 13 shows the time-dependent concentration profile of ^{237}Np at the outlet boundary of the geosphere and sensitivity coefficients of parameter, i.e., the diffusion coefficient and the retardation factor of ^{237}Np in the buffer material, the thickness of the buffer material, the radius of the waste form, and the solubility of ^{237}Np. Both parameters, the radius of waste form and the solubility of ^{237}Np, hold constant and positive sensitivities as long as ^{237}Np is being released from the waste form, indicating that the release flux of ^{237}Np is directly proportional to the solubility and the radius. The three parameters, the diffusion coefficient, the retardation factor and the thickness of the buffer, give a high sensitivity at initial periods of time, suggesting that these parameters shorten or lengthen the arrival time of ^{237}Np plume in a larger magnitude than the radius of the waste form and the solubility. The sensitivity of all parameters tested here reaches zero at the same point of time where a postulated stable Np without decay will give the peak concentration, while the realistic peak concentration of ^{237}Np shifts to a shorter time because of radioactive decay. The interval between the times appearing a zero sensitivity at about 2.5×10^7 y and giving an inflection point at about 1.5×10^7 y may correspond to the travel time of ^{237}Np from the source to the outlet of the geosphere.

The comparison of Figs. 12 and 13 shows that the sensitivity of the thickness of the buffer is much larger in the Model-2 simulation than that of Model-1. This might be ascribed to the difference in the boundary conditions used in solving the diffusion equations in the both models. As described before, Model-1 assumes that zero concentration of the nuclide is prescribed at the infinite spherical surface, and uses the flux boundary condition at the interface between the waste form and the buffer. On the contrary, Model-2 assumes zero concentration at the outer boundary of the buffer, and uses the concentration (solubility) boundary condition at the interface between the waste form and the buffer. Therefore, the release flux of the nuclide from the waste form more strongly depends on the thickness of the buffer in the Model-2 simulation. This difference in the conceptualizations adopted in Model-1 and Model-2 might be clearly reflected in the sensitivity behaviors observed in Figs. 12 and 13.

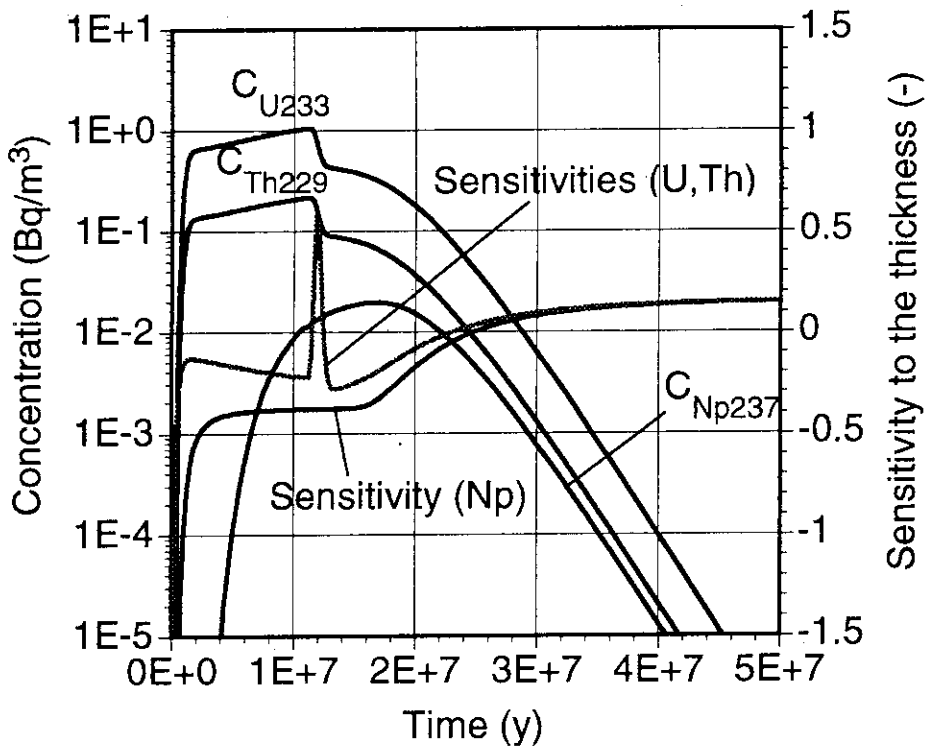

Fig. 14 Time-dependent concentration profiles of ^{237}Np, ^{233}U and ^{229}Th at the outlet boundary of the geosphere, and sensitivity coefficients of the thickness of buffer

Model-2 (solubility control) has the difficulty in dealing with nuclides, e.g., ^{129}I and ^{135}Cs, having higher solubilities than that of glass matrix, because all inventories of these nuclides are instantaneously released from the waste form, and gives unreasonably high concentrations at the boundary between the waste form and the buffer. It might be therefore said that the conceptualization adopted in this model is clearly reasonable only for nuclides having low solubilities. On the contrary, Model-1 (congruent dissolution) assumes that radionuclides incorporated in the waste form are released at the same rate as that of glass matrix, and then holds the difficulty in estimating the release flux of nuclides having significantly lower solubilities than that of glass matrix. The basic assumption used in Model-1 is therefore not rationalized especially for such the nuclides giving unreasonably high release fluxes.

Figure 14 shows the time-dependent concentration profiles of ^{237}Np, ^{233}U and ^{229}Th at the outlet boundary of the geosphere, and sensitivity coefficients of the thickness of buffer material. The concentration profiles of ^{233}U and ^{229}Th, the daughters of ^{237}Np, consists of two breakthrough curves: one is due to those decayed from the parent in the buffer and they migrate faster than the parent in both the buffer and the geosphere, and other is caused by those decayed from the parent during the migration in the geosphere (transient equilibrium). This might be reflected in the apparent discontinuity (non-differentiability) observed in the sensitivity curves of the daughters at the point intersecting the two breakthrough curves. This discontinuity might be due to rapid decreases of the concentrations in the first breakthrough curves of the daughters.

The effects of parameters on the performance measure of the disposal system, i.e., the concentration at the outlet of the geosphere, are correlated to each others. In order to examine the multi-parametric effects on the sensitivities, at the peak concentration of ^{237}Np, of the diffusion coefficient in the buffer (and also the solubility of ^{237}Np) and the thickness of the buffer were analyzed by changing the values of the thickness of the buffer and the solubility of ^{237}Np, as shown in Figs. 15a and 15b in a three-dimensional form, respectively. Here, it should be note that the sensitivity analysis method used in this report has a concept of, e.g., the sensitivity of the solubility to the peak concentration as a function of the solubility. The sensitivity behavior of the solubility is identical to that of the diffusion coefficient at the peak concentration, and then is shown in the same surface (sensitivity surface) in Fig. 15a, as also is observed in Fig. 13. In a region of high solubilities, both the diffusion coefficient and the solubility have relatively low sensitivities to the peak concentration, while they increase with decreasing of the solubility. The sensitivity of the thickness of the buffer, as shown in Fig. 15b, is significant in a region of low thickness and low solubility. This suggests from a radiological protection aspect that the thickness of buffer may be optimized to be greater than at least 0.5 m, because a significant improvement may be expected of the ability of engineered barriers reducing the peak concentration at the outlet of the geosphere. In addition to this view, one may consider, in optimizing the thickness of buffer, geochemical and mechanical stability of the buffer to realize an adequate hydraulic condition, i.e., a low hydraulic conductivity, around the waste package.

6.2 Sensitivity of the geosphere model

The geosphere models used here consists of two parts corresponding to the structure of the geosphere assumed here: one (named 3D model hereinafter) which is based on a mass transport equation with one-dimensional advection and three-dimensional dispersion deals with a homogeneous rock mass surrounding the repository, and other (called 1D model hereinafter) analyzes the transport of radionuclides in a fractured zone connecting with the biosphere assuming one-dimensional advection and dispersion.

Figure 16a shows for 3D models the time-dependent concentration profile of ^{237}Np at the outlet boundary of the geosphere and sensitivity coefficients of parameter, i.e., the dispersion lengths (α_x, α_y and α_z), the retardation factor, the migration length, the water velocity in the homogeneous rock, and the distances between the waste packages (Δx and Δy). The transverse dispersion lengths (α_x and α_y) and the distances between the packages have relatively insignificant sensitivities to the concentration of ^{237}Np under the conditions used in this analysis, while other parameters, the water velocity, the longitudinal dispersion length (α_z), the retardation factor and the migration length, show significant sensitivities. Sensitivity behaviors of these important parameters for the daughters are given in Fig. 16b and 16c, respectively. The discontinuity in the sensitivity curve is also observed for all the parameters analyzed here, corresponding to that the concentration profile consists of two different breakthrough curves, as seen in Fig. 14 for the source term model (Model-2). Except for

