safety series

safety series No.58

PROCEDURES AND DATA

Concepts and Examples of Safety Analyses for Radioactive Waste Repositories in Continental Geological Formations

INTERNATIONAL ATOMIC ENERGY AGENCY, VIENNA, 1983

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CONCEPTS AND EXAMPLES OF SAFETY ANALYSES FOR RADIOACTIVE WASTE REPOSITORIES IN CONTINENTAL GEOLOGICAL FORMATIONS

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Printed by the IAEA in Austria
April 1983

SAFETY SERIES No. 58

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INTERNATIONAL ATOMIC ENERGY AGENCY
VIENNA, 1983

A PUBLICATION WITHIN THE IAEA PROGRAMME ON THE UNDERGROUND DISPOSAL OF RADIOACTIVE WASTES

CONCEPTS AND EXAMPLES OF SAFETY ANALYSES
FOR RADIOACTIVE WASTE REPOSITORIES
IN CONTINENTAL GEOLOGICAL FORMATIONS
IAEA, VIENNA, 1983
STI/PUB/632
ISBN 92–0–123383–3

FOREWORD

This document is addressed to authorities and specialists responsible for or involved in planning, performing, and/or reviewing safety assessments of underground radioactive waste repositories. It is a companion to a general introductory document on the subject, Safety Assessment for the Underground Disposal of Radioactive Wastes, IAEA Safety Series No. 56, 1981, and reference to this earlier document will facilitate the reader's understanding of the present report. Since examples of safety analyses are summarized here, it is hoped that this document will contribute to providing a basis for a common understanding among authorities and specialists concerned with the numerous studies involving a variety of scientific disciplines. While providing technical information, this document is also intended to stimulate further international discussion.

The IAEA has been active in the field of radioactive waste management for many years. In 1977, a draft proposal was prepared for a future IAEA programme on the underground disposal of radioactive wastes. An Advisory Group meeting from 30 January to 3 February 1978 confirmed this proposal and recommended that a set of guidelines be published for the field of underground disposal of radioactive wastes. These guidelines are intended to cover the needs and interests of both developed and developing countries and to include the following subjects:

(a) General and regulatory activities and safety assessments;
(b) Investigation and selection of repository sites;
(c) Waste acceptance criteria;
(d) Design and construction of repositories;
(e) Operation, shutdown and surveillance of repositories.

The general introductory document was part of this IAEA programme. The present document is an extension designed to illustrate how these guidelines are implemented. The initial working draft was written by the Scientific Secretary to facilitate the preparation of the document by an Advisory Group. Following review by the Technical Review Committee on Underground Disposal of Radioactive Waste, which met in Vienna from 10 to 14 November 1980, the draft was revised and expanded by an Advisory Group meeting in Vienna from 17 to 21 November 1980. Subsequently, the final draft was examined by the Technical Review Committee in Vienna from 2 to 6 November 1981.

Another companion report to the general introductory document cited above is being prepared: it is entitled Concepts and Examples of Safety Analyses for Radioactive Waste Repositories in Shallow Ground. These three documents are supported by a series of other relevant publications recently issued by the IAEA:

Site Selection Factors for Repositories of Solid High-Level and Alpha-Bearing Wastes in Geological Formations, IAEA Technical Reports Series No. 177 (1977)

Development of Regulatory Procedures for the Disposal of Solid Radioactive Waste in Deep, Continental Formations, IAEA Safety Series No. 51 (1980)

Underground Disposal of Radioactive Wastes: Basic Guidance, IAEA Safety Series No. 54 (1981)

Shallow Ground Disposal of Radioactive Wastes: A Guidebook, IAEA Safety Series No. 53 (1981)

Site Investigations for Repositories for Solid Radioactive Wastes in Deep, Continental Geological Formations, IAEA Technical Reports Series No. 215 (1982)

Site Investigations for Repositories for Solid Radioactive Wastes in Shallow Ground, IAEA Technical Reports Series No. 216 (1982)

Other appropriate IAEA publications, prepared under the Radiological Safety Programme, might also be consulted for information on related topics. For the present document, the most important are:

Basic Safety Standards for Radiation Protection, IAEA Safety Series No. 9, 1982 edition (1982).

Principles for Establishing Limits for the Release of Radioactive Materials into the Environment, IAEA Safety Series No. 45 (1978)

Governmental Organization for the Regulation of Nuclear Power Plants, A Code of Practice, IAEA Safety Series No. 50-C-G (1978).

The Agency records with deep regret the death of Everett Irish in 1982, soon after he had completed all the work necessary for the preparation of this publication. Both before he joined the Agency and for the two and a half years he was on its staff he made a most valuable contribution at the international level to studies on all aspects of safety assessments connected with the handling, treatment and disposal of radioactive wastes. He and his expertise will be greatly missed.

CONTENTS

1. INTRODUCTION

The safe management and disposal of radioactive wastes from the various parts of the nuclear fuel cycle are important aspects of nuclear power development for both developed and developing countries. Authorities in these countries are faced with selecting and using appropriate waste disposal systems for the numerous types of waste, with their various radiochemical, chemical and physical forms. They also need methods to ensure that the safety of these systems is adequate, that the ultimate objective of waste disposal will be met, and that no unacceptable detriment to humans will occur at any time as a result of disposal operations.

Underground disposal of wastes, with the wastes appropriately immobilized and packaged, is generally agreed to be an adequate way of providing the necessary protection for humans and the environment [1]. Five types of underground disposal system are used or under development. Three involve emplacement of solid wastes in (a) deep geological repositories; (b) repositories in man-made or natural rock cavities; and (c) shallow ground repositories. The remaining two involve (d) injection of self-solidifying fluids containing wastes into fractures within impermeable strata; and (e) injection of liquid wastes into isolated porous and permeable strata. This report deals with only deep geological repositories, the others being dealt with in other reports.

Safety assessments are necessary to determine the expected performance of a repository system, to compare it with acceptability criteria, and to present the results for judgement by the appropriate authorities. They are important in every phase of system development: system selection; site confirmation; repository design, construction, operation, shutdown and sealing; and licensing processes relevant to these phases.

Safety assessments are of two general types, generic and site-specific, and they are normally performed in an iterative manner until the system being analysed is well understood and conclusions can be drawn. Generic studies are useful for making decisions regarding a choice of a disposal concept and the appropriate use of available resources. Generic assessments are also helpful in gaining acceptance for a disposal concept itself. Site-specific assessments are necessary for decisions affecting siting, design, and licensing for construction, operation, shutdown and sealing of a specific repository.

Overall approaches to making safety assessments and descriptions of general methods that may be employed are discussed in a companion report [2]. Building on the information presented there, this report presents some concepts and examples of safety analyses and the methods used to make safety assessments; however, the comparisons with acceptability criteria required to complete a safety assessment are incomplete because acceptability criteria are not yet generally agreed upon.

The purposes of this introductory report are:

(a) To identify the factors to be taken into account in radiological safety analyses of deep geological repositories, indicating as far as possible their relative importance during the various phases of system development;
(b) To show how these factors have been analysed in various safety assessment studies; and
(c) To comment on the merits of the selected and alternative approaches.

Thus, the report seeks to broaden and enhance the understanding of these rapidly developing methodologies [3–5] through use of examples of generic safety assessment studies for deep geological repositories. In doing so, credit is given to those who performed the actual studies and certainly there is no intent to criticize their work in any regard. In the context of their use, the safety analyses served their intended purposes well.

2. SCOPE

Safety analyses carried out for generic assessments of six hypothetical deep geological waste repositories are summarized in Appendixes A–F of this report:

INFCE (International Nuclear Fuel Cycle Evaluation) salt repository [6];
Netherlands domed salt repository [7];
INFCE hard rock (granite) repository [8];
Swedish hard crystalline rock repository [9];
Canadian Shield crystalline rock repository [10];
Belgian clay repository [11].

With the exception of the Swedish study, the safety assessments [7, 12–15] consider only the post-sealing phase of a repository; they are generic in nature and are relevant primarily to the feasibility of the concept of mined geological repositories for radioactive wastes in the specific host rocks.[1]

Safety analyses for the site selection and design phases for a repository would be similar to those summarized in the Appendixes. Safety assessments for the construction and operation phases, not included here, would be quite

[1] A Belgian safety assessment is not yet completed or documented, but the work is discussed generally in Appendix F of this Safety Series.

different and would be similar to those for other types of nuclear installation; however, these would also have to include assessments of non-nuclear and nuclear risks (e.g. risk to miners of constructing a very deep repository and risks to the public resulting from a shallower construction). Thus, the radiological safety assessment studies are also relevant to the construction safety studies.

Wastes assumed to be emplaced in these repositories were generally vitrified high-level waste from reprocessing plants, unreprocessed spent fuel when considered as waste, and other alpha-bearing wastes in solidified forms; for the INFCE studies, however, all wastes from the nuclear fuel cycle except mill tailings were assumed to be emplaced in the repositories. The conditioning of the wastes varied for the different studies, as described in the specific examples in the text.

Because the bases and purposes of the example analyses had significant differences and the methodologies used are different, comparisons of the results of the various analyses may be misleading. Thus, no inferences should be drawn directly from this report with regard to the relative capabilities of specific systems to isolate wastes.

3. DISPOSAL IN DEEP GEOLOGICAL REPOSITORIES

Disposal of radioactive wastes underground has been studied for more than 20 years. From these studies various underground disposal options have evolved, as mentioned previously. A number of factors will influence the decision on which underground disposal option is appropriate in a given circumstance. The most important of these factors are [1]:

Waste types, quantities and conditioning;
Repository design;
Geological and environmental conditions;
Radiological protection considerations; and
Socio-economic conditions.

The option to be selected depends upon a myriad of considerations, but the dominating factor is the waste category to be emplaced in the repository. Once the option has been selected, the two other major aspects to be considered are the site selection and design for the repository system. These aspects are briefly discussed in this chapter to provide a background for the subsequent discussion on the concepts and examples of safety analyses.

TABLE I. GENERAL CHARACTERISTICS OF WASTE CATEGORIES WITH REGARD TO DISPOSAL

Waste category	Important features[a]
I. High-level, long-lived	High beta/gamma Significant alpha High radiotoxicity High heat ouput
II. Intermediate-level, long-lived	Intermediate beta/gamma Significant alpha Intermediate radiotoxicity Low heat output
III. Low-level, long-lived	Low beta/gamma Significant alpha Low/intermediate radiotoxicity Insignificant heat output
IV. Intermediate-level, short-lived	Intermediate beta/gamma Insignificant alpha Intermediate radiotoxicity Low heat output
V. Low-level, short-lived	Low beta/gamma Insignificant alpha Low radiotoxicity Insignificant heat output

[a] The characteristics are qualitative and can vary in some cases; "insignificant" indicates that the characteristic can generally be ignored for disposal purposes, because safety analyses have shown that it is not important.

3.1. WASTE CHARACTERISTICS

Radioactive wastes can be categorized in several ways. For the purpose of this discussion they are grouped into five categories as given in Table I [1]:

Category I wastes include the high-level waste from reprocessing of spent fuel, and the spent fuel itself if it is declared a waste. Category II wastes include primarily fuel element cladding hulls, associated fuel hardware, and insoluble dissolver residues. Category III includes those wastes having significant levels of long-lived alpha-emitting radionuclides (i.e. neptunium, plutonium, americium, and curium) but also low beta/gamma levels. Whereas deep geological repositories are considered suitable technically for disposal of all categories and types of conditioned radioactive waste, they are generally preferred only for types of waste (long-lived) in Categories I, II, and III. The cost of disposal is an important consideration that influences the selection of other options for wastes in Categories IV and V, which have no need for the long-term isolation provided by deep geological repositories, if other adequate sites exist.

A few specific comments on the nature of the wastes from a radiation standpoint can be made to give a perspective on how the radioactivity and thermal power change with time [16]:

(a) *Beta and gamma emitters:* The greatest contribution to beta-gamma radiation comes from fission products. A second source consists of neutron activation products, i.e. elements in fuel, cladding, or surrounding structures that absorbed neutrons and as a result have been converted from stable to radioactive isotopes. With a few exceptions (e.g. ^{129}I with a half-life of 17 million years, ^{99}Tc with a half-life of 200 000 years, ^{73}Zr with a half-life of 1.5 million years, and ^{135}Cs with a half-life of 2 million years), the beta-gamma emitters produced in significant quantities have half-lives of a few decades or less. Their radioactivity will be reduced to very low levels within about 1000 years of their production. The longer-lived fission products constitute only 0.00001 wt% of the fission products of typical commercial reactors. Also, a small beta contribution comes from the actinide elements.

(b) *Alpha emitters:* The major sources of alpha radiation are the transuranic isotopes produced in the fuel by neutron absorption and their decay products (called daughters). These alpha emitters include isotopes of neptunium, plutonium, americium, curium, and their daughter products. The uranium isotopes and their daughter products are also sources of alpha radiation. The range of half-lives for alpha-emitting isotopes is comparable to that for beta-gamma emitters; however, the decay of a beta-gamma emitter generally leads immediately to a stable isotope; whereas the decay of an alpha emitter generally leads to another alpha-emitting isotope. The long-term buildup of alpha-emitting daughters is of importance because these are the major contributors to the potential risk from high-level wastes or spent fuel for periods greater than 1000 years.

The radioactivity and thermal power of long-lived radionuclides per unit mass of high-level waste or spent fuel is several orders of magnitude less than for short-lived radionuclides (principally fission products) for the first few hundred years after discharge from a nuclear reactor.

Before the wastes can be considered for disposal they must be properly conditioned (i.e. immobilized and packaged). Acceptability criteria for storage and underground disposal of conditioned wastes have not yet been commonly adopted, although several countries have published recommendations that describe the criteria for acceptance or rejection of such wastes and the IAEA is dealing with the topic [1, 17]. The setting of acceptability criteria is a responsibility of national regulatory bodies and is a topic under active discussion. In addition to general acceptability criteria, criteria of a specific nature must be developed in relation to safety analyses of each specific waste disposal system, some aspects of which are dependent on site-specific conditions [2]. Thus, the acceptability of specific disposal operations will be determined through specific safety analyses.

The technology for immobilizing wastes is quite well developed [5, 16] both for high-level waste [18, 19] and low- and intermediate-level wastes [20], whereas spent fuel (with the exception of small amounts of gaseous fission products) is already in an immobilized form. The waste form (e.g. vitrified waste) is only one of the engineered barriers that will be used in a waste repository system to delay or prevent radionuclide movement away from the waste package into the geosphere; other engineered barriers that might be used include the container, overpack and/or migration retardants. Although continuing to be developed, the technology for conditioning wastes, as emplaced in a geological repository, can be used to design a system for more than one thousand years of waste containment within geological repositories in basalt, granite, salt and shale [16]. Detailed descriptions of these technologies are not given here, although the types of waste conditioning assumed for each safety assessment study presented in this document are described in Appendixes A–F. It is interesting to note how the example safety assessment studies vary in their consideration of the treatment of the engineered barriers; some have an extensive system of barriers and others consider that only the waste form is present.

3.2. REPOSITORY SYSTEM

3.2.1. Basic principles

Deep geological repositories for the disposal of high-level and alpha-bearing wastes have been the subject of much research and development work for many years and pilot facilities are now being planned and designed [21].

The basic principle governing these facilities is that they must be sited, designed, constructed, operated, shut down and sealed in such a way that the operating personnel and the public in general will be adequately protected at all times from radiological hazards arising during their operation and after shutdown and sealing [18]. Other principles that must be observed relate to environmental protection and the non-radiological impacts on future generations; since this document is concerned only with radiological safety analyses, no further consideration will be given to these two factors.

3.2.2. Repository site selection

The characteristics (e.g. stability, natural barriers to radionuclide migration etc.) of the site and host rock for a geological repository are selected to offer assurance of adequate capability to isolate the radionuclides in the waste, preventing them from being released into the biosphere in unacceptable quantities and concentrations. An Agency publication presents a stepwise process for investigations and selection of an appropriate site [22]; in addition, Ref. [1] discusses the topic extensively.