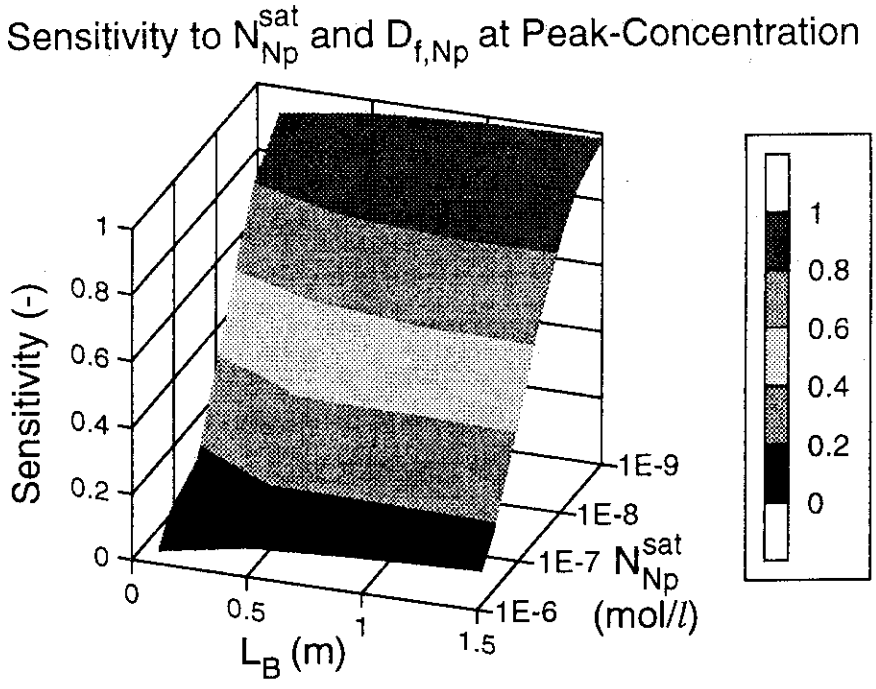

Fig. 15a Multi-parametric effects on the sensitivities, at the peak concentration of ^{237}Np, of the diffusion coefficientnin the buffer (solubility of Np) by changing the values of the thickness of the buffer and the solubility of Np

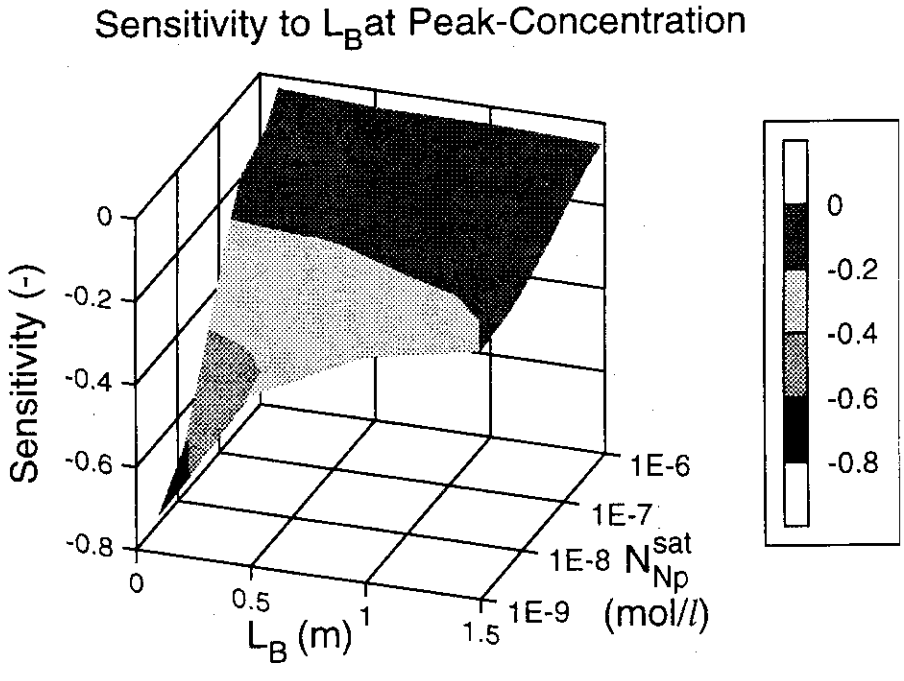

Fig. 15b Multi-parametric effects on the sensitivities, at the peak concentration of ^{237}Np, of the thickness of the buffer by changing the values of the thickness of the buffer and the solubility of Np

the retardation factors of ^{229}Th, the sensitivity behaviors of the other parameters are completely identical between the daughters. The retardation factor of ^{237}Np shows a lower sensitivity than those of ^{233}U at an early period of time, reflecting from that the daughters produced from the parent during the transport in the buffer migrate in the homogeneous rock faster than the parent, and then control the increasing behavior of the concentrations of the daughters at the outlet of the geosphere. After the time giving the peak concentration, the retardation factor of ^{237}Np tends to be dominant among the factors to determine the concentration profile, suggesting that the decrease tendency in the concentration is governed by the parent having a longer half life and a lower retardation factor than the daughters. The retardation factor of ^{229}Th has no effect on its concentration profile because of its shorter half-life than the other nuclides and the travel time in the homogeneous rock. The sensitivity of the parameters tested here at an early period of time for 3D model follows the order; migration length, water velocity, retardation factor, longitudinal dispersion length.

The sensitivity coefficients are shown in Figs. 17a, 17b and 17c for the parameters involved in 1D model (for fractured zone), i.e., the dispersion length, the migration length, the water velocity and the retardation factors, together with the concentration profiles ^{237}Np, ^{233}U and ^{229}Th, respectively. It might be said through the comparison of Figs. 12, 13, 14, 16a, 16b and 16c that these parameters tested here are significantly less sensitive to the concentrations of radionuclides at the outlet of the geosphere than the parameters related to the homogeneous rock, and slightly lower than those involved in the source term, except for the sensitivities of the retardation factors of ^{233}U and ^{229}Th to the concentration of ^{229}Th. This remarkable difference in the sensitivity behavior between the homogeneous rock and the fractured zone is ascribed to the fact that the 3D model application (homogeneous rock) used higher retardation factors and lower water velocity than those used in the 1D model application (fractured zone), resulting in higher migration velocities of nuclides in the fractured zone. This higher migration velocity might reduce the sensitivities of the parameters involved in 1D model into a lower level than those for 3D model.

As shown in Fig. 16c, the retardation factor of ^{229}Th in the homogeneous rock has no sensitivity to the concentration of ^{229}Th at the outlet of the geosphere, while that of ^{229}Th in the fractured zone is reasonably sensitive to the concentration which is governed by the transient equilibrium in the fractured zone, because of an significantly shorter half-life of ^{229}Th than ^{237}Np and ^{233}U.

It may be suggested through the sensitivity analyses conducted here that as far as engineered and natural barriers concern, the migration length and the water velocity in the homogeneous rock are of relative importance among the parameters related to the both barriers to control the performance of the system consisting of the both barriers.

In order to determine the relationship between the sensitivities of the four important parameters (the water velocity, the migration length, the longitudinal dispersion length and the retardation factor), sensitivity analyses at the peak concentration were also implemented by changing values of the migration length and the water velocity simultaneously, as shown in Figs. 18a, 18b, 18c and 18d in a three-dimensional form, respectively. As shown in Fig. 18a, the water velocity shows extremely high sensitivities especially in a region of low velocities and long migration lengths, and

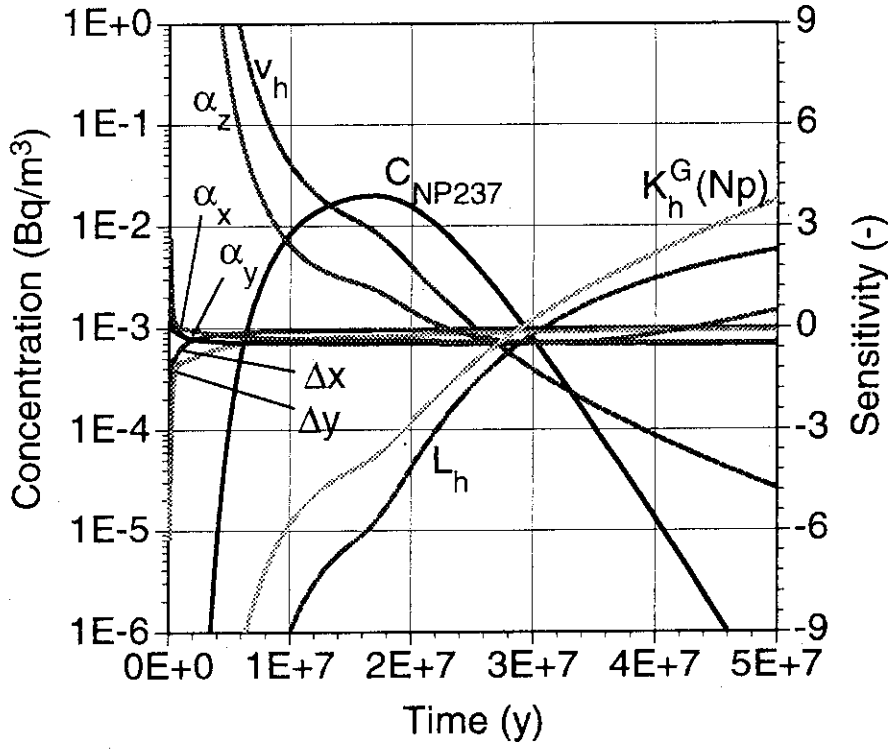

Fig. 16a Time-dependent concentration profile of ^{237}Np at the outlet boundary of the geosphere and sensitivity coefficients of parameters (3D model)

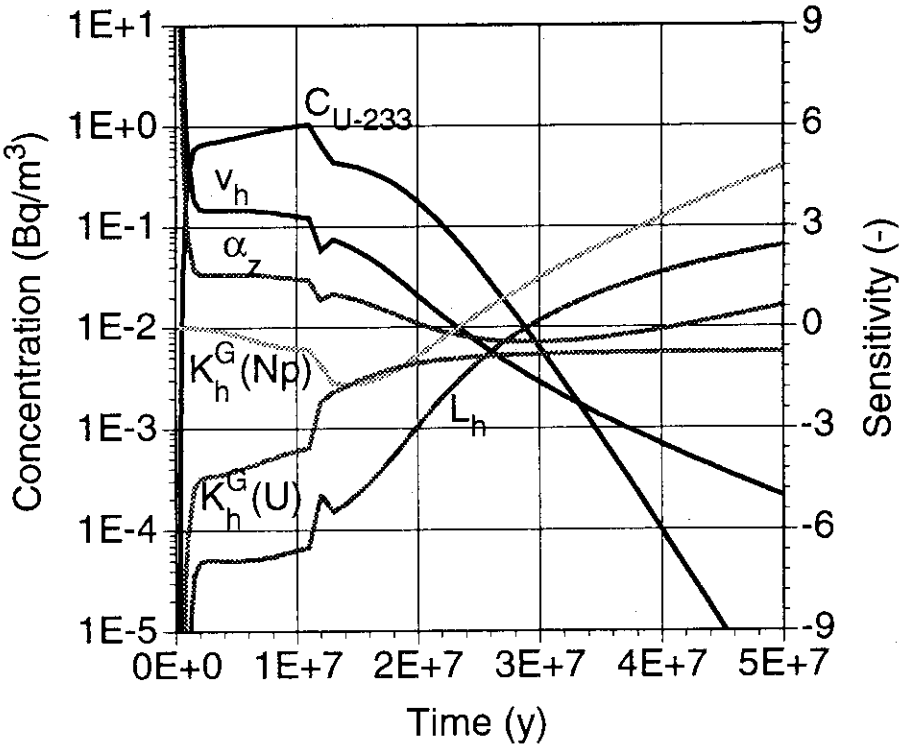

Fig. 16b Time-dependent concentration profile of ^{233}U at the outlet boundary of the geosphere and sensitivity coefficients of parameters (3D model)

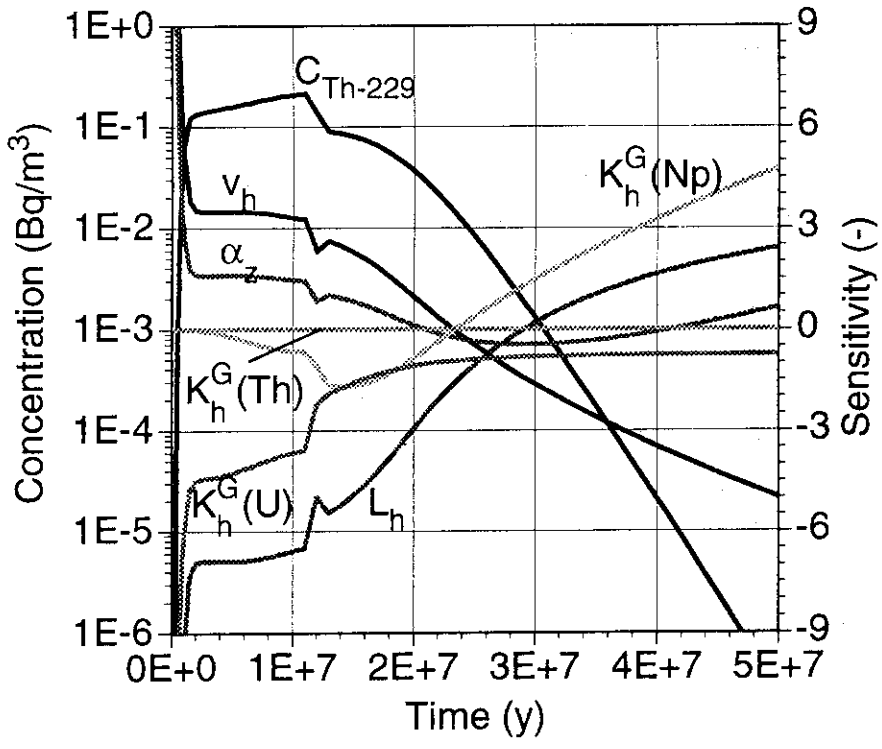

Fig. 16c Time-dependent concentration profile of ^{229}Th at the outlet boundary of the geosphere and sensitivity coefficients of parameters (3D model)

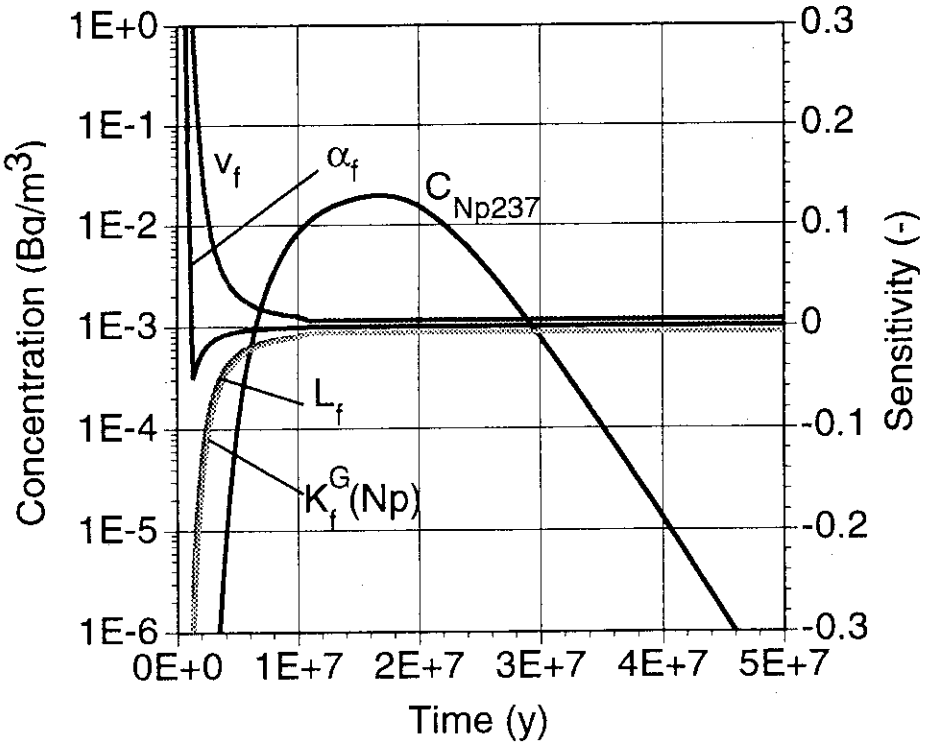

Fig. 17a Time-dependent concentration profile of ^{237}Np at the outlet boundary of the geosphere and sensitivity coefficients of parameters (1D model)

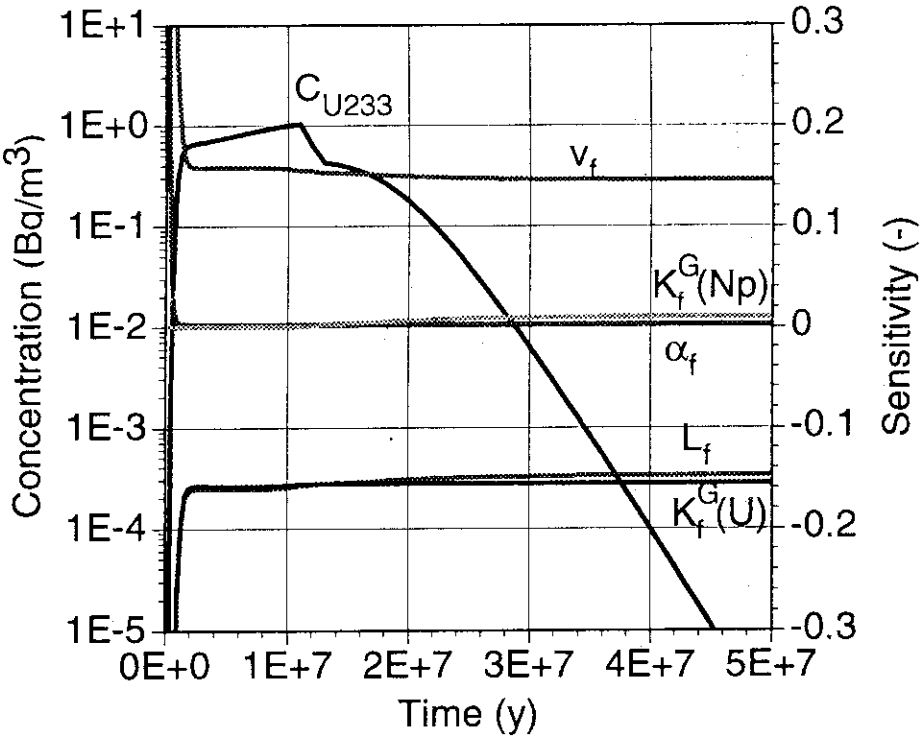

Fig. 17b Time-dependent concentration profile of ^{233}U at the outlet boundary of the geosphere and sensitivity coefficients of parameters (1D model)