The characteristics of geological formations may be sequentially evaluated at the national, regional, and site-specific levels in order to permit the selection of a site for a repository. The assessment of potential repository sites with regard to site selection factors permits the identification of favourable locations. These factors enter the process at two stages, namely the identification of areas worthy of more concerted study, using generic criteria, and the evaluation of sites which emerge as having potential for the location of an underground repository on a site-specific basis.

The parameters that affect the suitability of a potential site are highly site-specific. Thus, general criteria cannot be used to select individual sites. Evaluation of data and development of criteria must be done for specific cases. However, factors can be identified that indicate the general suitability of sites as potential repositories. By identification of these factors and of the manner in which they may affect the safe use of a site, general guidance can be developed for selecting suitable repository sites. Major site-selection factors are described in Ref. [23].

It should be recognized that it is unlikely that any site will be found that incorporates all the advantageous factors. Neither is this necessary. For example, factors relevant to a salt dome will not be similarly relevant to crystalline rocks. However, it is essential that the long-term safety requirements are satisfied, and to ensure this, an analysis of all the confinement factors should be made.

Three general groups of potential host rock types for geological repositories are (a) evaporites, (b) other sedimentary rocks, and (c) igneous and metamorphic rocks. A few comments about these rocks and their properties relative to geological repositories will be made; more extensive discussions are found in the summaries of the safety assessments in Appendixes A–F and in Ref. [23].

The evaporite most strongly considered for geological repositories is rock salt, as either bedded deposits, domal masses, or as other salt structures. Salt has favourable properties and is widespread in occurrence as undisturbed units within geological settings. Salt is more plastic than almost any other rock and is thus able to seal naturally formed fractures as well as man-made and backfilled canister bore-holes and repository rooms. This property makes salt largely impermeable to gas and liquids. Other favourable properties of rock salt are: good compressive strength; good thermal conductivity; and ease of mining to provide a cost-efficient subsurface excavation. However, because of its high solubility, circulating unsaturated groundwater could, over a long time, breach the geological integrity of a repository. Other potential disadvantages include: small pockets of entrapped brine and inclusions of hydrated minerals whose fluids can migrate towards heat sources under certain thermal conditions; chemical aggressiveness of the environment for many container materials; low sorptive capacity; and possible salt movement (both the natural post-diapiric movement and the expansion movement due to the thermal loading from the emplaced high-level waste).

Other sedimentary rocks, mainly argillaceous rocks, have also been proposed as host formations for geological repositories. Research and development work in this field is largely centred in a few countries that contain extensive deposits. Argillaceous formations such as clays, claystones, and certain shales and marls also display plasticity; these argillaceous rocks display very low permeability, good sorptive characteristics, and low solubility. Potential disadvantages include: dewatering of the hydrous clay minerals in response to the thermal load, and adverse effects upon rock-mechanical properties; possible presence of organic matter and gases; existence of inhomogeneities; and possible difficulties in mining and keeping excavations open.

Igneous and metamorphic rocks are considered by several countries as prime candidates for repositories for deep underground disposal of radioactive wastes. A variety of rocks such as granite, gabbro, basalt and tuff are examples of igneous rocks that have been considered as potential host rocks for waste repositories. They commonly occur in large volumes. Similarly, various types of gneiss, quartzite, psammite and migmalitic complexes are included in metamorphic rocks that are potential host rocks. These rock types generally demonstrate long-term stability, high rock strength, high chemical stability, moderately high thermal conductivity, and low porosity. In fractured crystalline rocks, alteration in the fractures can produce secondary minerals, which are usually clay minerals with high sorptive capacities. On the basis of considerable mining experience in a variety of subsurface facilities, it may be said that the man-made openings of a repository can be expected bo be very stable. Rocks formed by igneous and metamorphic processes tend to be brittle, or non-plastic, at depths considered for repositories, and thus likely to have fractures and other secondary openings which commonly contain groundwater. The presence of rock inhomogeneities, largely the result of the nature, orientation, and magnitude of fractures, makes the modelling of the hydrogeology in such systems difficult.

3.2.3. Repository design concepts

For high-level and alpha-bearing wastes, the leading concept for a geological repository being pursued by several countries involves the emplacement of heat-emitting, high-level waste canisters into spaced bore-holes in the floors of specially excavated repositories that are relatively deep, typically more than 200 metres below the surface. The same basic concept could be used if spent fuel were to be disposed of as a waste. After disposal of the canisters, the bore-hole and chambers would be backfilled suitably and sealed. If other waste packages to be emplaced are containers of low- and intermediate-level or alpha-bearing wastes, they could be either stacked or dumped into disposal rooms. Once a room has been filled with containers, it also would be backfilled with suitable material and sealed. Post-emplacement data collection and repository assessment in and around the repository may be performed for a limited period of time.

From a physical standpoint the conceptual geological repositories are all based on a similar idea, illustrated in Fig.1. The repositories have a system or network of tunnels in a horizontal plane(s) some hundreds of metres below grade. These tunnels are serviced through a set of access shafts for movement of people, equipment and materials, waste packages, and for ventilation purposes. Because this report deals only with the post-sealing phase of a repository, no further mention of these shafts will be made except to point out the importance of plugging and sealing bore-holes and shafts to (a) eliminate preferential pathways for transporting radionuclides between a repository and subsurface or surface water; and (b) to prevent abnormal water flow into a repository.

The depths of the conceptual repositories in salt and hard crystalline rock are generally considered to be 500 m or more below grade. However, the depth of clay repositories must be limited to a few hundred metres because of the geomechanical properties of clay. Although the high plasticity of clay is an advantage as far as its physical barrier integrity is concerned, it creates mining problems with regard to working depths, the diameter of the excavated tunnels, and the nature and thickness of the linings. For example, for the Belgian clay repository only circular tunnels or galleries with a free diameter of 3 to 4 m are technically achievable at its depth of 225 m; this limits the size of waste containers that can be handled. In addition, a hole cannot be dug without providing a lining to resist the lithostatic pressure; thus, holes and tunnels are

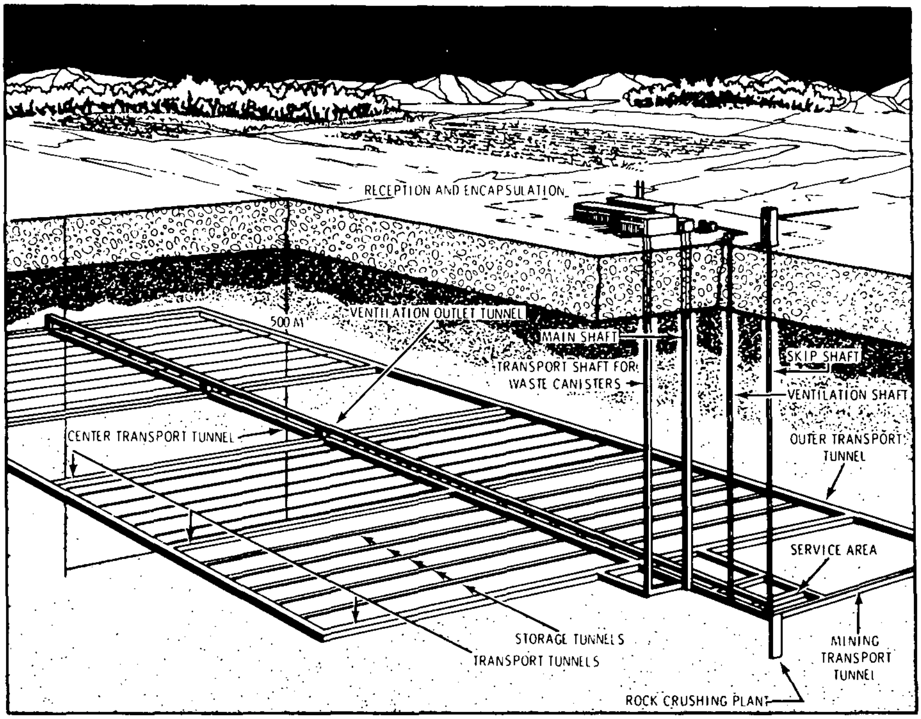

FIG.1. Sketch of a repository.

lined with welded steel pipes which are installed as excavation or drilling proceeds. These requirements modify the subsurface facilities in a clay repository so that they are, in reality, physically quite different than those for salt and granite even though conceptually they are similar. Besides having steel linings for the holes and circular tunnels, the holes are inclined at 45° and drilled alternatively from the right and left sides of the tunnel. Nevertheless, the sketch in Fig.1 can serve to illustrate features of the various repository emplacement systems that are important for purposes of safety analyses.

In conclusion, it should be stated that when appropriate sites and repository designs are selected, the risks of many of the potentially important phenomena relevant to the isolation of wastes in geological repositories are reduced to very low levels; for example, the probabilities of potential detrimental impacts and/or risks will be extremely low for natural events like earthquakes, flooding, meteorite impact and perhaps human activities like inadvertent future drilling. Chapter 4 deals with the safety assessment process itself.

4. REVIEW OF SAFETY ASSESSMENT

Before the safety analyses for the selected conceptual repositories are described, it would be beneficial to review the safety assessment process [2] so that the context of the individual steps of the analyses can be better understood. First, it should be emphasized that the assessment must consider the repository and its environment as a system; for convenience the system can be described as a combination of the following components:

(a) Repository, its engineered barriers (backfill and seals) and its contents (waste form, container, overpack, migration retardant);
(b) Geosphere (host rock and surrounding material and, where applicable, interstitial fluids in the host rock, deep groundwater and natural resources); and
(c) Biosphere (soil, surface waters, shallow aquifers, atmosphere and biota).

Some of the elements in the first two components represent barriers. The role of these barriers is to prevent or delay the initiation of radionuclide release from the waste, to distribute the release over time and to retard transport of radionuclides through the geosphere and into the biosphere. Depending upon the type of host rock chosen and the type and form of waste disposed, the repository systems vary; not all the components mentioned above are relevant to all concepts.

As stated earlier, safety assessments are of two general types, generic and site-specific, and they are normally performed in an iterative manner until the system being analysed is well understood and conclusions can be drawn. Generic studies are useful for making decisions regarding a choice of a disposal concept and the appropriate use of available resources. Generic assessments are also helpful in gaining acceptance for a disposal concept itself. Site-specific assessments are necessary for decisions affecting siting, design, and licensing for construction, operation, shutdown and sealing of a specific repository.

Various classification schemes are used to subdivide a safety assessment into components. The classification scheme used in this report differentiates two major components: scenario analysis and consequence analysis:

(a) *Scenario analysis* involves identification and quantitative definition of phenomena which could initiate and/or influence the release and transport of radionuclides from the source to man. Thus, scenario analysis provides initial and boundary conditions for subsequent consequence analysis. It also provides estimates of the probabilities of occurrence of phenomena.
(b) *Consequence analysis* involves estimation of the subsequent transport of radionuclides from the source to man and the resulting radiation doses, using the system descriptions derived from the scenario analysis.

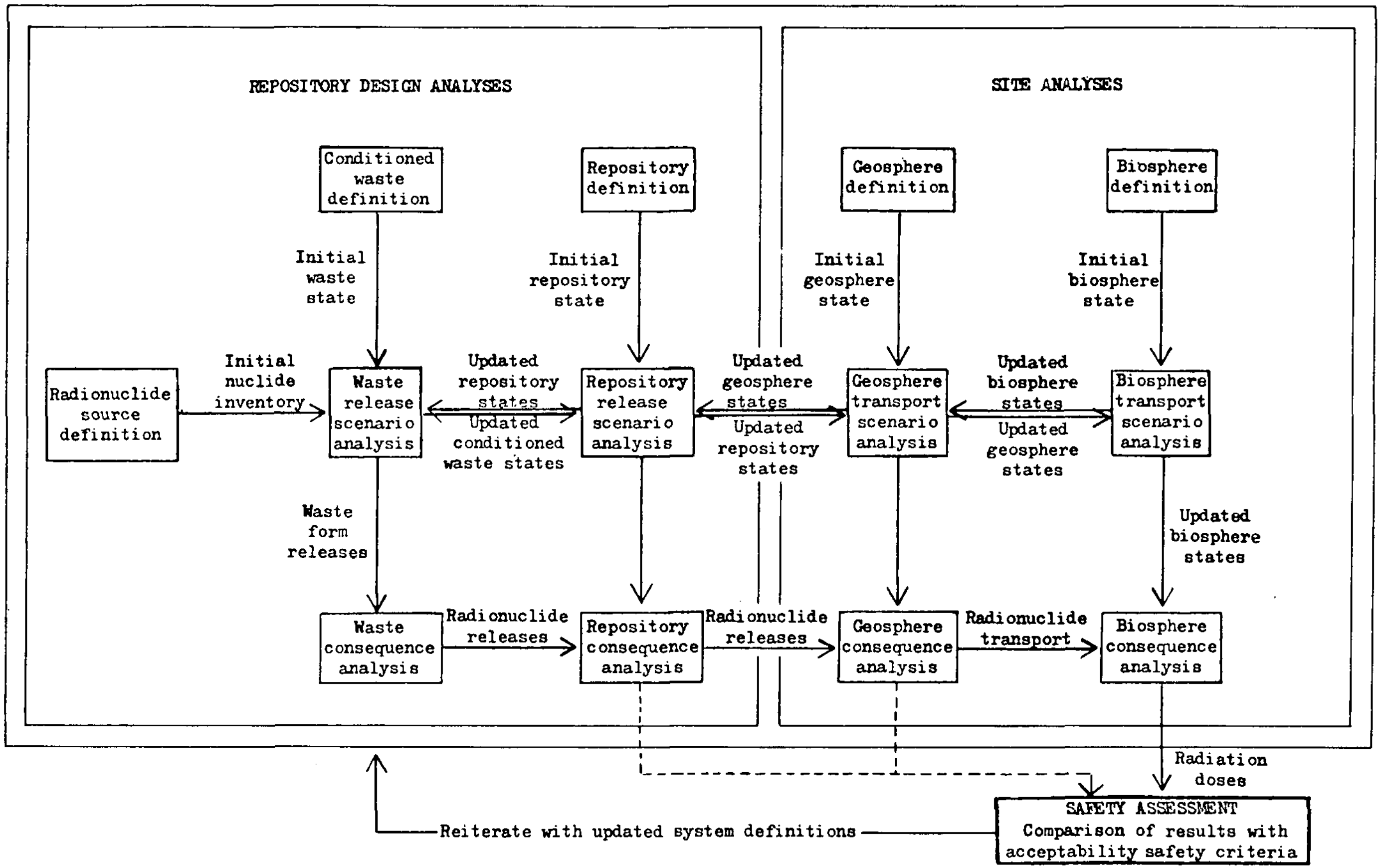

FIG. 2. Safety assessment components and their interactions.

Comparison of the results of biosphere consequence analyses with acceptability criteria completes the safety assessment. Figure 2 illustrates these components of the safety assessments and their interactions. Iterations within specific analysis steps or complete iterations of the safety assessment process are normally performed; however, for some of the illustrative examples, only one set of analyses is described. Some of these iterations are made to obtain information about the uncertainty of the results and means for compensating for this uncertainty.

Safety analyses require the use of models which quantify the ways significant phenomena occur. (A model is a mathematical representation of a real system which is sufficiently simplified and compact to be amenable to useful quantitative analysis without excluding important phenomena. The inclusion of unnecessary precision and insignificant third- and fourth-order phenomena in models should be avoided since this can make the time and cost associated with making the important sensitivity studies considerable.)

Individual or sub-models can be combined to carry out integrated studies which enable the estimation of the total system performance; that is, how the total set of sub-systems calculations (including the anlysis of the engineered barriers, the natural barriers of the geological formation and the biosphere) are considered together in evaluating the possible dose to humans. A major reason for requiring adequate integration of sub-systems is that sub-systems are not independent. For example, the dissolution or leaching rate of the waste form is dependent on the rate of water into the repository from the geosphere. At one extreme, the flow rate could be so low that the dissolution rate would be solubility-limited and, at the other extreme, so fast that it would be chemical reaction-rate limited.

4.1. SCENARIO ANALYSIS

As indicated above, scenario analysis involves the identification and quantitative definition of phenomena which could initiate and/or influence the release and transport of radionuclides from the source to man,.and may also include the estimation of the probabilities of occurrence of these phenomena.

Several types of occurrence could lead to the release of radionuclides and in some cases enhance such releases and/or radionuclide transport rates. To analyse release and transport scenarios, it is necessary to identify the phenomena which are relevant. These phenomena could be due to:

(a) Effects of natural processes and events (e.g. groundwater flow, erosion, faulting etc.);
(b) Effects of human activities (e.g. alterations of hydrology, mining, drilling etc.); and/or

TABLE II. PHENOMENA POTENTIALLY RELEVANT TO RELEASE AND TRANSPORT SCENARIOS FOR WASTE REPOSITORIES

Natural processes and events[a]

Climatic change
Hydrology change
Sea-level change
Denudation
Stream erosion
Glacial erosion
Flooding
Sedimentation
Diagenesis
Diapirism
Faulting/seismicity
Geochemical changes
Fluid interactions
- Groundwater flow
- Dissolution
- Brine pockets

Uplift/subsidence
- Orogenic
- Epeirogenic
- Isostatic

Undetected features
- Faults, shear zones
- Breccia pipes
- Lava tubes
- Intrusive dykes
- Gas or brine pockets

Magmatic activity
- Intrusive
- Extrusive

Meteorite impact

Human activities

Faulty design
- Shaft seal failure
- Exploration bore-hole seal failure

Faulty operation
- Faulty waste emplacement

Transport agent introduction
- Irrigation
- Reservoirs
- Intentional artificial groundwater recharge or withdrawal
- Chemical liquid waste disposal

Large-scale alterations of hydrology

Undetected past intrusion
- Undiscovered bore-holes
- Mine shafts

Inadvertent future intrusion
- Exploratory drilling
- Archaeological exhumation
- Resource mining (mineral, water hydrocarbon, geothermal, salt, etc.)

Intentional intrusion
- War
- Sabotage
- Waste recovery

Climate control

Waste and repository effects

Thermal effects
- Differential elastic response
- Non-elastic response
- Fluid pressure, density, viscosity, changes
- Fluid migration

Mechanical effects
- Canister movement
- Local fracturing

Chemical effects
- Corrosion
- Waste package – rock interactions
- Gas generation
- Geochemical alterations

Radiological effects
- Material property changes
- Radiolysis
- Decay-product gas generation
- Nuclear criticality

[a] Explanations of natural phenomena are provided in Ref. [23].

(c) The combined effects of the waste and repository (e.g. thermal, chemical, mechanical, radiological etc.).

Table II suggests a list of phenomena potentially relevant to release and transport scenarios for waste repositories. Although the list may not contain all relevant phenomena, it provides a comprehensive view of the types of phenomenon that might be considered in safety analyses. Through a careful site investigation and site selection process [22–24], a major fraction of these phenomena can usually be eliminated from detailed consideration. In the studies described here, this process of elimination has been assumed even though it is stated explicitly for only one study.

Scenario analysis models are used for defining and analysing potential phenomena which might change the state of the system. For example, faulting through or in proximity of a repository might change the permeability of the formation and the pattern of groundwater flow; or, with respect to glaciation, it would be necessary to estimate the effect of the overburden pressure on the degree of fracturing of the geological formation and the groundwater flow.

4.2. CONSEQUENCE ANALYSIS

The ultimate aim of consequence analysis is to determine the radiation doses to man resulting from the disposal of radioactive waste. The analysis of the radiological consequences involves calculations of the release, dispersion and transport of radionuclides from the waste form through engineered barriers, the repository, the geosphere and the biosphere, and finally calculations of radiation doses to man. These calculations lead to the estimation of individual and collective doses as a function of time after disposal. The total impact of the repository, considered as a source of exposure, is given by the collective dose commitment.

Consequence analysis begins with analysing the release of radionuclides from the waste form and the repository. Temperature distributions, mechanical stress conditions, radiolytic effects, corrosion and sorption are the major items involved. Subsequently, the geochemical and geohydrological processes in the host formation are analysed to estimate the transfer rate of radionuclides through the geosphere to the biosphere. Finally, the processes of biological uptake by man and resultant exposure are analysed to arrive at the doses to man.

Making the necessary calculations requires that the observed systems be mathematically described by models. It may be possible to work with specific models for each process in the pathway from the waste to man, but integrated models are also useful. The models must, however, be linked together in such a way that the calculations can be performed reliably. The combinations of models used will depend upon the situation.

4.3. DATA REQUIREMENTS

For safety analyses of particular systems the following data are typically required:

(a) Waste characterization (composition versus time, quantity, heat generation versus time etc.);
(b) Container characteristics (mechanical, chemical etc.);
(c) Repository characteristics (dimensions, backfill/buffer, structural material etc.);
(d) Geosphere characteristics (geology, hydrogeology, geochemistry etc.);
(e) Biosphere characteristics (atmosphere, aquatic, terrestrial, demographic etc.).

Data are collected for components of the total system from existing data files and literature, laboratory experiments, in-situ tests and field observations; the site investigation work is a particularly important part of the data collection process. An important and unique aspect is that the data need to be applicable for times far in the future. In some cases long-term processes may be accelerated in experiments that provide data which can be applicable to long-term conditions. In other cases conceivable variations in data (e.g. due to climatic change) can be accounted for by variation analyses.

4.4. EVALUATION AND APPLICATION OF RESULTS

The principal numerical results of safety analyses for the post-sealing phase of a repository are predicted doses to individual members of the public and collective doses to present and future generations; the results may also include estimates of the probabilities that these doses will be received. Doses and probabilities may be combined to calculate expected doses, presented as dose distribution functions, or as separate sets of results. Predicted numbers of health effects can also be calculated, using appropriate dose/effect relationships.

The final stage of safety assessment is the comparison of the safety analyses results with acceptability criteria developed by the appropriate national and international authorities. At present, generally agreed criteria for disposal in continental geological formations do not exist, although the IAEA and many national regulatory bodies are dealing with the subject [1, 17].

4.5. DOCUMENTATION OF RESULTS

Clear and effective documentation of safety assessment results is necessary for communication, not only with the regulatory body but also within the implementing organization and with other interested parties. Such documentation is needed especially because safety assessments of waste repositories are not numerous or familiar documents and the technology is rapidly developing.

5. EXAMPLES OF SAFETY ANALYSES

The purpose of this chapter is to introduce and describe six safety assessments that have been selected to provide examples of the methods used for safety analyses. Chapter 6 provides commentaries on these safety analyses. The selected examples, described in Appendixes A–F, are:

Appendix	Repository
A	INFCE salt
B	Netherlands domed salt
C	INFCE hard rock (granite)
D	Swedish hard crystalline rock
E	Canadian Shield crystalline rock
F	Belgian clay

Each of these examples has been organized and directed towards specific purposes, with different assumptions, so that the results may not be directly comparable; however, each provides some insights into the methods that can be used for assessing the safety of geological disposal of radioactive wastes.

5.1. INFCE SALT REPOSITORY

Both the INFCE salt and hard-rock repository assessments [12, 13] were designed to evaluate and compare the predicted long-term health and safety impacts of each of seven different reference fuel cycles when all radioactive wastes (except those associated with mining and milling) are placed in a geological repository. As with the rest of the INFCE studies, the purpose was only to compare the fuel cycles; it was not to prove or disprove the safety of disposal in a particular medium – quite the contrary, the assumptions in almost all cases were chosen for their extreme conservatism and the sites were kept generic so they would not be interpreted as representative of any specific site in any particular country.

The generic site of the salt repository was assumed to be located in a large, undeformed sedimentary basin. The top of the salt formation was assumed to be 250 m below the surface, and the horizon of the single-level repository was 600 m below the surface. The salt formation was assumed to extend downwards indefinitely, with no aquifers beneath the repository. The hydrological system above the salt is typical for a layered sedimentary sequence found in conjunction with salt. The aquifers connect to a river, 6.2 km from the repository with a flow rate of 500 m^3/s. The hydraulic gradients in all aquifer systems were assumed to be 1 m/km in the direction of the river.

Two general cases were analysed for each fuel cycle, one without a major disturbance (normal scenario) and one in which a major geological perturbation (abnormal scenario) breached the repository containment. For the normal scenario the repository site would not experience within several million years a disruptive event that would release radionuclides to the biosphere. Solid materials buried in salt generally will not move by themselves. Without flowing water only solid-state diffusion can move the radionuclides from their point of burial.

The abnormal scenario assumed an incredible, violent geological event which resulted in the creation of a saturated brine pathway from the overlying aquifer system through the repository and back to the river or biosphere. Further conservative assumptions regarding this abnormal scenario included breach initiation 50 years after closure, no engineered barriers and dissolution of half the entire repository waste in a period of 3000 to 4000 years as a result of the elevation in temperature and the assumption that the waste had disintegrated into small particles.

Analysis of the geohydrological system for the abnormal scenario indicated a groundwater travel time of 100 000 years; radionuclide decay, retention and dispersion were also modelled with a transport code.

Concern has been expressed that the abnormal scenario, described above, is not a reasonable one; the authors acknowledge the unreasonable nature of the scenario but chose it only to facilitate an analysis that would allow the fuel cycles to be compared. It was considered only as representative of a worst case flow through a salt repository and it did account for the mitigating effects of the geohydrological system isolating the radionuclides from the biosphere. The modelling results confirmed that deep geological systems such as the hypothetical salt site described cannot maintain very productive flow systems through a ruptured repository even for the postulated occurrence of an extremely violent geological event such as that modelled for the abnormal scenario.

Relatively simple dose models, derived from more complex dose codes, were utilized for assessing dose to the most exposed individual as a result of 50 years' exposure after 50 years' environmental buildup at the peak value. To obtain maximum possible doses, the times of peak isotope concentrations were obtained from the transport model, and the contribution to dose by all nuclides was calculated at each peak time. For assessment purposes these doses were compared with a dose an average individual might receive from natural background radiation during the same 50-year period.

5.2. NETHERLANDS DOMED SALT REPOSITORY

A generic safety assessment was performed by the Radioactive Wastes Subcommission of the Interministerial Nuclear Energy Commission to determine the feasibility and the acceptability of the disposal of high-level radioactive wastes into rock salt formations. The safety analyses for this assessment included:

(a) Geohydrological modelling to establish the isolation properties of a salt dome for waste disposal purposes; and
(b) An analysis of the radiation doses following a radionuclide release in the distant future from the salt dome repository along different pathway models.

The report of the working group on safety assessment included geological data from the State Geological Service, geohydrological model calculations made by the Geohydrological Division of the National Institute for Water Supply, and radionuclide release scenarios and radiation dose calculations provided by the Institute for Application of Nuclear Sciences in Agriculture. The report on the feasibility of radioactive waste disposal in salt formations in the Netherlands was published by the Interministerial Nuclear Energy Commission [7].

For the generic safety assessment the repository tunnels were assumed to be situated at a 600 m depth in a salt dome structure which had its top rock salt at a depth of 300 m. There was a minimum isolation shield of 200 m of rock salt on the flanks of the salt dome. The conceptual design studies evolved from (a) the disposal of 50 000 canisters of high-level waste (HLW) in 50 m deep bore-holes at three consecutive disposal levels at 600, 750 and 900 m deep to (b) the disposal of the same number of canisters in 300 m deep bore-holes from 600 to 900 m. Each canister was assumed to contain 50 litres of solidified HLW derived from reprocessing 0.6 t LWR fuel.

The scenario used for establishing an average groundwater velocity over and around the top of the salt dome was based on existing geological, geophysical and geohydrological knowledge. A model was developed to calculate the rate of groundwater flow in the system of two aquifers separated by a semi-pervious layer. The process of uplift of salt domes (diapirism) is of great importance with respect to the possible dissolution of the isolation shield of rock salt surrounding the repository. Substantial uplift can lead to contact with the upper aquifer where relatively high flow velocities occur in comparison with the lower aquifer.

For a future radionuclide release from the salt dome repository, several scenarios were considered feasible depending upon the rate of continuous upward movement of the salt dome, the groundwater velocity, and the effect of long-term climatological changes. A series of values of upward movement of the salt, varying from 0.25 to 2.5 mm/year, was used in the model calculations. Even with the maximum uplift rate of 2.5 mm/year, approximately 250 000 years would elapse before the disposed waste would reach the surface. For the generic safety assessment no credit was taken for repository design or for any engineered barriers; thus, the waste was considered to surface almost intact.

Biosphere analyses considered several pathway models to calculate the consequences of the potential releases (initiated 250 000 years after disposal) in the form of radiation doses to a future population living near the contaminated area. These included:

A drinking water model;
An inhalation model;
Two agricultural models, one in which the radionuclides are taken up in plants and reach humans directly through the food chain and another in which they are eaten by cattle and reach humans via the milk or meat chain;
An external radiation exposure model; and
A fishery model.

The radiation doses calculated from the different uptake models were compared with the exposure due to natural radiation.

5.3. INFCE HARD ROCK (GRANITE) REPOSITORY

The purpose of this hard rock repository assessment [13] was discussed in Section 5.1.

The generic site of the repository was assumed to be composed of relatively large areas of granite or gneiss consisting of solid rock blocks surrounded by small fracture planes or joints that are interconnected to some degree. This degree of interconnection results in low permeability and porosity associated with these relatively large areas. These large areas of low permeability rock, dominated by flow through joint systems, are bounded by fractured zones which need not be continuous throughout the rock mass and which vary in length and width according to the stresses which now exist, or caused these highly fractured zones. The extent of the area considered is a square 25 000 m on a side. Within this area there exists a sea to the north along with lakes and rivers.

A subdued version of the topography was used to represent the groundwater table elevations throughout the region. The groundwater divide to the south has an altitude of 40 to 45 m above the level of the sea to the north.

The objective of the modelling efforts for this study was to predict the transport rate of radioactive contaminants from the repository through the geosphere to the biosphere, and thus estimate the potential dose to humans so that the release consequence impacts of the various fuel cycles could be compared. Because the data on hard crystalline rock indicate breach of the repository to be highly improbable, only a normal scenario was studied. In this scenario, radionuclides are moved by the small amounts of water (normally present in hard crystalline rocks at depth) out of the repository area after the waste canisters have failed.

It was assumed that a groundwater flow rate of 2.6 m^3/year passed through the entire repository and slowly leached the radionuclides from the wastes. The release of radionuclides from the vitrified waste within a diffusion barrier was assumed to correspond to that for diffusion of amorphous hydrated silica. Other species were assumed to be released in proportion to their concentration relative to silica.

Prediction of radionuclide transport required an estimate of groundwater movement because water is the main transport medium for waste movement in a geohydrological system. A numerical three-dimensional groundwater hydrological code was used. For the generic site the average streamline parameters (indicating the length of path and time for movement of contaminants from the repository to the biosphere) were estimated to be:

Average distance	– 7 100	m
Average velocity	– 0.61	m/year
Average water travel time	– 11 700	years

The output from the hydrological model was used as input to the one-dimensional transport model that was used to model radioactive decay and rock-nuclide-water reactions.

The radionuclides in the groundwater were assumed to enter the biosphere environment as seepage into a fresh-water lake. Subsequent transport in the biosphere was modelled using a multicompartment model with parameters selected to represent the reference granitic site. In this model the radioactivity in each compartment is described mathematically by a system of linear first-order differential equations with constant transfer rates between the compartments. Production of radioactive daughters within each comparment is considered. The exchange of radionuclides between compartments is described by transfer coefficients which give turnover per unit time.

The model of the biosphere was divided into three sub-systems of progressively increasing size referring to regional, intermediate and global ecosystems.

The structure of the model permits the recirculation of radionuclides between different compartments. The regional ecosystem includes a fresh-water lake (receptor of groundwater activity), lake sediments, soil and subsurface groundwater. The intermediate ecosystem is represented by a large lake or sea with associated sediments. The atmosphere above the regional area and intermediate sea area is the tropospheric air volume up to an altitude of one kilometre. The global ecosystem models the oceans, the continents, and the global atmosphere.

The calculated concentrations of radionuclides in the biosphere were used in the exposure pathway analysis to estimate the total intake by the most exposed individual. Pathways considered include inhalation, ingestion of food and drinking water, and external exposure from material deposited on the ground; pathways found to be of principal importance were ingestion of food and drinking water. The pathway analysis provides the individual's external dose and the inhalation and ingestion rate for each radionuclide. The intake rates are used to calculate weighted whole-body doses for each radionuclide.