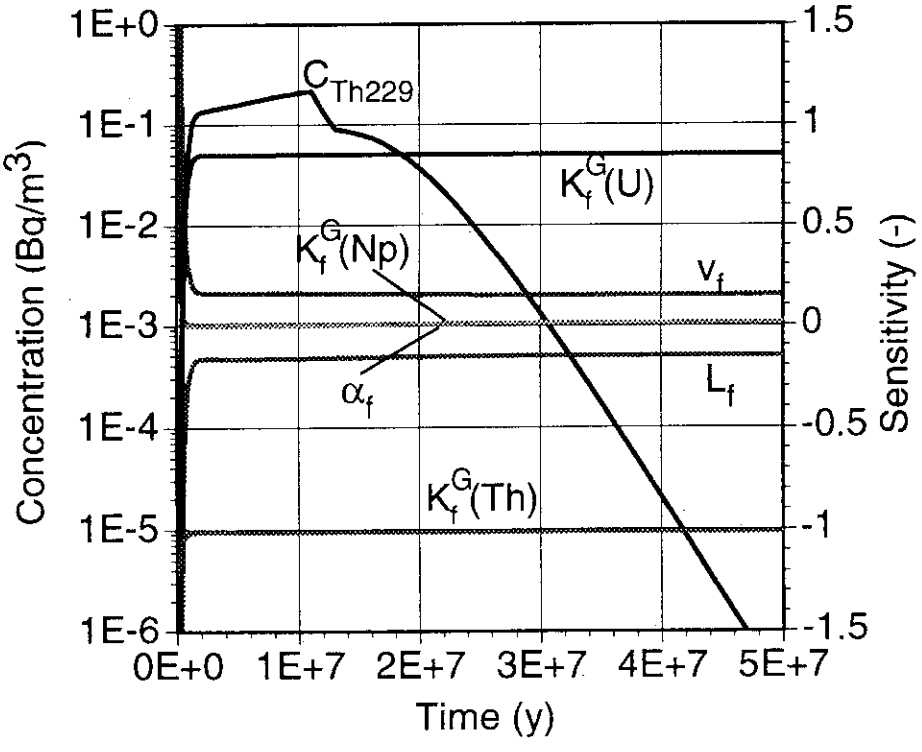

Fig. 17c Time-dependent concentration profile of ^{229}Th at the outlet boundary of the geosphere and sensitivity coefficients of parameters (1D model)

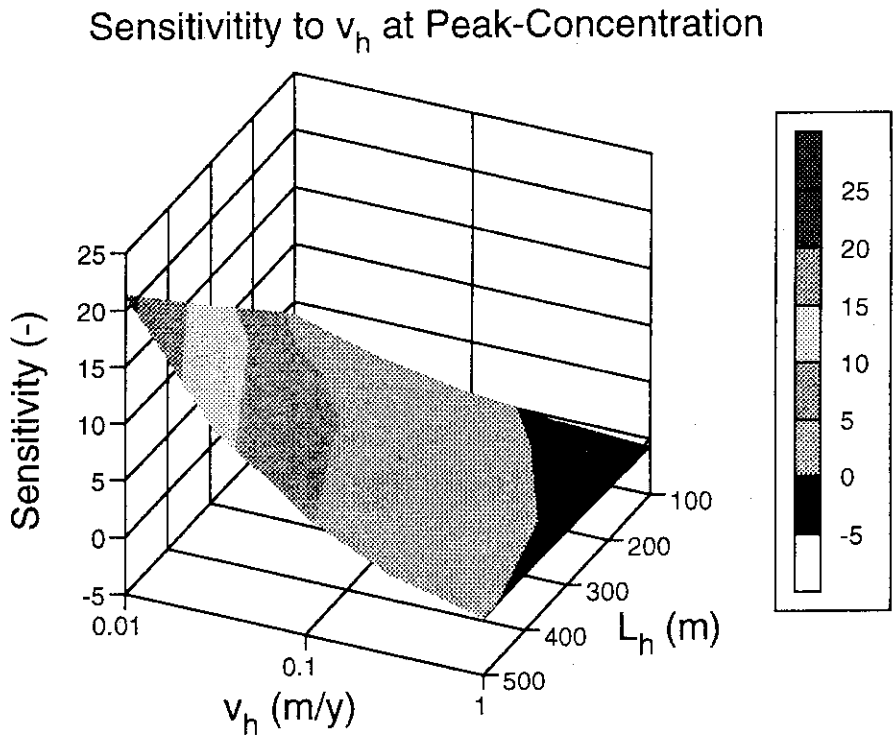

Fig. 18a Multi-parametric effects on the sensitivities, at the peak concentration of ^{237}Np, of the velocity in the homogeneous rock by changing the values of the velocity and migration length

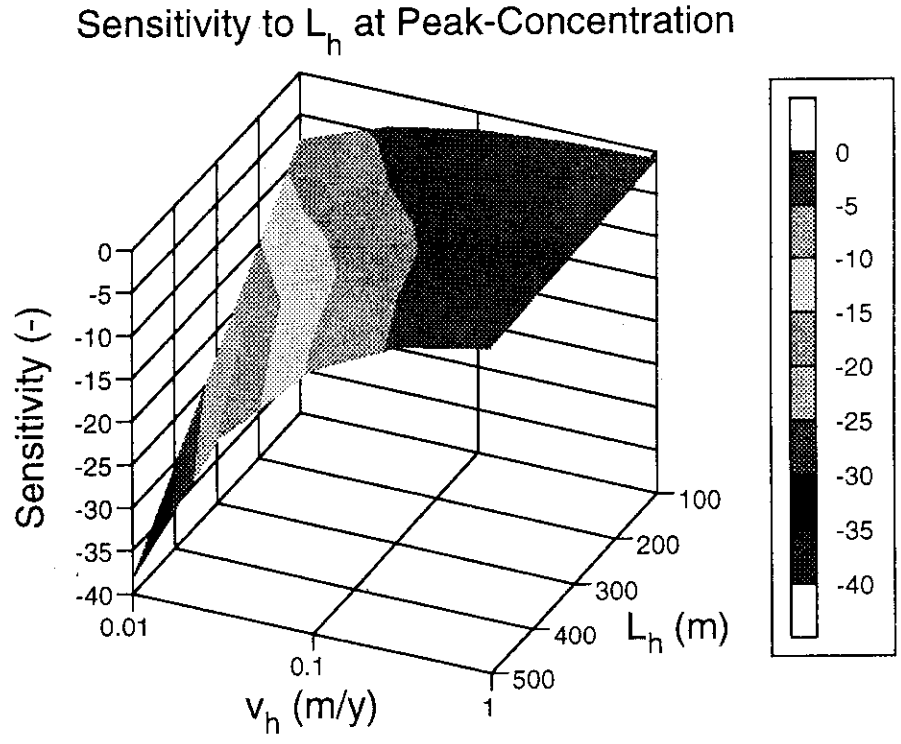

Fig. 18b Multi-parametric effects on the sensitivities, at the peak concentration of ^{237}Np, of the migration length of the homogeneous rock by changing the values of the velocity and migration length

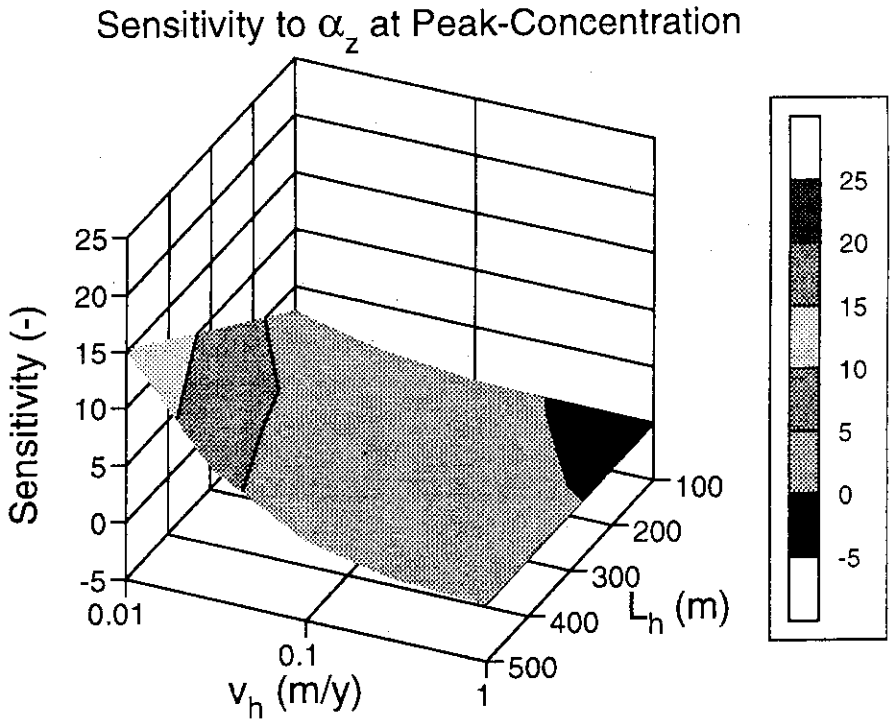

Fig. 18c Multi-parametric effects on the sensitivities, at the peak concentration of ^{237}Np, of the longitudinal dispersion length in the homogeneous rock by changing the values of the velocity and migration length

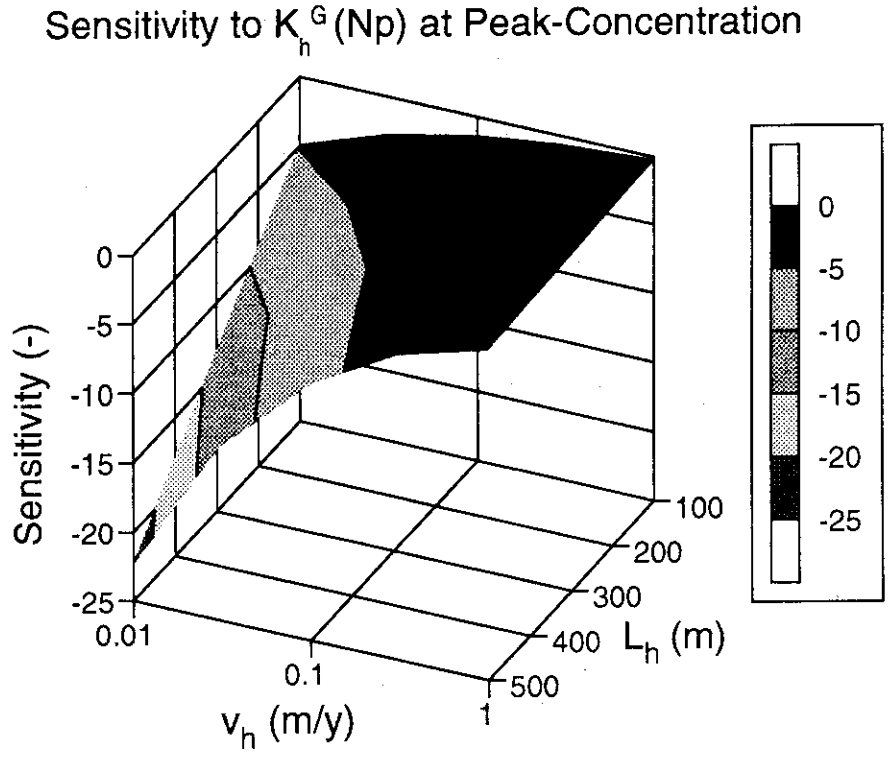

Fig. 18d Multi-parametric effects on the sensitivities, at the peak concentration of ^{237}Np, of the retardation factor (Np) in the homogeneous rock by changing the values of the velocity and migration length

even in a short migration length region it still keeps a relatively high sensitivity. In a region of high velocities and short migration lengths, the sensitivity turns to a negative value corresponding to that a dispersion term (expressed by the product of the velocity and the dispersion length) which has the effect to reduce the concentration becomes more visible than the other regions (the same tendency can be seen in the sensitivities of longitudinal dispersion length as shown in Fig. 18c). As given in Fig. 18b, the migration length has also remarkably high sensitivities in a negative value, especially in a region of low water velocities and long migration lengths, while it gives insignificant sensitivities at a high water velocity region (1 m/y). In this lower sensitivity surface, the sensitivity coefficient of the migration length changes from -0.8 to -2.0 with the increment of the migration length from 100 to 500 m, indicating that a holding of large rock mass with high groundwater velocity in the homogeneous rock (~1 m/y) might be not necessary effective to heighten the performance of the disposal system (the same tendency can be seen in the sensitivities of retardation factor as shown in Fig. 18d). Therefore, it might be important to select a geologic formation, as a host rock, having a low water velocity field. At the same time, it must be recognized, however, that the water velocity in the formation to be analyzed should be determined as possible as precisely especially for a low water velocity region, because the output of safety analyses might depend sharply on the velocity in such a region. Otherwise the result of the assessment which is based on site specific characterization data might inevitably have a great deal of uncertainties.

7. SAFETY ANALYSIS

In order to provide basic information for detailed safety assessment which will be implemented in a future on a specific site based on characterization data, a generic safety analysis was conducted by using data reported by SKI[5], SKB[3], TVO[6], PAGIS[10], NAGRA[4]. The scenario considered here is based on a normal groundwater migration scenario, assuming that values of the parameters used are time invariant, in other words basic characteristics of engineered and natural barriers, and the biosphere remain unchanged throughout the periods to be subjected to this analysis.

The reference disposal system assumed here is given in Fig. 11. There are 4000 holes excavated in a homogeneous low-permeable rock, in each of which three containers are emplaced vertically with surrounding buffer materials. Radionuclide fluxes from a single hole are assumed to be three times of the fluxes from one container, in other words a single hole containing three containers is regarded as a point source in this analysis. Totally 12000 containers are thus disposed of in the reference repository, corresponding to the total inventory of HLW generating from the reprocessing of 16000 MTU of the spent fuels. The migration lengths in the homogeneous rock to the fractured zone are 110 m at the right hand side and 490 m at the left hand side of repository as shown in Fig. 11 and the length in the fractured zone to the biosphere is 1000 m.

We assume here that the whole contaminated groundwater migrates through the geosphere and finally enters into an adjacent aquifer which is defined as one of the compartments involved in the biosphere. Water in the aquifer is assumed to be consumed by a local individual as drinking

even in a short migration length region it still keeps a relatively high sensitivity. In a region of high velocities and short migration lengths, the sensitivity turns to a negative value corresponding to that a dispersion term (expressed by the product of the velocity and the dispersion length) which has the effect to reduce the concentration becomes more visible than the other regions (the same tendency can be seen in the sensitivities of longitudinal dispersion length as shown in Fig. 18c). As given in Fig. 18b, the migration length has also remarkably high sensitivities in a negative value, especially in a region of low water velocities and long migration lengths, while it gives insignificant sensitivities at a high water velocity region (1 m/y). In this lower sensitivity surface, the sensitivity coefficient of the migration length changes from -0.8 to -2.0 with the increment of the migration length from 100 to 500 m, indicating that a holding of large rock mass with high groundwater velocity in the homogeneous rock (~1 m/y) might be not necessary effective to heighten the performance of the disposal system (the same tendency can be seen in the sensitivities of retardation factor as shown in Fig. 18d). Therefore, it might be important to select a geologic formation, as a host rock, having a low water velocity field. At the same time, it must be recognized, however, that the water velocity in the formation to be analyzed should be determined as possible as precisely especially for a low water velocity region, because the output of safety analyses might depend sharply on the velocity in such a region. Otherwise the result of the assessment which is based on site specific characterization data might inevitably have a great deal of uncertainties.

7. SAFETY ANALYSIS

In order to provide basic information for detailed safety assessment which will be implemented in a future on a specific site based on characterization data, a generic safety analysis was conducted by using data reported by SKI[5], SKB[3], TVO[6], PAGIS[10], NAGRA[4]. The scenario considered here is based on a normal groundwater migration scenario, assuming that values of the parameters used are time invariant, in other words basic characteristics of engineered and natural barriers, and the biosphere remain unchanged throughout the periods to be subjected to this analysis.

The reference disposal system assumed here is given in Fig. 11. There are 4000 holes excavated in a homogeneous low-permeable rock, in each of which three containers are emplaced vertically with surrounding buffer materials. Radionuclide fluxes from a single hole are assumed to be three times of the fluxes from one container, in other words a single hole containing three containers is regarded as a point source in this analysis. Totally 12000 containers are thus disposed of in the reference repository, corresponding to the total inventory of HLW generating from the reprocessing of 16000 MTU of the spent fuels. The migration lengths in the homogeneous rock to the fractured zone are 110 m at the right hand side and 490 m at the left hand side of repository as shown in Fig. 11 and the length in the fractured zone to the biosphere is 1000 m.

We assume here that the whole contaminated groundwater migrates through the geosphere and finally enters into an adjacent aquifer which is defined as one of the compartments involved in the biosphere. Water in the aquifer is assumed to be consumed by a local individual as drinking

water with the dilution factor of 10^{-3}, which can be estimated by the ratio of flow rate (Darcy velocity) in the homogeneous rock ($3x10^{-4}$ m/y) and that in the aquifer (10^{-3} m/day[27]). Possible other exposure pathways to the individual are neglected in this analysis, because it was evident through a preliminary analysis that the pathway used here gives a predominant radiological consequence among possible pathways. Individual committed effective dose equivalent (Sv) from the drinking water pathway was thus calculated using the internal dose conversion factors recommended by the ICRP publications 26[28], 30[29] and 48[30].

In order to estimate a range of potential radiological consequences associated with geologic disposal of HLW, two kinds of analyses were performed: one (conservative case) intends to analyze the upper bound of the consequences by using the conservative values of geochemical parameters listed in Table 5, and other (realistic case) aims at the evaluation of likely consequences to occur by using the realistic values of those. The doses thus evaluated for the conservative case are shown in Fig. 19a for fission products (FP) and Fig. 19b for α-nuclides, respectively.