5.4. SWEDISH HARD CRYSTALLINE ROCK REPOSITORY

In 1977 the Swedish Parliament passed a Nuclear Stipulation Law which requires that prior to the loading of fuel and operation of any new nuclear power reactor in Sweden the reactor operator shall, among other things, show how and where high-level waste from reprocessing (or spent unreprocessed nuclear fuel) can be finally disposed of in an "absolutely safe" way. The Swedish nuclear power industry responded to the proposed bill by organizing the Nuclear Fuel Safety Project (KBS).

The KBS project investigated both the alternatives (high-level waste from reprocessing and spent unreprocessed fuel) which were mentioned in the Stipulation Law. A report of the handling and final storage of high-level vitrified waste was published in December 1977 [14]. Thereafter, this report was supplemented by additional geological investigations. Based on these reports and an extensive review by several Swedish and foreign organizations and individuals, the Swedish Government approved the fuel loading and startup of additional nuclear power reactors in June 1979 and in April 1980. The KBS project has also completed and published a study on final storage of unreprocessed spent fuel [25].

Appendix D describes some of the methods and data used for the safety analyses included in the first KBS reports. These methods reflect the development status by mid-1977 when the analyses were made and are very similar to those used for the INFCE hard-rock repository study just described; however, the KBS studies were much more extensive and complete.

The safety analyses were carried out with the requirements and with the interpretation of the law in mind. No effort was made to perform the kind of analyses which would be required for licensing a repository at a specific site.

The repository studied was supposed to be located in hard crystalline rock of granite or gneiss. This type of bedrock is abundant in Sweden. The size of the generic repository was about 1 km^2 and it was located at 500 m depth. It would contain vitrified high-livel waste from the 300 GW(e)·a nuclear programme with light-water reactors, authorized in Sweden by a 1975 parliamentary decision.

Because of the legal requirements, very cautious assumptions, models and data were used throughout the safety analyses. The results show calculated maximum dose rates that are within the limits recommended by Swedish Authorities for nuclear installations. The maximum levels of radionuclides in recipients (i.e. receiving bodies such as a lake or a well) were found to be comparable to the natural levels of radionuclides. The estimated doses were found to be less than the fluctuations in the natural radiation level.

5.5. CANADIAN SHIELD CRYSTALLINE ROCK REPOSITORY

The Canadian Disposal Concept is to immobilize the fuel waste by rendering it stable chemically and mechanically and emplacing it in a deep underground repository (see Fig.2) in a stable geological formation [10]. Since no decision has yet been made in Canada on fuel recycling, immobilization technology is being developed for two options: disposal of irradiated fuel itself and disposal of the separated wastes that would result from reprocessing Candu fuel.

Work by the Geological Survey of Canada (GSC) in the early 'seventies led to the conclusion that the geological hosts meriting highest priority for study in Canada would be igneous rock formations in the Canadian Shield known as plutons. (A pluton is an intrusive igneous rock formation which resembles an underground mountain.) Since most nuclear power production in Canada will be centred in Ontario for some time to come, the search for disposal formations is at present restricted to that province. Within the Ontario portion of the Canadian Shield, approximately 1400 plutons have been identified by the GSC.

The current programme for nuclear fuel waste disposal is in a Concept Assessment phase in which the objective is to carry out the research and assessment of the concept of geological disposal, without consideration of specific sites. If the results of the Concept Assessment phase are deemed to be satisfactory, this will be followed by site selection.

The general objective of the environmental assessment studies for nuclear waste disposal is to estimate the total effect on humans and the environment of the construction, operation and continued existence of all disposal facilities and auxiliary systems. The assessments are divided into two parts: the pre-closure assessment and the post-closure assessment. For the pre-closure phase the total impact of transportation, immobilization, emplacement and decommissioning are considered. Pre-closure assessment evaluates social and economic impacts, and radiological and non-radiological impacts on the public, workers and the environment. Most of the research and development has focused on the post-closure phase, the scope of this report. Post-closure assessment includes the evaluation of processes within the vault (i.e. the waste repository), the geological formation and the biosphere.

Detailed computer programs are being developed and applied for hydrogeological and chemical modelling. Hydrogeological modelling is carried out with finite-difference and finite-element codes for three-dimensional porous flow, heat and mass transport. Computer programs are being developed to calculate flow in interconnected rock fractures. They include routines for analysing field measurements statistically to prepare the fracture matrix input. The complex equilibria between solutions and solids are being analysed from two directions: chemical modelling programs, together with measured fundamental data, are being used for detailed calculations; and empirical relationships are being derived from experimental observations on representative systems.

Systems analysis programs link together the hydrogeological, chemical and mass transport processes within the vault, geosphere and biosphere, respectively.

The systems variability analysis code integrates the total system and samples data from distributions reflecting the uncertainty and variability in the data values. The resulting output is a histogram of consequence (dose to humans) versus probability, indicating the most probable consequence of a project, and other consequence estimates, together with their probability of occurrence.

The present status of the methodology development and application is described, together with results obtained to date.

5.6. BELGIAN CLAY REPOSITORY

In 1974, the decision was taken in Belgium to start research and development work on geological disposal of conditioned reprocessing wastes. This programme is managed by the CEN/SCK (Nuclear Energy Research Centre) of Mol. The safety analysis studies for post-closure long-term repository performance are still in progress and no comprehensive safety assessment report is currently available; the work is introduced below and discussed in Appendix F.

An inventory of potential geological disposal formations indicated that only clays and shales could be considered as potential host rocks in Belgium [26]. Several potential areas with underground clay and shale formations were identified. One of them is situated in the north-eastern part of the territory where the Nuclear Energy Research Centre (CEN/SCK) and some other nuclear industrial and research facilities are located. Thus, it was decided to focus the research and development programme on the Boom clay underlying the CEN/SCK site at Mol, because it is believed that this formation presents favourable characteristics for meeting the requirements for disposal of conditioned high-level and/or long-lived radioactive wastes.

The site-specific programme covers two main areas of investigation: concept development and safety assessment. Both are supported by modelling, field, in situ and laboratory investigations.

Safety analyses for two distinct scenarios are being developed. In one scenario (normal case) the future evolution of the repository, owing to its natural degradation, is considered. A second scenario (abnormal case) is based on a potential disruption of the geological and geochemical barrier function of the Boom clay. Both scenarios will be based on the specific repository concept that has been developed for disposal in the Boom clay at the Mol site [11].

The abnormal case study is now further developed than the normal case study and is being performed in close collaboration between CEN/SCK and the Joint Research Centre (JRC) of the Commission of the European Communities at Ispra. In this study the JRC methodology for risk assessment [27, 28] is being applied for the specific case of a Boom clay repository. The selection of the abnormal release scenario is made on the basis of a probabilistic fault-tree analysis [29] of how natural and geological events or processes, and non-intentional human activities, could breach the barrier function of the Boom clay formation. The first abnormal release scenario thus considers a faulting phenomenon triggering a release to the enveloping aquifers, and water flowing in a contaminated water plume. An ingestion pathway with direct consumption of the water, and an inhalation pathway due to re-suspension in the air of soil dust polluted by soil irrigation, are considered for estimating the dose rates to a most exposed individual. Other abnormal release scenarios, related to glacial erosion and non-intentional human intrusion, have also been examined, but are not discussed here.

The normal case study scenario is based on the loss of the engineered barrier functions provided by the waste package (waste form and container). The subsequent migration of radionuclides through the water-filled pore space of the clay formation is modelled, and releases in the overlying and underlying aquifers are estimated. For migration within the clay formation a three-dimensional specific model has been developed [30]. The pathways back to man to be considered in the normal case scenario will be similar to those of the abnormal case.

6. COMMENTARIES ON SAFETY ANALYSES

In any radioactive waste disposal programme, safety analysis methods play more than the single role of providing information for the final safety assessment required for repository licensing, construction and operation. As stated in Ref. [2], they are applicable for:

(a) Concept evaluation;
(b) Site evaluation;
(c) Repository design evaluation; and
(d) Repository system licensing.

The level of sophistication of the analyses and the quality of the results increase as the repository system design and development proceed.

Safety analysis studies may proceed from generic to site-specific studies. As the studies progress through the various stages, the data vary from scanty or non-existent to sufficient, and the uncertainty bands become narrower. The missing data (or design information), or broad range in the uncertainty band for the data, will generally require that the modelling or analysis vary from simple to more complex as the study type and purpose move from (a) to (d). The examples introduced and described in Chapter 5 illustrate studies of all but the last general type. With the exception of the KBS and Netherlands studies, the examples are of the first two types. The KBS and Netherlands studies represent more complete studies approaching the last type. The KBS and the Canadian studies illustrate the use of sensitivity analyses and validation methodologies that are informative for all types of study and for safety analyses for repository system licensing.

The diagram for safety analysis components and interactions shown in Fig.2 in Chapter 4 indicates a philosophy of scenario analysis followed by consequence analysis. Some more recent thinking indicates that scenario selection and consequence analysis may need to be more closely integrated and expanded to deal with the uncertainty question in a stochastic way, as is illustrated by the Canadian approach. In addition, analyses require close integration to study the interaction between processes and events that through time have the potential to affect the integrity of a repository (e.g., uplift alone may present no problem, erosion alone may present no problem, but uplift might lead to increased erosion which might lead to loss of integrity of the repository).

In the following sections of this chapter on scenario selections and repository, geosphere and biosphere analyses, the methodology requirements, data requirements, some sense of ranking and potential problem areas are discussed. Comments are based on the specific studies described in Chapter 5 and described more extensively in the Appendixes. Some possible directions for future efforts are also suggested.

6.1. SCENARIO SELECTIONS

To date little work has been published on a systematic review of potential release and transport scenarios. Such a review would examine a wide range of scenarios, considering their likelihood of occurrence and their consequences. Obvious difficulties arise in allocating probabilities to unlikely events as well as in quantitatively describing the resulting system state. In hard rock most attention has been concentrated on the highest probability, "normal" case of long-term corrosion and subsequent nuclide migration. In salt, where it is very difficult to postulate natural release mechanisms of significant probability, the analyst's desire to produce a calculable result has led to the inclusion of highly improbable scenarios.

In the examples reviewed, both the salt cases concentrate on single very conservative scenarios. Although they can be justified within the scope of the studies performed, unrealistic fracture events (as in the INFCE study) or high rates of uplift (as in the Netherlands study) should not figure as important scenarios in any final salt disposal safety assessments. The INFCE and KBS hard-rock studies cited do not include consideration of the consequences of low-probability release scenarios. The more recent Belgian clay study describes the consideration of a wider range of potential release scenarios in a manner which is also being applied now to other host-rock types.

There is a clear need to concentrate on development and analysis of realistic release and transport scenarios. In all safety assessments, consideration of a wide range of scenarios, together with their probabilities of occurrence, will increase understanding of the relative importance of events and processes considered. Comprehensive scenario lists are not possible for generic studies of the type most commonly performed to date since the characteristics of the specific site chosen will play a decisive role in improving the quantitative description of the system.

6.2. REPOSITORY ANALYSES

The scope of comments to be included in this section is set by the items included in the left-hand box in Fig.2. The broad aim of all modelling of the repository analyses is to produce a radionuclide source term for input to geosphere analyses or, in some cases, directly to the biosphere analyses. As the modelling requirements for repository analyses are discussed below, individual models describing specific parts of the system are emphasized. Clearly such separation is somewhat artificial and many interdependencies must be accounted for in the combined system. A simple example is the dependence of waste leach rates upon the water flows predicted by hydrogeological models. Currently, various groups are working at integrating models for use in iterative, intercoupled analyses.

6.2.1. Modelling requirements

For the repository part of the total analysis we have identified five components, the characteristics of which can affect the radionuclide source term which is required for subsequent use in geosphere or biosphere analyses. For each component there are various processes which can influence these characteristics and which must be capable of being modelled by the safety analyst. The repository components and processes are summarized in Table III.

Summarized below are some remarks upon the general importance and the status of the modelling areas indicated in Table III. It is obvious, however, that not all components are relevant for disposal in all host rocks and that the importance of the processes which can occur varies greatly with host rock. In the following sections the most important modelling areas are therefore highlighted for each host rock in turn. Comments are also included on the studies reviewed, with respect to the range of models employed and the level of these models.

(a) Waste form

A description of the form of the waste is clearly an important starting point for any analysis. The first of the various processes affecting the waste form and requiring modelling is radioactive decay. One should be able to predict, as a function of time, the nuclide concentrations, the heat emission and consequent temperature distributions and the radiation doses delivered to the waste matrix and the surrounding media. Examples of models used to treat these problems are the ORIGEN [31] and BEGAFIP [32] codes, which were used in studies reviewed in this report.

If fluids can contact the waste, leaching or dissolution processes should be modelled. Leach rates are affected by a wide range of parameters including geometry, physical state of the waste form, temperature, pressure, groundwater flow and chemistry etc. Current models do not handle all these effects.

TABLE III. REPOSITORY COMPONENTS AND PROCESSES TO BE MODELLED

Component	Processes
(a) Waste form	Radioactive decay – composition changes – heat production – radiation Leaching/dissolution Mechanical stresses
(b) Canister	Corrosion Mechanical stresses Radiation effects
(c) Buffer/backfill/structural material	Nuclide migration Heating effects Wash out Radiation Long-term chemical and structural changes
(d) Host rock immediately surrounding the repository	Heating effects Mechanical stresses Dissolution Nuclide migration Faulting Diapirism
(e) Repository seals	Long-term chemical and structural changes

Mechanical stresses can also affect the waste form if they can reach levels which influence its physical integrity, but the importance is generally lower than that of the processes mentioned above.

(b) Canister

For cases where a canister is expected to act as a protective barrier, the useful lifetime should be assessed. This can obviously be affected by corrosion, which depends upon the chemical environment of the repository. Mechanical stresses on the canister, whether built in or externally imposed, can also affect the integrity and lifetime. Direct irradiation effects upon the canister should be estimated, although they may be less important than indirect chemical effects caused by corrosive agents produced through radiolysis of water by radiation.

(c) Buffer, backfill and structural material

The function of the first two of these components, when included in a disposal system, is to control the access of chemically active agents to the canister and/or waste form and to retard radionuclides which may leave the waste. Accordingly, migration through the buffer/backfill should be modelled. The effect of heating, irradiation and long-term chemical changes should also be considered. The possibility of buffer properties being affected by chemical interaction with structural materials can also be important if such materials remain in the repository after sealing.

(d) Host rock immediately surrounding the repository

Processes which occur throughout the body of the repository host rock will normally be treated in geosphere analyses. The rock immediately adjacent to the repository, however, may be affected by processes such as heating or stressing caused by the presence of the repository, or may be affected by external processes such as faulting or diapirism in a manner which may need modelling.

(e) Repository seals

The direct connections of all repositories to the biosphere during the operational phase must be sealed off at closure. It is conceivable that long-term chemical or structural changes might lead to deterioration in the quality of such seals; the mechanisms for failure and also the possible consequences of failure should be considered for each host-rock type.

6.2.2. Comments on salt repository analyses

For salt repositories, in general, many of the modelling issues included in Table III are irrelevant, or of reduced importance, because of the absence of significant water in the repository in all the more probable scenarios.

In the repository design the component that needs most modelling is the waste form because this component defines the radionuclide source term and the decay heat development. Modelling will also be required to define a disposal geometry in which the maximum rock-salt temperatures will not exceed certain preset limits.

The other component requiring good modelling is the host rock surrounding the repository. Potentially important processes here are the displacements caused by thermal expansion and by repository closure due to plastic deformation of the salt. A further important process is the dissolution of salt which might occur owing to circulating groundwater contacting the salt formation.

The main differences between bedded and dome salt are related to the geometries of the formations and to the fact that salt domes usually contain smaller amounts of clay and brine. The presence of aquifers underlying a bedded salt formation can increase modelling requirements for the processes involving water; diapiric uplift, on the other hand, can normally be ruled out for a bedded salt repository.

The repository designs are based on the geological medium as the dominant isolation barrier. Other components such as canister, backfill and buffer material will therefore be less important areas for modelling. On the other hand, a failed shaft seal may short-circuit this geological barrier so that analysis of this possibility is necessary.