In the conservative case, the maximum dose contributed from FP components amounts to 3.7×10^{-7} Sv at 1.5×10^{5} y after disposal, and is contributed mainly from ^{99}Tc (about 89% of the maximum dose) and ^{79}Se (about 11%). The dose curve of ^{99}Tc increases gradually after the initiation of release of the nuclide from the container at 1000 y after disposal, reaches the maximum 3.2×10^{-7} Sv at 2.0×10^{4} y, and decreases rapidly at 5.3×10^{5} y with the exhaust of its inventory in the waste form. This rapid decrease in the dose is also reflected from a low value, 1, of the retardation factor of ^{99}Tc. However, this type of decrease is not observed in the dose curve of ^{129}I, nevertheless the same value, 1, of the retardation factor was used for them. The difference in the dose curves between them might be ascribed to the fact that ^{99}Tc was analyzed with Model-2 while ^{129}I with Model-1. As seen from Fig. 19b which shows the dose curves coming from α-nuclides, the maximum dose amounts to about 2.2×10^{-7} Sv at 2.2×10^{6} y which is longer than that for FP components, reflecting from the retardation factors used for α-nuclides being higher than those for FP components. The total dose curve given in Fig. 19b is mainly governed by ^{229}Th (about 38% of the maximum dose), ^{231}Pa (about 20%), ^{233}U (about 14%) and ^{226}Ra (about 12%), in other words the dose of this case is controlled by the daughters of ^{237}Np, ^{235}U and ^{238}U.

The total maximum dose due to both FP components and α-nuclides is 4.5×10^{-7} Sv at 5.2×10^{5} y after disposal, which is predominantly controlled by ^{99}Tc (about 72% of the total maximum dose) with slight contributions from ^{229}Th (about 8.0%), ^{135}Cs (about 7.5%), ^{231}Pa (about 3.8%), and ^{233}U (about 3.1%).

Figures 20a and 20b show the individual doses from FP components and α-nuclides for the realistic case, respectively. The doses due to FP components are significantly higher than those from α-nuclides, reflecting from values of the retardation factors for α-nuclides being 10 times or more higher than those in the conservative case. The total dose curve consists of three peaks coming from ^{129}I, ^{79}Se and ^{135}Cs, respectively. The total maximum dose, 2.6×10^{-8} Sv, observed at 3×10^{4} y is dominated only by ^{129}I, this is because that the value of the retardation factor was the same between the both cases. As mentioned before, this analysis arbitrarily assumes that the vitrified

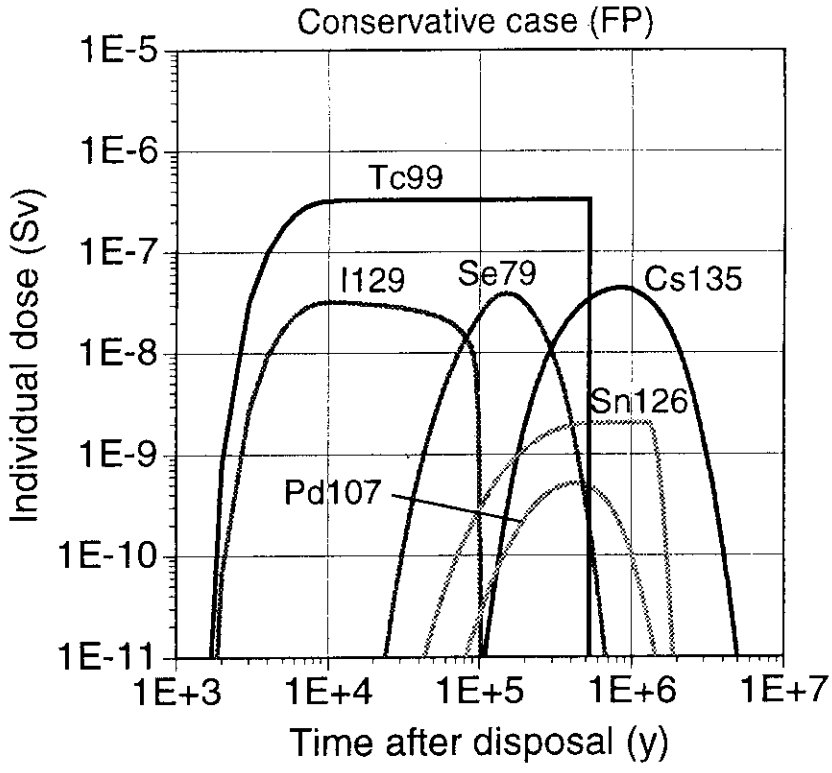

Fig. 19a Individual dose from fission products of disposed HLW for conservative case

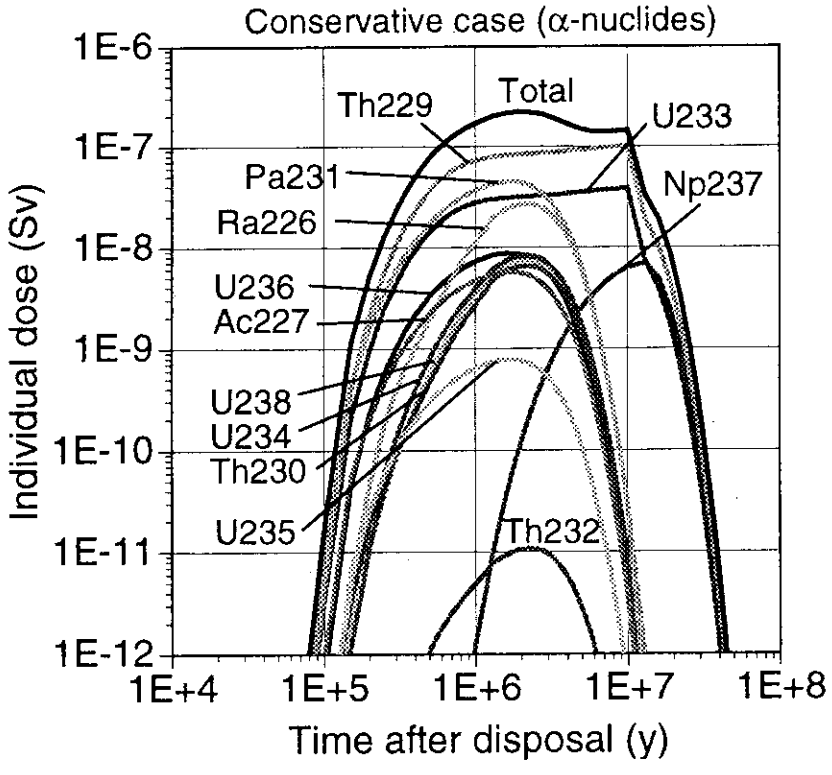

Fig. 19b Individual dose from α-nuclides of disposed HLW for conservative case

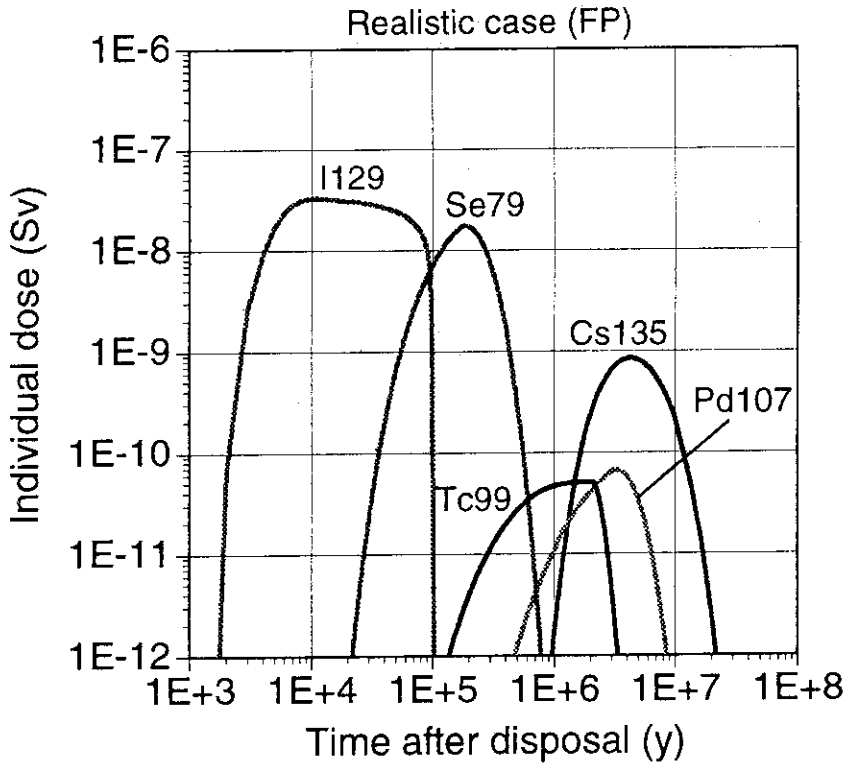

Fig. 20a Individual dose from fission products of disposed HLW for realistic case

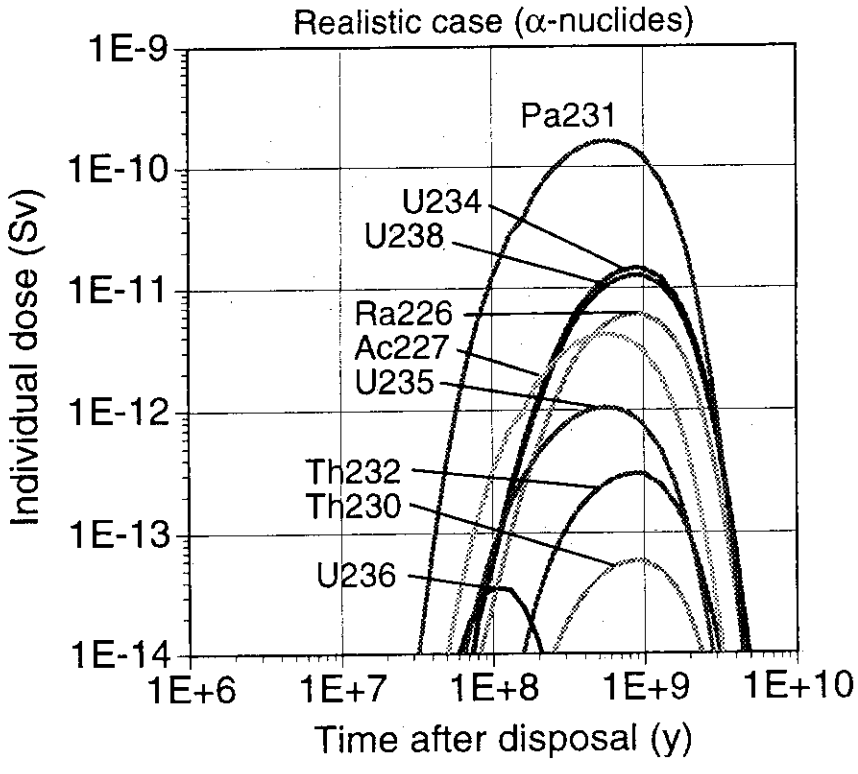

Fig. 20b Individual dose from α-nuclides of disposed HLW for realistic case

waste form contains 1% of iodines generated in the fuel during irradiation. However, if it is confident to eliminate all of the iodines during reprocessing and vitrification, one may disregard the doses from ^{129}I. In such the case, the total maximum dose, 1.7×10^{-8} Sv, observed at 1.9×10^{5} y is controlled mainly by ^{79}Se. Taking into account that ^{129}I might offer significant radiological consequences, while depending on its concentration, especially for long-term assessment, it might be worthwhile to note that even if the iodines in the fuel have been successfully eliminated from HLW, the resulting wastes containing the iodines should finally be disposed of in safe manners.

In conclusion, the committed dose equivalent evaluated here, even in the conservative case, is the order of 10^{-7} Sv being lower than the dose limit 1mSv, and thus geologic disposal of HLW may be feasible if the disposal conditions assumed here remain unchanged throughout the periods assessed here. The inventory of radionuclides in HLW, which was used in the analysis, is still less than that is possibly assumed to be generated in Japan. However, even if the inventory increases 10 times that used here, the resulting dose may be less than 10 times of the dose evaluated here, taking into account that the dose increases with the increasing of inventory while not linearly because the source term models used here deal with the steric configuration of point sources in the repository. The inventory also increases in proportion to the burnup of the fuel as shown in Appendix-1. This study assumes 27,500 MWD as the averaged burnup. If one may assume the burnup as 40,000 MWD, the numbers of atoms of major FP components increase 1.4 ~ 1.8 times those used here, and 0.5 ~ 14 times for major α-nuclides. Among α-nuclides, the inventory of Cm increases significantly, even so the resulting amount of Cm is still not enough high to cause noticeable doses, and the remarkable effect on the dose of an increase in the burnup was observed for ^{241}Pu, ^{237}Np and ^{235}U. In the case of 40,000 MWD, the maximum dose is estimated to be 6.3×10^{-7} Sv for the conservative case, and 4.0×10^{-8} Sv (^{129}I) or 2.4×10^{-8} Sv (^{79}Se) for realistic case, respectively. The doses from α-nuclides differ significantly between the conservative and realistic cases, corresponding to the differences in geochemical conditions assumed for them. This means while α-nuclides do not give predominant contributions to the total dose that extensive geochemical studies on α-nuclides in a realistic conditions might be essential to reduce the uncertainty to be involved in the results of safety analyses.

8. CONCLUSIONS

8.1 Methodologies of the safety assessment and automated sensitivity analysis

The methodologies of the safety assessment for HLW disposal and the automated sensitivity analysis based on the DA-method have been described. The safety assessment methodology is based on the normal evolution scenario assuming that the disposal system is not affected by probabilistic events. The computer code system GSRW is structured by non site-specific models simulating the release of radionuclides from the engineered barriers, the transport of radionuclides in and through the geosphere and biosphere, and radiation exposures of public.

waste form contains 1% of iodines generated in the fuel during irradiation. However, if it is confident to eliminate all of the iodines during reprocessing and vitrification, one may disregard the doses from ^{129}I. In such the case, the total maximum dose, 1.7×10^{-8} Sv, observed at 1.9×10^{5} y is controlled mainly by ^{79}Se. Taking into account that ^{129}I might offer significant radiological consequences, while depending on its concentration, especially for long-term assessment, it might be worthwhile to note that even if the iodines in the fuel have been successfully eliminated from HLW, the resulting wastes containing the iodines should finally be disposed of in safe manners.

In conclusion, the committed dose equivalent evaluated here, even in the conservative case, is the order of 10^{-7} Sv being lower than the dose limit 1mSv, and thus geologic disposal of HLW may be feasible if the disposal conditions assumed here remain unchanged throughout the periods assessed here. The inventory of radionuclides in HLW, which was used in the analysis, is still less than that is possibly assumed to be generated in Japan. However, even if the inventory increases 10 times that used here, the resulting dose may be less than 10 times of the dose evaluated here, taking into account that the dose increases with the increasing of inventory while not linearly because the source term models used here deal with the steric configuration of point sources in the repository. The inventory also increases in proportion to the burnup of the fuel as shown in Appendix-1. This study assumes 27,500 MWD as the averaged burnup. If one may assume the burnup as 40,000 MWD, the numbers of atoms of major FP components increase 1.4 ~ 1.8 times those used here, and 0.5 ~ 14 times for major α-nuclides. Among α-nuclides, the inventory of Cm increases significantly, even so the resulting amount of Cm is still not enough high to cause noticeable doses, and the remarkable effect on the dose of an increase in the burnup was observed for ^{241}Pu, ^{237}Np and ^{235}U. In the case of 40,000 MWD, the maximum dose is estimated to be 6.3×10^{-7} Sv for the conservative case, and 4.0×10^{-8} Sv (^{129}I) or 2.4×10^{-8} Sv (^{79}Se) for realistic case, respectively. The doses from α-nuclides differ significantly between the conservative and realistic cases, corresponding to the differences in geochemical conditions assumed for them. This means while α-nuclides do not give predominant contributions to the total dose that extensive geochemical studies on α-nuclides in a realistic conditions might be essential to reduce the uncertainty to be involved in the results of safety analyses.

8. CONCLUSIONS

8.1 Methodologies of the safety assessment and automated sensitivity analysis

The methodologies of the safety assessment for HLW disposal and the automated sensitivity analysis based on the DA-method have been described. The safety assessment methodology is based on the normal evolution scenario assuming that the disposal system is not affected by probabilistic events. The computer code system GSRW is structured by non site-specific models simulating the release of radionuclides from the engineered barriers, the transport of radionuclides in and through the geosphere and biosphere, and radiation exposures of public.

We have developed the automated procedure for performing large-scale sensitivity studies based on the use of computer tools. The procedure is composed of the preprocessor PRESANA and the FORTRAN precompiler SANA. The PRESANA and SANA codes have been applied to the simple program, and the sensitivity results have been verified using the exact solution. The results shows that this procedure is accurate and useful tool to perform the sensitivity analysis.

8.2 Sensitivity Analysis

The following remarks might be derived through sensitivity analyses on the transport behavior of the ^{237}Np decay chain and ^{135}Cs in engineered and natural barriers, by applying the sensitivity analysis methodology (DA method) to GSRW code.

(1) The sensitivity analysis methodology applied here allows the quantitative evaluation of a relative importance of various parameters which potentially affect the safety of geologic disposal system. The results of analyses clearly indicate that parameters related to a homogeneous rock surrounding a disposal facility have higher sensitivities to the performance measure analyzed here than those of a fractured zone and engineered barriers. Among the parameters involved in the geosphere model used for a homogeneous rock, the migration length, the retardation factor, the water velocity, and the longitudinal dispersivity hold a high relative importance to govern the safety of the disposal system.

(2) The methodology permits sensitivity analyses of a single parameter with changing values of other parameters simultaneously, and thus gives quantitative information on the interrelationship between the parameters. For example, the parameters for engineered barriers are generally insensitive to the output, while they are somewhat sensitive to the output only in a case of the low solubility condition.

(3) The methodology provides technical information which might be basis for the optimization of design of the disposal facility, e.g., the thickness of buffer material, and for the selection of the disposal site, e.g., the water velocity, the geochemical conditions, the migration length and so on.

8.3 Safety Analysis

Safety analyses were performed using GSRW code on the reference disposal system which involve HLW in amounts corresponding to 16,000 MTU of spent fuels. The committed effective dose equivalent due to the exposure pathway ingesting drinking water was calculated using both the conservative and realistic values of geochemical parameters. The conclusions obtained are as follows:

(1) In the conservative case, the total dose curve consists mainly of the contributions from ^{99}Tc, ^{229}Th, ^{231}Pa, ^{135}Cs, ^{79}Se and ^{129}I, and the maximum dose, 4.5×10^{-7}Sv appeared at 5.2×10^{5} y, is mainly controlled by ^{99}Tc (about 72% of the total maximum dose) with slight contributions from ^{229}Th (about 8.0%), ^{135}Cs (about 7.5%), ^{231}Pa (about 3.8%), and ^{233}U (about 3.1%).