The Netherlands study described in this report defined the waste form, disposal geometry and leach rates. Because of the high rates of diapirism assumed, the source term derived from the repository analysis feeds directly into a biosphere analysis rather than the geosphere transport models which follow in analyses of other host rocks. As was mentioned earlier, such high rates are unrealistic since repositories would be located only in salt domes, whereby geological history indicates that a post-diapiric stage with low extremely upward movement is in progress. At maximum upward movements of around 0.1 mm/a or less, the most important process to be modelled may be slow dissolution of the top of the salt dome by groundwaters below the cap-rock.

6.2.3. Comments on hard-rock repository analyses

A hard-rock repository will contain some engineered barriers designed to isolate the waste from the surroundings. The primary purpose of these barriers is to delay leaching of the waste. Further, they should extend the dispersal of long-lived nuclides over a long period. These goals should be kept in mind when considering the modelling requirements for hard-rock repository analysis.

The release rates from a hard-rock repository to the geosphere are dependent on three main parameters. These are the lifetime of the waste canisters, the leach rate from the waste form, and the migration of radionuclides through the buffer material. The relative importance of these parameters is a function of the design of the repository and of the particular release scenario being modelled. The following comments on some of the general modelling requirements indicated in Table III reflect the experience gained in analyses aimed at establishing the relevant ranges of values for the above-mentioned parameters.

In hard rock, long-term corrosion and slow leaching by small quantities of groundwater is the "normal" scenario considered. Leach modelling should realistically account for the relevant environment of the waste. This means that leach rates measured in the laboratory with large quantities of water available should not be applied in normal repository conditions. In the Swedish study reviewed, this conservative assumption was made, although more realistic models were also considered; the INFCE hard-rock study with more realistic models arrived at times for complete leaching which are much longer.

The canister is an independent barrier which can isolate the waste during the initial phase of relatively high heat and radiation production. Modelling the corrosion rate of the canister must account for the groundwater chemistry in the repository. It is important to include consideration of how the choice of the buffer material composition adjacent to the canister will influence the water chemistry. The importance of modelling the production of corrosive agents by radiolysis depends upon the level of radiation penetrating the canister. Thick-walled canister designs, like those in the Swedish study, significantly reduce radiation levels.

In hard rock the buffer material can act both as a chemical and a mechanical buffer between canister and host rock. Migration can occur through flow of water in the buffer or through diffusion. Modelling migration through the buffer, as in the INFCE hard-rock study, is valuable for calculating transport of corrosion agents to the waste as well as transport of radionuclides from the waste. Since the buffer material can be affected by heating or irradiation, modelling of the temperature and radiation fields in the buffer can be important. Finally, the possibility of loss of buffer material through cracks or fault in the surrounding host rock should be considered. Extensive analyses of the properties and behaviour of bentonite as a buffer material have been performed in the Swedish work subsequent to the early studies reviewed in this report.

Thermal and mechanical influences on the host rock are important for hard rock because of potential problems with spallation from the rock surface and effects on hydrogeological characteristics. Comparisons between experimental and calculational results have shown that the prediction of temperature distributions in hard rocks is generally more straightforward than prediction of stresses, since the latter are more affected by details of the fracture systems in the rock.

Careful selection of repository sites in hard rocks can reduce to a very low level the probability of a major fault intersecting the repository, but the possibility will be considered in a complete analysis.

6.2.4. Comments on clay repository analyses

The two components of principal importance are the waste form and the host rock, the latter being of prime importance because of the properties of clay and the repository concept proposed. Depending, however, on release scenarios selected and on time-scales involved, other components may have to be considered. The interactions between the two principal components are also significant.

Clay contains a certain amount of interstitial water which, because of the low permeability, can flow only extremely slowly. Removal of radionuclides from the waste could thus be represented by a slow dissolution model. Modelling of the migration of dissolution and corrosion products is important. The chemical and physical properties of clay determine the diffusivity and retardation. Processes like heating, which can affect these properties, should be properly modelled. Clay repositories contain liners to provide structural stability during the operational phase, and their potential long-term effects on the chemistry in the repository should be studied.

Belgian work has dealt with some of the issues described above in analysing long-term deterministic release scenarios as well as some probabilistic "abnormal" release scenarios. The studies, however, represent a first approach, and thus simplified and over-conservative models are employed in some areas.

6.3. GEOSPHERE ANALYSES

Comments in this section on geosphere analyses deal with the three general areas shown in the left-hand side of the box labelled "Site analyses" in Fig.2. The broad aim of geosphere analysis methodology is to provide the methods for characterization and modelling of the geological, hydrogeological and transport processes of potential importance. Use of these methods then provides estimates of the locations, concentrations, times and rates of entry into the biosphere of any radionuclides leaving the repository.

6.3.1. Hydrogeological studies

6.3.1.1. Modelling requirements

The groundwater flow will have profound significance for any repository involving emplacement of waste in a non-salt geological formation. Consequently, great importance is attached to the topic of hydrogeological modelling.

In general, the phenomena that should be taken into account in modelling the hydrogeology in host rocks and their adjacent media include porous flow; fracture flow; diffusion and dispersion; thermal effects; and solute concentration gradients (e.g. brine gradients).

Important data required for porous flow hydrogeological modelling include permeabilities, hydraulic gradients and appropriate boundary conditions.

For fracture flow modelling the requirements include fracture distribution and dimensions, including width, orientation (dip and strike), spacing and degree of interconnection.

Measurements of existing, and inference of past, water-flow fields are of value for validation of computer models.

Particular attention should be directed towards interpretation of a limited amount of field information to produce a picture of the whole geological formation to be modelled. Statistical methods are probably appropriate to do this.

The influence of elevated temperature on the major hydraulic parameters of the geosphere should be evaluated, including the development of thermo-hydraulic gradients. For the hard-rock formations the thermomechanical modelling should also account for long-term crack growth and changes of hydraulic gradient due to thermal stresses.

6.3.1.2. Comments on salt repository analyses

For the case of a repository in salt, the hydrogeological modelling problem is essentially to predict the rate at which groundwater flows to the boundary of the salt formation, dissolves the salt and carries the solute away. If sufficient salt is dissolved to expose the waste to water, the problem becomes one of predicting how the water will dissolve the waste and carry it to the biosphere. It is necessary to model the effect of high salt concentrations on the hydraulic properties of the solution. For such modelling requirements the choice of a code is strongly influenced by the properties of the formations adjacent to the salt and the impurities in the salt itself. Generally a three-dimensional code capable of treating high solute concentrations (brine), and possibly heat through the use of a porous media approximation, would be selected. It is not always necessary, or sensible, to carry out a full three-dimensional flow calculation. For generic studies an approximate calculation may be more appropriate, using 'lumped' parameters such as effective average permeability and hydraulic gradient with some approximate representation of the location of streamlines. However, in order to give credibility to the parameters used, it is necessary that they be derived from real formations which are similar in structure to the formations assumed for the generic assessment. One way of deriving such lumped parameters is to model a real formation and adjust the programme and data until a good match is obtained between the predicted and measured results. The effective permeability can then be derived from flow through a selected region and hydraulic gradient across the region.

For the flow of water in the Netherlands domed salt study, a two-dimensional model was used, with changing boundary conditions caused by the upward movement of the salt itself. The model used for the INFCE salt study was a three-dimensional finite-element mock-up of the total site. The capability of the code used was well validated by comparison with a field situation. However, the scenario modelled is unrealistic because of assumptions of a maintenance of a flow path through salt when such a path could not, in fact, remain open. The return flow would become super-saturated owing to the geothermal gradient and would thus precipitate salt to cause the fractures to seal. Nevertheless, before this scenario is criticized further, its limited purpose should be kept in mind. It was only to be used to compare different fuel cycles, and thus any absolute estimates of consequences were unnecessary.

The results from the detailed model were then used to estimate averaged parameters for use in the MMT mass transport code. This approach, using detailed modelling to derive averaged parameters and then transport modelling using those parameters, seems to be most suitable and practical for the waste disposal problem.

6.3.1.3. Comments on hard-rock repository analyses

The major consideration for modelling groundwater movement in hard rock is fracture flow, which is a complex problem not yet treated comprehensively. Large fractures could exist and form major conduits for flow. The host rock will be interlaced with systems of minor fractures and even the rock itself has a certain degree of permeability. At the microfracture and porous scale the transport mechanisms could be diffusion-controlled.

Modelling flow in such a system requires a conceptualization or model of the system that permits practical mathematical analysis, adequately predicts quantities of interest, and does not require information which would be impractical to obtain from field observations. Insufficient information is available at present on flow in fractured rock to say with certainty which of several possible conceptualizations is adequate. The only one which has actually been applied so far to safety analyses is to approximate the fracture flow system with a porous flow model.

Similarly, as for the INFCE salt study, three-dimensional porous flow modelling was carried out in the INFCE hard-rock study to derive the hydrogeological parameters used in the GETOUT transport code. Again, this seems to be a very sensible and practical approach to the problem, keeping in mind that it still remains for continuing research and development studies to investigate the adequacy of the porous flow approximation.

For the Canadian study a range of hydrogeological parameters was used in a transport model called GARD [33], which is similar to GETOUT [34]. The parameters were derived from an interpretation of field measurements, expert opinion and some three-dimensional porous flow calculations with the SWIFT [35] code. The very wide range on the resulting estimations of the transit times to the surface reflects the fact that this was a generic study and a specific site was not used.

It also reflects the early stage of measurement and estimation of equivalent porous flow parameters for fractured systems.

6.3.1.4. Comments on clay repository analyses

There is some question as to whether there is any flow in clay at all. For plastic intact clays (not fissured or fractured) it is assumed that a porous flow model could be used.

The Belgian clay study involved two types of calculation. First, for the expected scenario a value of permeability was assumed for the clay and a simple one-dimensional Darcy flow approximation applied. This resulted in extremely low flow estimates as would be expected for clay.

In the second approach a faulting event was proposed to fracture the clay and allow leaching of the waste.

6.3.2. Radionuclide transport modelling

6.3.2.1. Processes

The migration of radionuclides in a saturated rock mass is governed by several mechanisms such as convection, dispersion and retention. In addition, radioactive decay must be included since it causes reductions in the concentrations of radionuclides in the medium and also a generation of new nuclides along the flow path. Migration of the decay products will take place according to their own retention properties and needs to be considered in the transport models.

Convection is the movement of dissolved radionuclides with the average velocity of water. Its description requires the knowledge of the flow pattern within the rock mass which, in turn, can be influenced by temperature and other changes which affect the density of the fluid such as macro-solute concentrations (e.g. brine concentration).

Dispersion is the combination of molecular diffusion, which can occur even under the condition of no flow and of hydrodynamic dispersion. Hydrodynamic dispersion represents the effect of the variation of the local velocity of water in the medium with respect to its average value as described by the convection. This variation of velocity exists at any scale, from microscopic (in the pore) to macroscopic (owing to heterogeneity of the medium) and even megascopic (owing to large-scale variations in the rock properties, such as fractures). Dispersion causes mixing and spreading (longitudinally and transversally with respect to the flow direction) of the transported elements.

Under conditions of no flow, molecular diffusion is the only mechanism generating movement of elements through the rock mass. However, if temperature gradients exist, thermal diffusion (Soret effect) can increase this migration.

Retention is a general term covering all kinds of interaction that can take place between the transported elements and the rock matrix. A non-exhaustive list could be:

Filtration;
Molecular diffusion of elements into immobile water (e.g. dead-end fracture, matrix porosity adjacent to the fracture ...);
Ion exchange (electrostatic binding of ions on the surface of the solid);
Chemical reactions between radionuclides, other elements transported in the water and the rock matrix; these processes are time dependent (kinetics of the chemical reaction);
Precipitation/dissolution;
Flocculation.

Some physical mechanisms (filtration, diffusion) can be treated separately for each element since there is no interaction or competition between these mechanisms. Some of the other processes, generally of a chemical nature (e.g. ion-exchange reactions, precipitation) must be modelled simultaneously for all elements present in the water as these mechanisms can interact and compete with each other.

So far, these basic phenomena are often not specifically distinguished in models, their total effects being empirically represented by a distribution coefficient referred to as K_d, which assumes that there is a constant ratio between the amount of radionuclides retained on rocks and the amount in solution. K_d is defined either per unit mass of the rock or by unit area of fractures and can be supported by tracer tests in situ or from batch, column and water press experiments in the laboratory. Very seldom do the two types of measurement match, the tracer tests giving generally smaller values than the latter. The use of the K_d implies that each element migrates independently of the others, under a unique chemical form (e.g. ion of a given valence, or complex molecule electrically neutral) and that retention is instantaneous and reversible. Improvements have introduced a kinetic factor in the modelling of retention. Consequently, it must be pointed out that chemical exchange phenomena are not the only phenomena involved in the retention factor.

6.3.2.2. Models

Most models developed so far solve numerically the so-called "dispersion equation" which gives, for each radionuclide, the variation with time of its concentration, C, in the water in relation with its concentration, F, retained by the solid. If the medium is porous and flow takes place throughout it, F is defined as the mass of element retained per unit mass of rock; if the medium is fractured, F is defined as the mass of element retained by unit area of fracture.

Methods of solution of the equation in one, two or three dimensions include:

Analytical solution;
Finite differences or integrated finite differences;
Finite elements;
Method of characteristics (particle in cells, random walk) which solve separately the convective and dispersive terms, the latter being modelled by the movement of a large number of particles through the medium.

Recent theoretical developments as well as field experiments have shown that the dispersion equation only very roughly approximates the transport process [36, 37]. It appears indeed that the variations of velocity which are represented by dispersion seem to vary with the average travel distance of each element, which cannot be accounted for in the usual formulation, and that a new type of equation, where the velocity variations are treated in a stochastic framework, should be developed. Such an approach is still in its infancy.

6.3.2.3. Data requirements

Convection requires information about the flow paths and velocities along these flow paths. They can be either measured in situ, or given by hydrological modelling.

Dispersion requires the measurement of the molecular diffusion coefficient in the medium and data on hydrodynamic dispersion. In the classical approach a single dispersion tensor is defined; coefficients are often assumed to be linear functions of the velocity. These coefficients can only be measured by performing tracer tests in situ, because values measured in the laboratory on cores are one or two orders of magnitude smaller than that measured in situ. Even so, these coefficients are known to be a function of the length of the travel distance of the tracer during the test, which is one of the drawbacks of the usual theory [38]. Argument also goes on about whether the dispersion coefficient is unique for a given medium, or whether it is also a function of the element transported [39].

In the new stochastic approach the information required is the probability distribution function of the velocity field in the medium or, at least, some of its moments (mean, variance, co-variance). Such a set of data has not yet been obtained in a real situation but could be sought.

For *retention,* even if each mechanism could be modelled precisely, the data requirements can be tremendous, especially for chemical speciation, chemical reactions, including precipitation and selective ion-exchange capacity. If this approach is used, the Eh and pH and the natural geochemical equilibrium of the water in the medium are needed, as well as a complete mineralogical description of the rock and numerous chemical reaction constants.

If the simple K_d approach is used, the equation becomes:

$F = K_dC$, if retention is instantaneous, and

$$\frac{\partial F}{\partial t} = K(K_dC - F), \text{ if linear kinetic is assumed.}$$

Data needed are K_d for each element and possibly kinetic constants, K, which can be different for sorption and desorption.

Boundary and initial conditions are also needed for the transport equation. Boundary conditions describe the physical processes occurring at the end of the rock mass (e.g. flow into a lake, river, shallow aquifer). Initial conditions describe the composition of the water in the system before any release of radionuclide occurs.

6.3.2.4. Dimensional factors

The transport pattern in real media is three-dimensional and ideally should be modelled in that way. But, as three-dimensional modelling is expensive and limited in terms of mesh refinement, two- or one-dimensional models may be sometimes preferred in the initial investigative stages, when other uncertainties preclude the need for the refinement that a more detailed three-dimensional model can provide.