(2) In the realistic case, the total dose curve is drawn mainly by the contributions from ^{129}I, ^{79}Se and ^{135}Cs, and the total maximum dose, 2.6×10^{-8} Sv observed at 3×10^{4} y, is dominated only by ^{129}I. If one can assume that HLW contains insignificant levels of radioactive iodines, the total maximum dose, 1.7×10^{-8} Sv, observed at 1.9×10^{5} y is controlled mainly by ^{79}Se.

(3) This study assumes 27,500 MWD as the averaged burnup. If one may assume the burnup as 40,000 MWD, the dose equivalent increases to 6.3×10^{-7} Sv for the conservative case, and to 4.0×10^{-8} Sv (^{129}I) or 2.4×10^{-8} Sv (^{79}Se) for the realistic case, respectively.

(4) In both the conservative and realistic cases, the committed dose equivalent evaluated here is the order of 10^{-7} Sv, and thus geologic disposal of HLW may be feasible if the disposal conditions assumed here remain unchanged throughout the periods assessed here.

8.4 Limitations in this study

There might be the following limitations in understanding of the results of this study.

(1) The safety assessment methodology used here is completely based on a deterministic approach, and the can deal with only a normal scenario. The safety of the disposal system should be assessed with respect to all of possible scenarios that potentially affect the performance of the system, including probabilistic scenarios such as disruptive scenarios and human intrusion scenarios. However, probabilistic scenarios are generally dealt with a probabilistic approach taking into account of the probability of scenarios, and then consequences of such scenarios might be expressed by a term of risk.

(2) It was assumed in this study that conditions of the environment involving the geosphere and the biosphere remain unchanged throughout the period assessed. It might be expected, however, that glaciation is periodically repeated every 10^{4} years, resulting in changes of environmental conditions such as sea level. The change in sea level affects a global hydrodynamics, but its effect on a local hydrodynamics, especially that in a deep geologic formation, is not certain. This is because the effect depends on the mode of connection between a global water circulation and a local on, and is thus site specific in nature.

REFERENCES

(1) Kimura, H., et al. : A Generic Safety Assessment Code for Geologic Disposal of Radioactive Waste : GSRW Computer Code User's Manual, JAERI-M-92-161, (1992).

(2) SKB/KBS, Swedish Nuclear Fuel Supply Co. : KBS-3, "Final Storage of Spent Nuclear Fuel", (1983).

(3) SKB, Swedish Nuclear Fuel and Waste Management Co. : SKB91, "Final disposal of spent nuclear fuel. Importance of the bedrock for safety", SKB Technical Report TR 92-20 (1992).

(4) NAGRA : Project Gewahr 1985, "Feasibility studies and Safety analysis", NGB 85-09 (1985).

(5) SKI, Swedish Nuclear Power Inspectorate, : SKI Project-90, SKI Technical Report 91:23 (1991).

(6) Vieno, T., et al. : TVO-92 Safety Analysis of Spent Fuel Disposal, YJT-92-33E (1992).

(7) PNC, Power Reactor and Nuclear Fuel Development Co., : Technical Report on Research and Development of Geological Disposal for High Level Radioactive Wastes -H3 (in Japanese), PNC TN 1410 92-081 (1992).

(8) Marivoet, J. :UPDATING 1990, BLG 634, SCK/CEN, (1992).

(9) Berz, M., : Differential Algebraic Description of Beam Dynamics to Very High Orders, SSC-152, LBL (1988).

(10) CEC, Commission of the European Communities : PAGIS, "Performance Assessment of Geologic Isolation Systems for Radioactive Waste", (1988).

(11) Croff, A. G., : User's Manual for the ORIGEN2 Computer Code, ORNL/TM-7175, ORNL, (1980).

(12) Chambre' , P. L., et al. : Analytical Performance Models for Geologic Repositories, LBL-14842, UC-70, (1982).

(13) Pigford, T.H., Chambre' , P. L., : Near-field Mass Transfer in Geologic Disposal Systems : A Review, Mat. Res. Soc. Symp. Proc. Vol. 112, pp125-141, (1988).

(14) Pigford, T. H., et al., : Migration of Radionuclides Through Sorbing Media, Analytical Solutions II, LBL-11616, UC-70, (1980).

(15) Harada, M., et al., : Migration of Radionuclides Through Sorbing Media, Analytical Solutions I, LBL-10500, UC-70, (1980).

(16) Boulton, G. S. : Time-dependent Modelling of Environmental Change: The Effect of Quaternary Glaciations, Safety Assessment of Radioactive Waste Repositories, OECD/NEA, pp363-375, (1989).

(17) Oblow, E. M. : An Automated Procedure for Sensitivity Analysis Using Computer Calculus, ORNL/TM-8776, ORNL (1983).

(18) Oblow, E. M. : GRESS Gradient-Enhanced Software System, Version D User's Guide, ORNL/TM-9658, ORNL, (1985).

(19) Oblow, E. M., et al., : Sensitivity Analysis Using Computer Calculus : A Nuclear Waste Isolation Application, Nucl. Sci. Eng. 94, pp46 -65, (1986).

(20) Andersson, K. : SKI Project-90, "Chemical Data", SKI Technical Report 91:21 (1991).

(21) Sato, K. and K. Watanabe : On the Relationship between Rock Variability and Some Crack Properties (in Japanese), Annual Report of Hydro-science and Geotechnology Lab., Vol. 4, pp. 104~118 (1978).

(22) Wolery, T. J. : Calculation of Chemical Equilibrium between Aqueous Solution and Minerals : EQ3/6 Software Package, UCRL-52658 (1979).

(23) Parkhurst, D. L. et al. : PHREEQE - A Computer Program for Geochemical Calculations, U. S. Geol. Surv. Water-Resour. Invest. 80-96, NTIS PB 81-167801 (1980).

(24) Eikenberg, J. : On the Problem of Silica Solubility at High pH, PSI-Bericht Nr. 74 (1990).

(25) Gmelin, L. : Gmelin Handbuch der Anorganischen Chemie, Protactinium, Springer-Verlag, (1977).

(26) Groundwater Handbook (in Japanese), Kensetsusangyo-chousakai Co., (1979).

(27) Abelin, H., et al. : 3-D Migration experiment - Report 3 Part I, Performed experiments, results and evaluation., Stripa Project, TR 87-21 (1987).

(28) ICRP : Recommendations of the International Commission on Radiological Protection, ICRP Pub. 26, Pergamon Press, (1977).

(29) ICRP : Limits for Intakes of Radionuclides by Workers, ICRP Pub. 30, Pergamon Press, (1982).

(30) ICRP : The Metabolism of Plutonium and Related Elements, ICRP Pub. 48, Pergamon Press, (1986).

Appendix The inventory of radionuclides in the spent fuel with various burnup rates (No. of atoms/MTU, BWR)

Nuclides	Burnup (MWD)					
	20000	25000	27500	30000	35000	40000
Se-79	2.76E+22	3.41E+22	3.72E+22	4.04E+22	4.65E+22	5.25E+22
Tc-99	2.98E+24	3.66E+24	3.99E+24	4.32E+24	4.93E+24	5.51E+24
I-129	4.77E+23	6.08E+23	6.74E+23	7.40E+23	8.73E+23	1.00E+24
Cs-135	1.07E+24	1.33E+24	1.45E+24	1.58E+24	1.84E+24	2.10E+24
Pu-240	3.61E+24	4.58E+24	5.02E+24	5.39E+24	5.98E+24	6.38E+24
U-236	6.56E+24	7.49E+24	7.86E+24	8.17E+24	8.64E+24	8.92E+24
Cm-245	1.31E+20	4.98E+20	8.73E+20	1.44E+21	3.43E+21	7.04E+21
Pu-241	1.70E+24	2.25E+24	2.51E+24	2.77E+24	3.27E+24	3.68E+24
Am-241	4.28E+22	6.63E+22	7.81E+22	8.95E+22	1.11E+23	1.30E+23
Np-237	4.95E+23	6.94E+23	8.00E+23	9.10E+23	1.13E+24	1.34E+24
Cm-246	8.11E+18	4.15E+19	8.31E+19	1.56E+20	4.67E+20	1.18E+21
Am-242m	7.23E+20	1.22E+21	1.47E+21	1.72E+21	2.21E+21	2.64E+21
Pu-242	2.67E+23	4.75E+23	6.00E+23	7.38E+23	1.05E+24	1.40E+24
Pu-238	9.35E+22	1.69E+23	2.17E+23	2.73E+23	4.06E+23	5.62E+23
U-238	2.16E+27	2.15E+27	2.15E+27	2.14E+27	2.13E+27	2.12E+27
Am-243	3.74E+22	8.64E+22	1.23E+23	1.68E+23	2.86E+23	4.43E+23
Pu-239	9.19E+24	9.67E+24	9.85E+24	9.99E+24	1.01E+25	1.02E+25
U-235	3.43E+25	2.69E+25	2.37E+25	2.08E+25	1.58E+25	1.18E+25