(a) Two-dimensional models are adequate if a symmetry can be found in the system (e.g. radial flow, parallel flow); otherwise they slightly underestimate transverse dispersion.
(b) One-dimensional models are seldom representative of real field situations, and they grossly underestimate transverse dispersion (i.e. the calculated concentration will be higher than the real one, but the real outlet area will be much larger than predicted).
(c) One-dimensional models are, however, very useful for generic studies and sensitivity analysis, as they permit running inexpensively a large number of tests.

6.3.2.5. *Discussion*

In all the study examples described in Chapter 5 it should be noted that:

The dispersion equation was assumed valid, with a constant dispersion coefficient; the values used reflect either laboratory data (small dispersion) or average field condition (large dispersion);

The modelling was in one dimension; and

The simple concept of a linear isotherm with instantaneous sorption (the K_d approach) was used. (The K_d values for each radionuclide were determined from laboratory measurements.)

6.4. BIOSPHERE AND DOSIMETRY ANALYSES

Current models for estimating radiation doses to individuals of population groups from internal and external sources of radiation consider various exposure pathways ranging from drinking water to consumption of fish, meat, milk, vegetables, and to direct external radiation. These models can also be used to estimate collective dose commitments. It should be realized that these environmental and dosimetric models were developed to assess the consequences of radionuclide releases from nuclear facilities in the present time. The level of detail of the models reflects the extensive knowledge about the present environment, the present population and their interactions.

In relation to underground waste repositories, it should be kept in mind that release of radionuclides into the environment is expected to take place in the very distant future. There is no possibility of predicting reliably the environmental conditions and the population distribution over such a long period. In addition, the uncertainties are compounded by the fact that the long-term development of the human species and its living habits are unknown. A possible way to overcome the difficulty of predicting future conditions for the environment and humankind is to analyse a variety of potential futures.

The necessity of providing a method for assessing the possible consequences of future releases from underground repositories is obvious. But it is doubtful that detailed biosphere models are needed to analyse the radiological results of releases from underground repositories of radioactive wastes. Therefore, existing environmental and dosimetric models are considered more than adequate for the purpose of safety analysis of radioactive waste repositories; the only modifications that might be needed are to account for long-term effects of slow environmental processes.

In all the safety studies covered by this report, the outcome of the analyses resulted in expressions of annual doses for individuals of a population group living in the proximity of the repository. In some cases the collective dose commitment was also calculated. In one of the studies comparisons were also carried out with natural levels of ^{238}U and ^{226}Ra in certain environmental compartments and with variations in natural background radiation. Only in the Canadian study were probability distributions of doses derived.

As previously indicated, to complete the safety assessment the results of safety analyses need to be compared with acceptability criteria; at present, generally accepted criteria have not been established. Although acceptability criteria, per se, are outside the scope of this report, some comments are necessary to indicate the type of information needed from safety analyses to complete a safety assessment.

A Canadian report [10] presents the following alternative bases of comparison on which to judge the acceptability of a future release from an underground repository:

(a) The predicted dose to individuals in the most exposed group should not exceed some specified fraction of ICRP recommended levels [41];
(b) The predicted dose to individuals in the most exposed group should not exceed some specified fraction of natural background dose;
(c) The predicted radionuclide levels in the environment due to the disposal operation should not exceed some fraction of the natural radionuclide levels;
(d) The predicted long-term impact (dose to man) of the disposal operation should not exceed the predicted impact (using similar methods) of the ore which would have remained in place if it had not been mined for uranium.

None of these alternatives considers the issues of how to evaluate very low-probability events that might affect the behaviour of a repository or what weight should be assigned to doses arising in the near and distant future.

Dose limits cannot be simply applied to abnormal events affecting underground disposal, because it is usually possible to identify a scenario (even if it has a very low probability of occurrence) for which calculated individual doses exceed a selected limit. It will probably be necessary either to specify the scenarios to which dose limits should be applied (e.g. so-called "normal" cases) or to develop criteria which take probabilities explicitly into account (e.g. by framing criteria in terms of risk, where risk is defined as a combination of the probability that a dose will occur and the probability that the dose will give rise to harmful effects). If collective doses to future populations and collective dose commitments are calculated, the uncertainties associated with the calculated values will be very large. The range of values may be too large for optimization and realistic comparisons of disposal options. For this reason judgement must be used regarding the extent to which future collective doses are considered in acceptability decisions.

The issues discussed briefly above are matters to be resolved by the appropriate national and international authorities. In the absence of established acceptability criteria, safety analyses should provide as much information as possible for decision-makers about predicted doses and their probabilities, uncertainties, and timing. This information could also include comparisons with natural background doses, background radionuclide concentrations or the predicted radiological impact of, for example, uranium ore bodies.

6.5. UNCERTAINTY AND VARIABILITY

The uncertainties in repository modelling are of three different basic types: (a) random variabilities due to "random" variations in nature, (b) uncertainties due to insufficient or inadequate data, and (c) uncertainties due to the use of inadequate or wrong models. The first two kinds of uncertainties and variabilities can be accounted for by stochastic methods. The third kind of uncertainty will, however, also affect the results of any sensitivity analysis in an unknown way. One way around this problem, which has been frequently used, is application of so-called "worst-case" models. However, this approach has drawbacks, and the problem can best be resolved by validating computer codes by comparing predictions with laboratory and field data.

Current methods for safety analysis of underground disposal are typically deterministic in nature. Several efforts are now being made to confront the uncertainty analysis question with a broader class of stochastic methods. The Canadian system variability analysis represents a Monte Carlo approach which utilizes multiple runs of simplified integrated deterministic models in order to develop a probability distribution of the results from a safety analysis.

In particular, the output format from the SYVAC code [15] is of interest. The process of sampling sets of parameters defines a large set of scenarios. These scenarios represent the whole variety of scenarios possible within the wide ranges of uncertainty in the parameters describing the system. The analysis of this large number of scenarios (over 1000) results in a histogram of consequence estimates versus the frequency of occurrence of those estimates. This is then a clear expression of the distribution of consequences from all situations, from the most probable to less probable, more severe ones. This approach thus gives a more balanced and complete description of possible consequences than the so-called "worst-case" analysis. It also facilitates sensitivity analysis and development of site selection and design criteria and guidelines.

Other methods now under development, but not at the same current state of advancement as the Canadian effort, are attempting to solve the stochastic partial differential equations which describe hydrological flow and transport in geospheric systems. There are also model development efforts to produce stochastic models to deal with the process events and time history difficulties associated with the development of realistic release scenarios.

In probabilistic analyses of uncertainties it is recognized that special care must be given to the actual existence of a probability distribution for the result of the analysis. This is especially true for Monte Carlo analysis, where the complete distribution computed experimentally can be simply a function of the number of runs, if the phenomenon under study is not stationary.

Stochastic models require estimates of the parameters and the probability distributions of these parameters, or variance, in order to be meaningful. Kriging methods [42, 43], developed by the French, provide a potential solution to this problem, at least for certain geospheric parameters.

6.6. SENSITIVITY STUDIES

Sensitivity studies evaluate the influence on the output results of changing particular input parameters. This is somewhat different from uncertainty analysis which evaluates the influence, on the output, of uncertainty in the input values. Sensitivity analysis should be performed in conjunction with uncertainty analysis.

Sensitivity studies can identify which parameters should be accurately defined and which thus may merit more detailed research attention. They can be used to guide further site characterization or other data gathering or design efforts.

Some design parameters could be identified as having a significant influence on the safety of the disposal facility, thus leading to the development of design criteria. Similarly, sensitivity analysis applied to site parameters could lead to site selection criteria.

Caution should be exercised in performing sensitivity studies for sub-systems only, in that some sub-systems responses may vary widely with certain parameters without having any significant impact on the predicted overall safety of the waste disposal system.

The extent to which sensitivity studies were carried out in the example cases varied. The KBS study was probably the most systematic and comprehensive. The KBS effort represents a thorough classical sensitivity analysis approach where the effects on safety of various bounding parameter variations were tested.

6.7. VALIDATION EFFORTS

Validation efforts are an extremely important part of any safety analysis. It must be demonstrated in some convincing way that the models predict that which is observed in the field and laboratory. It must also be demonstrated that the hydrological and transport models, constructed for the overall site as it exists today, reasonably represent this system. Some of the processes identified in the area of repository analyses can be validated against laboratory experiments. However, complete experimental validation of long-term behaviour is not possible; nevertheless, useful information can be gained from accelerated tests of various kinds and occasionally from natural systems where comparable processes have taken place.

The long time-frames of interest make model validation efforts difficult since only indirect methods can be used for this validation. One method used in the KBS studies was the comparison of model-predicted water dates with measured water dates. In the KBS studies the ability to model transport processes involving retention was demonstrated in field studies. The other studies discussed, being more generic and limited in nature, did not deal with the validation question. More direct or indirect methods which will allow safety analysis models and methods to be validated are important goals for research.

7. CONCLUDING COMMENTS

This is an introductory report on concepts and methods useful for safety analyses of deep geological radioactive waste repositories. Numerous studies, not included here, have been made while this report has been under preparation (e.g., see Refs [43–46]), and the reader is also encouraged to study these references. Nevertheless, it is hoped that the following comments of the Agency's Advisory Group will be beneficial since the methodologies for safety analyses are continuing to be used and improved.

1. The studies reviewed in this report were originally conducted for different purposes and, while great caution is required in making comparisons between them in terms of approaches or results, they are useful to illustrate the safety analysis methodologies for underground repositories.

2. Safety analyses are necessary steps in the iterative process going from repository concept evaluation to repository system licensing; the samples reviewed here are considered as early steps in such a process.

3. In generic safety analyses there is the tendency to make over-conservative assumptions about release scenarios and parameter values; this conservatism has resulted in a number of analyses which may be unrealistic. The situation is expected to improve as more realistic scenarios and input data derived from host-rock and/or site-specific investigations are utilized, and as safety analyses are extended to include the probabilities of occurrence of scenarios, as well as their consequences.

4. Many input data are characterized by significant uncertainty and variability, especially as a function of time. It is necessary to evaluate systematically the influence of these uncertainties and variations on the outcome of safety analyses.

Sensitivity analyses are required to establish the relative importance of different parameters and phenomena considered in safety analyses, and to give guidance on priorities in definition of research directions and data acquisition. In addition, sensitivity analyses can help in developing guidelines for repository system site selection and design.

5. Detailed models of components of the disposal system need to be validated by comparison with laboratory or field data. Similarly, parameter distribution functions, which reflect variability and uncertainty should be obtained, and it should be shown that they provide a good representation of the actual distributions of such parameters.

6. Modelling of the release of radionuclides from a repository, through the geosphere to the biosphere, has to account for a large range of phenomena. The relative importance of these phenomena is largely dependent on the chosen host-rock formation, on the site, and on the specific repository design. Different sets of models are required to cope with all relevant phenomena for the different types of repository illustrated by the examples.

7. The results of safety analyses of underground waste repositories need to be compared with acceptability criteria in order to complete a safety assessment. Generally accepted criteria have not yet been established, but the development of criteria is under active discussion by national and international authorities. In the absence of established acceptability criteria, safety analyses should provide as much information as possible for decision makers about predicted doses and their probabilities, uncertainties, and timing. This information could also include comparisons, for example, with natural background doses, background radionuclide concentrations or the predicted radiological impact of uranium ore bodies.

8. The examples reviewed here illustrate that safety analysis work, supported by laboratory and field data, can offer a practical approach to post-closure safety assessments of nuclear waste repositories in deep continental geological formations.

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Appendix A

INFCE SALT REPOSITORY

C. Cole

1. INTRODUCTION

The INFCE studies evaluated the relative safety impacts of seven different fuel cycles, including disposal of spent fuel in some cycles. For simplicity of presentation, and because the purpose of this document can be met in doing so, the example described is for only one of the fuel cycles, a LWR fuel cycle with plutonium recycle.

An integrated modelling system was used in this study to examine the potential consequences of a postulated release of nuclides from a hypothetical geological repository located in a salt formation. For purposes of the INFCE studies, all radioactive wastes from the fuel cycle (except mining and milling) were placed in the geological repository.

The objective of the modelling was to predict the transport rate of radioactive contaminants from the repository through the geosphere to the biosphere and to estimate the potential dose to humans so that the release impacts could be evaluated [1].

Currently available hydrological, transport, and dose models were used for this study. The hydrological model defines water-flow tubes and travel times from input describing the hydrological system and the disruptive event to be analysed. The transport model uses the output from the hydrological model and radioactive release source terms to describe the movement of the contaminants through the geosphere, and provides release rates and concentrations of radionuclides in the fluids released to the biosphere (in this case, to a surface-water body). This output then serves as an input to the dose model, which provides the estimate of the environmental dose resulting from the radioactive release.

Several hydrological models of varying complexity are operational and can assess a wide variety of groundwater flow systems, from very simple one-dimensional, homogeneous isotropic, single-layer flow conditions to the most complex three-dimensional anisotropic, multilayer aquifer case. To model most efficiently the multiple layers defined for this study, a three-dimensional finite element model was used [2].

Fewer groundwater transport models are available, but they also span a range of simple to complex conditions. A one-dimensional approach using the multicomponent mass transport (MMT) model was used in this analysis [3]. This approach, when used along actual flow tubes, gives a good approximation of two-dimensional transport.

Few environmental consequence models (dose models) exist which consider radioactive dose (both external and ingestion exposure) from the water pathway. One of these is a set of computer codes, ARRRG/FOOD [4, 5], which calculate the annual radiation dose and long-term dose commitments to the total body and selected organs of individuals and population groups from both internal and external sources of radiation. A shorter version of these codes was developed for the work conducted for the Advanced Waste Management Studies [6]. This shorter version was adopted for use in this study. The final results from the dose model are the radiation dose rates (mrem/a) to the most exposed individual resulting from 50 years' accumulation followed by 50 years of exposure to postulated releases of radioactivity from the repository.

2. PHYSICAL DESCRIPTIONS

2.1. Host rock site

The generic salt formation fo the INFCE salt repository was formulated by a step-by-step method, using data on the occurrence and stratigraphy of well-known domed and bedded salt deposits. Thus, the stratigraphic cross-section may be equally representative of either domed or bedded salt. The methodology used to determine the reference stratigraphic section was as follows. First, from the literature and from knowing the depth and thickness requirements from a repository, real stratigraphic sections from representative regions were obtained. Next, for each potential region of interest the real stratigraphic sections were combined into a composite stratigraphic section representative of a given region. Finally, the reference stratigraphic section was obtained by subjectively integrating the composite sections. Figure 3 shows the stratigraphic section for the INFCE salt formation.

The reference salt section is assumed to be located in an undeformed, or only slightly deformed large geological sedimentary basin. The top of the salt structure lies 250 m below the surface, and the structure extends a great distance (several hundreds or thousands of metres) below this point. The repository is located at a depth of 600 m with 350 m of salt above. The strata overlying the repository are nearly flat, lying in such a way that the recharge areas are not topographically much higher than the discharge areas; therefore, the regional groundwater gradient is low, around 1 m/km. In addition, the repository is assumed to be situated at a sufficient distance from any point at which the site groundwater flow system discharges to the biosphere, or is used by humans. The aquifers connect to a river, 6.2 km from the repository, with a flow rate of 500 m^3/s.

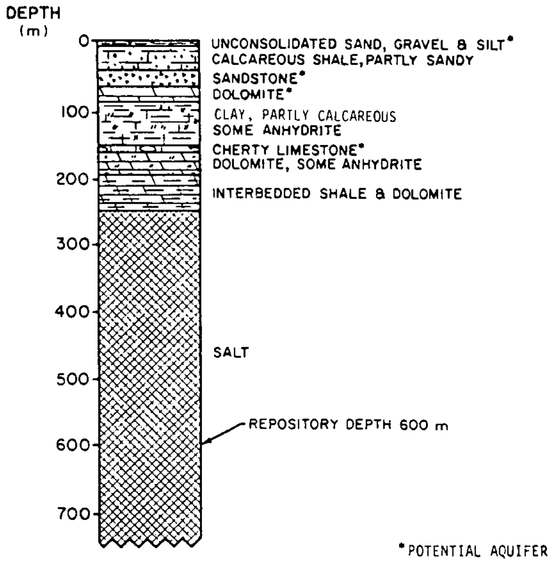

FIG.3. Representative stratigraphic section for salt.

The reference section contains a sequence of clastic or evaporative rock formations with varying permeabilities and porosities. These water-bearing formations contain and transmit various quantities of groundwater, depending upon the regional gradient and certain hydrogeological parameters such as porosity, permeability, and formation thickness. The values for these parameters vary considerably from site to site. The ranges of values obtained from the literature search and past experience are listed in Table V.

2.2. Repository and inventory

For this study the repository size is based on the area required to dispose of the wastes produced by a nuclear economy with a capacity of 100 GW(e), operating at this level for one year. Under this assumption the area covered by the repository would be about 30 ha; this would require about 15 tunnels for storing HLW, MLW and LLW.

The assumed waste packages are stainless steel canisters (30 cm i.d. and 300 cm long for HLW), steel canisters (86 cm i.d. and 115 cm long for cladding wastes), and 200-litre carbon steel drums, shielded (20 cm concrete) or unshielded,

TABLE V. PARAMETERS FOR GENERIC SALT STRATIGRAPHY[a]

Rock type	Hydraulic conductivity: Reference section (cm/s)	Hydraulic conductivity: Range (cm/s)	Porosity (%)
Unconsolidated sand, gravel, and silt	6.8×10^{-3}	$1 \times 10^{-7} - 1 \times 10^{-2}$	20
Calcareous shale, partly sandy	1.2×10^{-5}	$1 \times 10^{-10} - 1 \times 10^{-3}$	13
Sandstone	8.1×10^{-5}	$1 \times 10^{-10} - 1 \times 10^{-2}$	20
Dolomite	8.1×10^{-5}	$1 \times 10^{-10} - 1 \times 10^{-2}$	20
Shale, interbedded with clay, partly calcareous	1.2×10^{-5}	$1 \times 10^{-10} - 1 \times 10^{-3}$	13
Cherty limestone	5.8×10^{-5}	$5 \times 10^{-8} - 1 \times 10^{-3}$	20
Dolomite, some anhydrite	5.8×10^{-5}	$5 \times 10^{-8} - 1 \times 10^{-3}$	20
Interbedded shale and dolomite	6.8×10^{-6}	$1 \times 10^{-10} - 1 \times 10^{-3}$	20
Salt	2×10^{-18}	$2 \times 10^{-21} - 1 \times 10^{-8}$	0.5

[a] Figure 3 shows the representative salt stratigraphy assumed in this study.

for the other MLW and LLW wastes. The HLW is solidified as borosilicate glass, and the other wastes are immobilized in concrete. Decommissioning wastes are also included. The physical inventory of waste packages, representing 100 GW(e)·a of energy, is summarized below:

Unshielded drums	262 600
Shielded drums	71 200
Cladding waste canisters	4 300
HLW canisters	2 900

After emplacement of the waste packages, the tunnels (and holes) are backfilled with excavated salt.

The total inventories and isotopic distributions and concentrations for each nuclide in the wastes were calculated by the ORIGEN code [7].

3. RELEASE SCENARIO SELECTIONS

3.1. Normal scenario

The repository site would not be expected to experience a disruptive event that would release radionuclides to the biosphere within several million years. Solid materials buried in salt generally will not move significantly by themselves, notwithstanding the thermal conditions in a well-engineered HLW repository [8]. Without flowing water only solid-state diffusion can move the radionuclides from their point of burial. However, solid-state diffusion processes are very slow. This is known intuitively from the fact that ores in vein deposits have not diffused into the surrounding rock by measurable amounts in millions to tens of millions of years unless other driving forces were present. Generally a diffusion mechanism is inconsequential at temperatures below half the melting point of the material. Nevertheless, to provide perspective, the movement of one waste constituent through a continuous dense rock stratum was modelled.

A conservative estimate (fast movement) of the diffusion coefficient was found to be 1×10^{-10} cm²/s. Based on an assumed distance between source material and aquifer of 100 m, at least 500 million years would be required for the radionuclides to migrate to the aquifer. If the diffusion coefficient were 1×10^{-13} cm²/s, the minimum time would be 5×10^{11} years. With the reference repository, the radionuclides would have to diffuse through 350 m salt and 190 m shale/dolomite/clay before reaching the aquifer. For a diffusion coefficient of 1×10^{-10} cm²/s, this would require about 3×10^9 years, or approximately the age of the earth.

3.2. Abnormal scenario

Only one abnormal scenario was modelled. The design of reasonable scenarios which would expose the waste to aquifer fluids is extremely challenging because of their improbability. It should also be pointed out that any scenario modelled needs to be weighted according to its probability of occurrence.

In developing a postulated abnormal release scenario, an attempt was made to define one that would represent the worst possible release of contaminants while still being reasonable in terms of the release mechanism. The worst case would be one that provides maximum water flow through the repository area, since this would result in the greatest leaching of the waste and the quickest entry of the waste into the aquifer system. Given the geological structure defined previously, a reasonable mechanism allowing for flow through the repository is difficult to envision since the site description assumes no aquifer systems below the salt zone. As a result, flow through the repository area must occur by re-routing aquifer fluids from above the repository area. If one consideres the aquifer fluid to have no salt content and the salt's overlying rock (cap-rock) to be fractured, then salt dissolution must be modelled along with the associated cyclic solution mining of the salt and subsequent collapse of the overburden materials and resultant change in local permeability.

The questions to be answered for this kind of scenario are: How much fluid would be flowing past the salt surface, and how long would it take to dissolve the 350 m salt layer above the repository? This dissolution process can be estimated from determination of the maximum rate at which the water could be flowing past the salt's surface. A conservative estimate of flow rate was made by considering that the salt and cap-rock were fractured in a zone through the repository 2400 m wide to a depth of 450 m into the salt and of infinite extent parallel to the river. This extremely wide fracture zone was then assigned the same hydraulic properties as the limestone-dolomite-sandstone aquifer (hydraulic conductivity 8.1×10^{-7} m/s). The flow from the aquifer system into this fracture zone over the repository 1200 m length would be approximately 9.5 m^3/d. At this maximum rate it would require approximately 1.35 million years to dissolve the 1200 by 1200 by 350 m overburden of salt before any water reached the repository, if the aquifer fluid originally contained no salt.

Dissolution of the salt as a result of shaft failure, or some more reasonable fracture of faulting of the salt cap-rock, would be much slower because the initial flow rate of fresh water past the salt would be much lower than this maximum rate; moreover, the flow rate from the fractured cap-rock or failed shaft seal would be greatly retarded by the large density gradient between the fresh water in the limestone-sandstone-dolomite aquifer and concentrated brine in the shaft or fractured cap-rock. The resultant movement of brine from the shaft or fractured cap-rock would be more realistically determined by molecular diffusion (10^{-6} to 10^{-5} cm^2/s), or by second- and third-order flow effects arising from fresh water flowing past a hole or fracture filled with a dense brine.

Because of these considerations, the salt dissolution scenario was discarded, and the fracture scenario described above for estimating worst-case flow was chosen as representative. This worst-case fracture scenario allows the aquifer fluid to contact all the repository waste from the onset of the fracture process and allows the effects of maximum flow through the repository from the onset of the fracture under study. For the fracture scenario it was assumed that the fluid in the limestone-dolomite-sandstone aquifer was a concentrated brine so that effects of retardation because of density gradients could be neglected. The somewhat unrealistically wide fracture zone can be thought of as resulting from a multiple fracture pattern through the repository area, or from the collapse of an undetected solution pocket beneath the repository.

4. REPOSITORY ANALYSIS

The scenario modelled assumed failure to occur as soon as the repository is sealed (50 years after building) and all repository wastes to be exposed from the onset.

To simplify the analysis, no engineered barrier material was assumed to exist except the waste form itself. Accordingly, all nuclides were assumed to be in a typical unencapsulated borosilicate glass. Because of insufficient data, it was assumed that once exposed to brine, all waste leached (dissolved) at the same rate. Data availability did not allow the sophistication and detail required to analyse the behaviour of each type of waste form.

A leach-rate model was developed to determine the length of time and the rate at which the contaminants would be carried, via the aquifer water, through the fractured salt and repository and into the upper aquifer system. The leach-rate model was formulated using preliminary data on waste leach rates and the observed effects of flow rate and temperature on leach rate. The data from leaching experiments on waste glass indicate that the specific leach rates vary from nuclide to nuclide, and with caesium in solid and crushed HLW glass, for example, at 25°C they were found to be 4×10^{-7} and 4×10^{-9} g-leached/cm^2 of surface area per day, respectively. The data also indicate that for rates greater than one flush annually the specific leach rate is a function of the flushing rate. At less than one flush annually, the rates are essentially constant.

The data available on temperature effects indicate that specific leach rate versus temperature follows an Arrhenius-type relationship. Data available at T = 25°C and T = 250°C allow determination of the constants in this relationship. Leach-rate data comparisons between solid and crushed high-level wastes indicate that, although leach rate depends on actual surface area, the rate is not directly proportional to the surface area because the internal surface area within the mass of crushed waste leaches more slowly than the outer surface area. This is because of the interference between the boundary layer concentration gradients between adjacent waste particles.

For this initial leach-rate model the highest specific leach rate was used for all nuclides. The entire waste, including all drummed wastes, was assumed to leach at this rate. The glass was assumed to be crushed, and the crushed glass particles were modelled as spheres. Leach rate at any time, t, was calculated as follows:

$$LR\ (g/d) = SLR(T) \times SA(t) \times WSP \qquad (1)$$

where

t = time (d)

SLR(T) = specific leach rate (g/cm^2 of surface area/d) and is dependent on the temperature T

WSP = mass of one of the spherical glass particles
T = temperature of the glass at any time, t, in °C
SA(t) = surface area-to-mass ratio for a spherical waste particle, initially 0.424 cm^2/g.

The glass temperature is assumed to be proportional to the heat generation rate of the glass, and the heat generation rate is determined from a power function fit of heat generation rate (HGR) for spent fuels. Equation (2), the power function used,

$$HGR(t) = 7468\ t^{-0.723} \tag{2}$$

gives results close to published heat generation rates. The temperature is calculated by assuming that at t = 10 years, the crushed waste in the breached canister will be at the design canister temperature of 375°C, and at 1 000 000 years the waste will be at a background or ambient temperature of 50°C. Temperature at any time is thus calculated.

SLR(t) is assumed to follow an Arrhenius relationship which states that ln(SLR[T]) is proportional to the inverse of the absolute temperature. With experimental data at two temperatures, the leach rates at other temperatures are calculated.

The specific surface area of the glass (SA), based on a spherical model, is given by Eq. (3):

$$SA(t) = \frac{3}{R(t) \times D} \tag{3}$$

where

R(t) = radius of a spherical particle, initially 2.14 cm
D = density of glass, 3.3 g/cm^3

The weight of a spherical particle of glass at any time t is given by Eq. (4):

$$WSP = \frac{4}{3}\pi R(t)^3 \times D \tag{4}$$

The leach-rate model uses the above equations to calculate the fraction of the waste remaining in the repository at any time, t, by a simple rectangular integration scheme that determines the current radius of the glass particles. At the time t_0 when the leaching begins, the glass is assumed to be composed of equal-sized spherical particles. The values for T, SA, SLR, and WSP are calculated for time t_0 to determine the current rate, LR, at which glass is being leached from the surface of each spherical particle. This rate is assumed to hold for some time

Δt, thus dissolving some weight of glass from the particle, $WD = LR \times \Delta t$. From this leached weight the decrease in radius, DR, of the fuel particle is then calculated from Eq. (5):

$$DR(t) = R(t) - \sqrt[3]{R(t) - \left[\frac{WD(T)}{D \times 4/3 \times \pi}\right]} \tag{5}$$

$R(t_0 = \Delta t)$ is now $R(t_0) - DR(t_0)$ and the process is repeated until $(4/3\ \pi R(t)^3)/(4/3\ \pi R(t_0)^3)$ is <0.001, at which time all the waste is dissolved.

With the above assumptions and suitable corrections for waste temperature changes, half the waste was found to dissolve in a period of 3000 to 4000 years in a flow of 190 m^3/year.

5. GEOSPHERE ANALYSIS

5.1. Hydrological models

5.1.1. Three-dimensional finite element groundwater flow model

A three-dimensional finite element groundwater flow model [2] was used in this study to define water flow paths and travel times. This model was chosen for use because it has been applied and verified on large groundwater systems and, thus, is defendable as a predictive tool for generic or site-specific repository studies. The capacilities of the model were demonstrated by using a test case consisting of the multilayered groundwater system beneath Long Island, New York [9].

Geohydrological systems and surface water bodies (lakes, rivers, etc.) usually have irregular boundaries, making the finite element method (irregular grid) a more powerful tool for their space and boundary definition than the finite difference methods (using uniform square or rectangular grids). The model divides a region into a number of discrete nodes and elements at which all hydrological parameters are defined. Connecting the nodes results in subdividing the entire surface region into two-dimensional elements. Spacing of the model nodes can be varied as required, thereby allowing a closer spacing and smaller elements (i.e. higher resolution in the results) in areas of interest (i.e. repository and river), and a larger spacing in areas where limited data or less complex interactions are present.

Some of the complex generic and site-specific geological configurations consist of multi-aquifer systems, and these aquifers respond conjunctively to stress imposed on any of the various layers. The finite element model can simulate these multilayered systems where not only thickness can vary, but the number of layers can be changed to agree with the vertical geological section. Moreover, the hydraulic conductivity and the pumping stresses can change from layer to layer

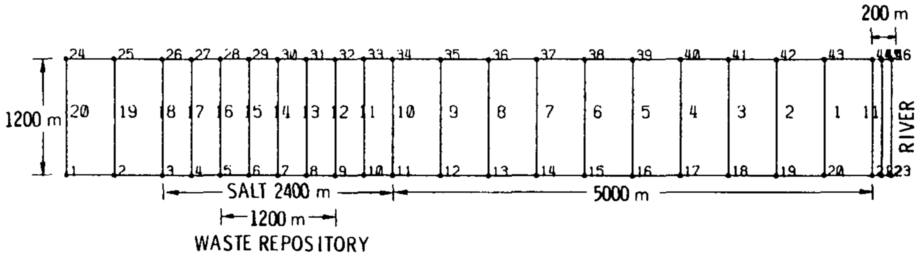

FIG.4. Plan view of the model region.

and/or from element to element, thereby allowing an accurate representation of confining layers and inter-aquifer transfer.

To provide for interaction with the various facets of the problem, supporting programs have been developed to plot grid values, contour maps, streamlines, and three-dimensional graphics. These programs can be used to interpret and check the accuracy of input data as well as output predictions.

5.1.2. Development of the conceptual waste repository hydrological model

The first stage of the modelling effort involved examining the geohydrological data and developing a conceptual model of the system. Much of this information was extracted directly from Ref. [1], but additions and adjustments were made as required to define the system for use in the modelling effort.

As described in Section 2.1, the model area consists of a hypothetical waste repository, a uniform flow region from the repository to the biosphere uptake point, and uniform layering of five distinct geological units. Data supplied on the aquifer systems deal only with the change in properties with depth; this requires modelling the system as an X–Z system, with uniform properties assumed along the Y dimension. Since the three-dimensional model was used, this X–Z system must be modelled with one row of three-dimensional elements in the X direction. As a result, the surface of the region was simulated by a row of 20 rectangular linear elements involving 46 surface nodes as shown in Fig. 4. The nodes are 1200 m long in the Y dimension (approximate length of one repository room), and 500 m wide in the X dimension. In the area of the salt layer and the river, the node spacing was changed to 600 and 100 m, respectively, to represent more accurately the actual size of the repository and a typical river. The total horizontal length of the region is 8600 m, and the distance from the repository to the river is 5000 m. The width over the entire region is 1200 m.

For modelling, the vertical dimension consists of four uniform geological units extending the length of the model region, plus a fifth layer of salt that exists only in the area of the hypothetical repository. Table V outlines a geological

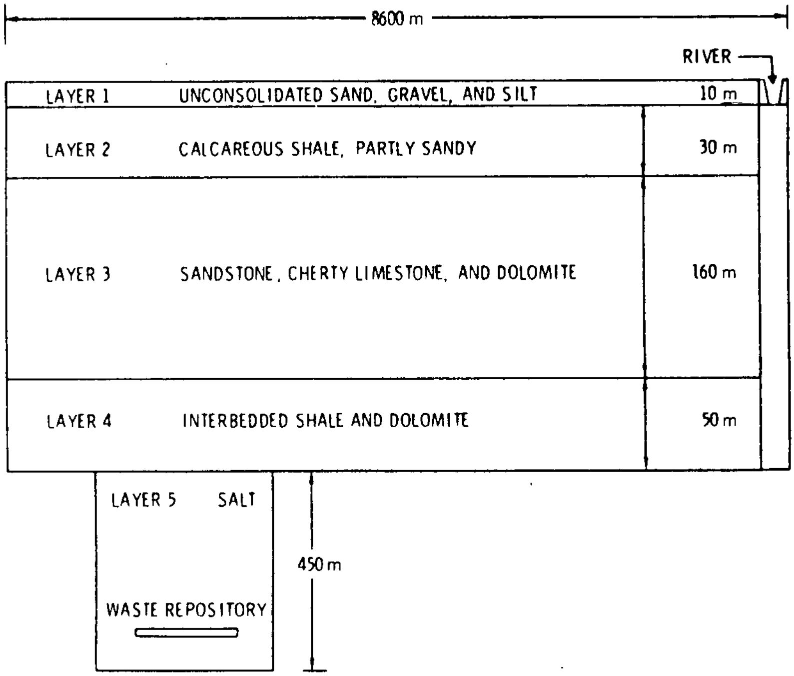

FIG.5. Cross-section of the five geological layers used to define the model region.

description of the layers and their applicable hydrogeological parameters. In the model, vertical nodes were placed at the intersection of the layers (Fig. 5). Additional nodes were evenly spaced in the thicker layers (i.e. layers 3, 4, and 5 in Fig. 5) to increase resolution and accuracy in the model results. The combined thickness of all vertical layers is 700 m.

The hypothetical model repository was assumed to be a 1200 m square. The repository was placed at 600 m below the surface and centred in the area of the fractured salt formation modelled.

The regional water table was assumed to have a uniform gradient of 1 m in 1 km, and the river elevation to be 190 m above sea level. The repository is up-gradient from the river, and the water-table elevation at the repository is about 195 m above sea level. To keep this gradient, a flux was calculated for each of the top four layers and was input into the model at the most upstream nodes. Figure 6 shows the calculations and the amount of flux.

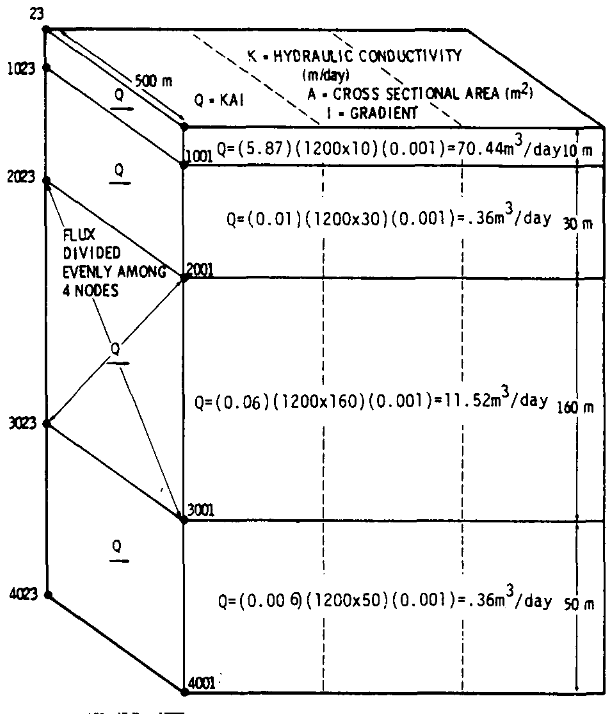

FIG.6. Model flux calculations.

The river was assumed to be the regional discharge site for all the water-bearing layers. To simulate this in the model, the river was made to intersect the upper layer at nodes 23, 1023, 46, and 1046, and all nodes below this are considered no-flow boundaries (Fig. 7). This would force the water in the lower layers to move up to the river.

5.1.3. Hydrological model results

The output from the hydrological model is the groundwater potential (elevations) distribution throughout the X–Z plane over the X–Z region modelled. A contour plot of these potentials with superimposed water flow paths is shown in the X–Z cross-section plot of Fig. 8. As a result of fracturing the salt formation and the overlying shale in the area of the repository, some of the water is able to flow down through the repository and eventually make its way up to the river.

An auxiliary program for the finite element model calculates the travel time, travel distance, and velocity of the water along any streamline within the region.

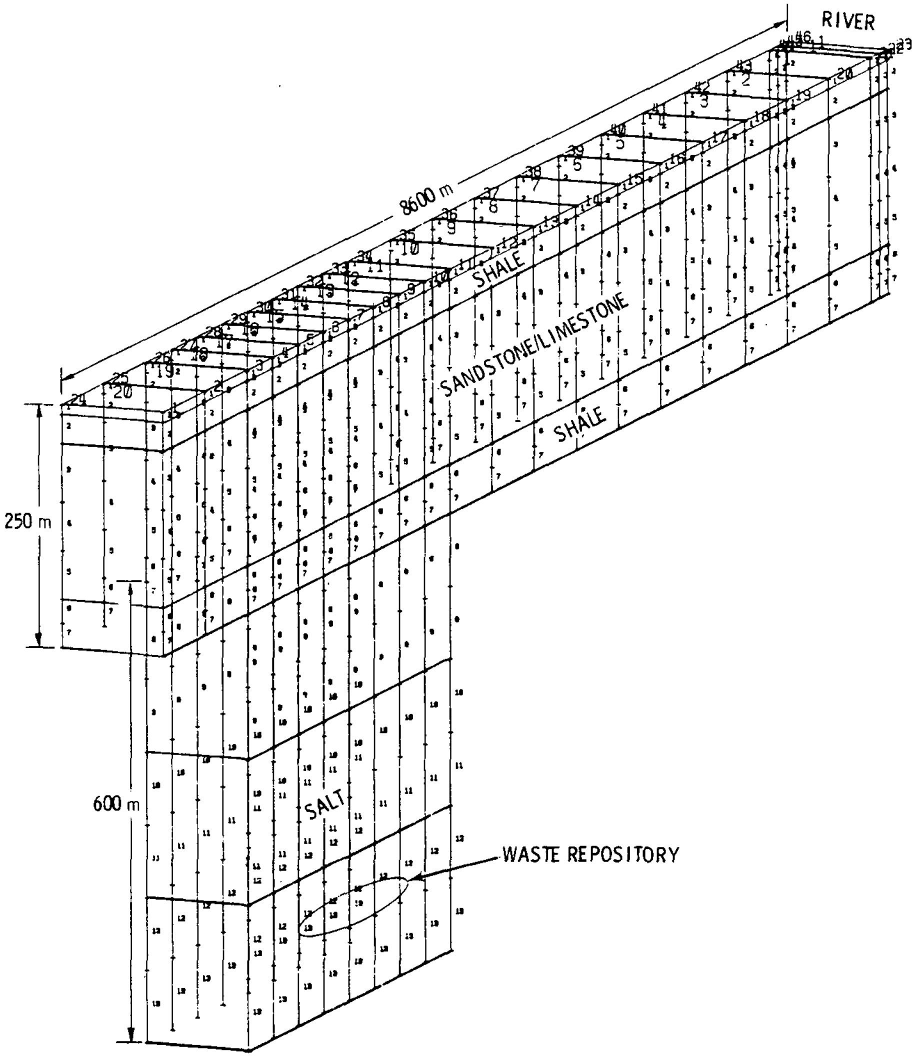

FIG. 7. Three-dimensional view of the model region.

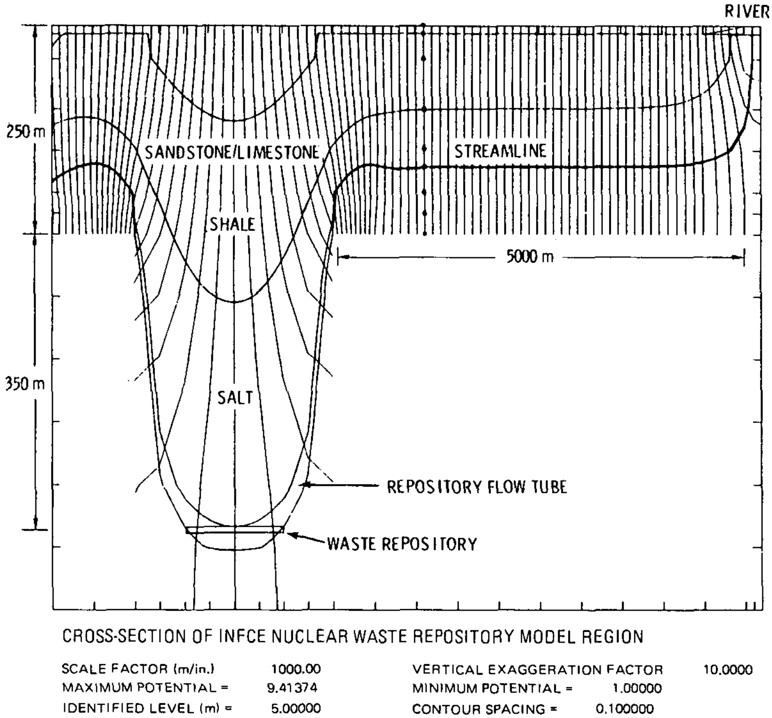

FIG.8. Contour plot of the model-predicted vertical groundwater potentials with superimposed streamlines.

These values are calculated according to the hydraulic conductivities, porosities, and gradients of the various layers along the flow path. The input required by the transport model is an average for each value in the flow tube encompassing the 600 m on the down-gradient side of the repository (see Fig. 8). The averages were calculated using a weighted average according to flow for multiple flow tubes and streamlines spaced along the repository. The average values used in the transport model are:

Average distance – 6100 m

Average travel time – 100 000 years

Average velocity – 0.06 m/a

Average width of the flow tube – 60 m

Average flow through the flow tube – 190 m^3/a

5.2. Transport models

5.2.1. One-dimensional multicomponent mass transport model

The transport model used for analysing the scenario described above was the one-dimensional multicomponent mass transport (MMT) model which has been modified to handle simple decay and linear chain decay as well as branch chain decay. The transport equation, which considers the retardation resulting from rock-nuclide-water interactions, has the form shown in Eq. (6):

$$K_1 \frac{\partial N_1}{\partial t} = D \frac{\partial^2 N_1}{\partial x^2} - V \frac{\partial N_1}{\partial x} - \lambda_1 K_1 N_1 \tag{6}$$

where

K_1 = retardation coefficient for nuclide 1
N_1 = activity concentration of nuclide 1
λ_1 = decay rate of nuclide 1
V = groundwater velocity
D = dispersion coefficient
x = distance in x-direction
t = time

The initial and boundary conditions used include:
$N_1(x,0) = 0$ (initial).
$N_1(L,t)$ = finite (boundary).
$N_1(0,t) = f$ (leach model and decay).

When accounting for decay chains a system of partial differential equations results, which are written as shown in Eqs (7) and (8):

$$K_2 \frac{\partial N_2}{dt} = D \frac{\partial^2 N_2}{\partial x^2} - V \frac{\partial N_2}{\partial x} - \lambda_2 K_2 N_2 + \lambda_2 K_1 N_1 \tag{7}$$

$$K_n \frac{\partial N_n}{\partial t} = D \frac{\partial^2 N_n}{\partial x^2} - V \frac{\partial N_n}{\partial x} - \lambda_n K_n N_n + \lambda_n K_{n-1} N_{n-1} \tag{8}$$

The units of $\dot{N}$ are in curies, *not* in gram-atoms. Branched chains are taken care of similarly; only the subscripts for the parent of the nuclide change.

The method of solution is by simulating the physical phenomena causing mass transport in groundwater aquifers. Convection is simulated by attaching mass to a parcel and moving that parcel a distance calculated by the product of the local groundwater velocity and the current time step. Dispersion is simulated

TABLE VI. TRANSPORT MODEL INPUT PARAMETERS

Nuclides	K_d (mL/g)	β (g/mL)
^{99}Tc	1.9	11.5
^{129}I	0.5	11.5
^{135}Cs	1.0	11.5
^{226}Ra	0.7	11.5
^{231}Pa	100	11.5
^{all}U	2.9	11.5
^{237}Np	23	11.5
^{all}Pu	80	11.5
^{all}Am	1130	11.5

by using random numbers between −1 and 1, multiplied by a length factor characteristic of the dispersion constant and the time step [3].

The model used for these simulations, as stated above, was a one-dimensional version of MMT. The data requirements for that model include:
Retardation coefficient; dispersion; half-lives for all nuclides; path length; groundwater velocity; flow tube size; initial inventory; time after repository closure when the breach occurs; leach information to control entry of waste into groundwater system; and a mapping illustrating the parent-daughter relationship.

The retardation coefficient is calculated from the rock-to-solution ratio, β(g/mL), and the distribution coefficient, K_d(mL/g), by Eq. (9):

$$1 + K_d = K \tag{9}$$

where β, the rock-to-solution ratio, is a function of the porosity and bulk density of the media. The path length and groundwater velocity were obtained from the hydrological model.

The results obtained from these simulations were groundwater concentrations of nuclides. These results were reported as a plot of maximum concentration (μCi/mL) in the groundwater versus time for each nuclide. From the radionuclide inventory in the repository, only the isotopes which would have any measurable inventory after 10 000 years were selected for study since the travel time to the river was 10^5 years for water alone. Table VI shows the distribution coefficients, K_d, and the rock-to-solution ratios, β, used as input to the transport code for each of the nuclides selected.

Isotopes not involved in chain-decay schemes can be followed one by one through the flow system. These isotopes include the fission products and the activation products. The actinides, however, are involved in essentially four decay chains, some of which are linear and some of which include branches. The total chain lengths are much longer than the list shown in Table VI, but the short half-lives of the intermediate members allow them to be ignored for the transport problem.

5.2.2. Transport model consequences

To represent properly the transport of the radionuclides, the model was run until all isotopes either reached the river or fully decayed. The results were reported by the computer as plots of concentration versus arrival time. The results for the example selected for presentation in this document are summarized in Table VII, which reports the maximum concentration in the groundwater and in the river, as well as the ratio of these maximum concentrations to the uncontrolled water radiation standard (MPC). An asterisk (*) after the ratio value indicates that MPC was exceeded.

Only three of the fifteen fission products (^{99}Tc, ^{129}I, and ^{135}Cs) arrive at the river. The other fission products decay along the flow path. Of the three fission products reaching the river, only the ^{129}I has a groundwater concentration greater than its MPC, and ^{129}I is the first isotope to reach the river. For this particular scenario, the ^{129}I peak groundwater concentration arrives at the river about 600 000 years after the release from the repository and is 40 times greater than MPC. The two remaining fission products take well over one million years to arrive at the river, and their peak concentrations are below MPC.

Of the 16 alpha-emitting actinides modelled, only the ^{226}Ra arrives at the river with a groundwater concentration greater than MPC. For this particular scenario, the ^{226}Ra peak groundwater concentration arrives at the river about three million years after release from the repository, and the peak concentrations are 1000 times greater than MPC. The plutonium and the americium decay before reaching the river and all remaining actinides arrive at the river in concentrations far below MPC. Peak arrival times for the actinides range from 3.4 million years to 74 million years.

All the above concentrations should be put in proper perspective by noting that brine-saturated groundwater is about 1000 times above the recommended drinking-water standard for salt content. In addition, when the groundwater

TABLE VII. SUMMARY OF INFCE SALT REPOSITORY TRANSPORT MODEL RESULTS

ISOTOPE	TOTAL CURIES RELEASED TO BIOSPHERE	RELEASE PERIOD (YEARS) MINIMUM TIME	RELEASE PERIOD (YEARS) MAXIMUM TIME	TIME OF PEAK RELEASE (YEARS)	MAXIMUM RATE OF RELEASE TO BIOSPHERE (Ci/a)	MAXIMUM CONCEN-TRATION IN GROUND WATER (μCi/mL)	RATIO OF GROUND WATER CONCEN-TRATION TO MPC**	MAXIMUM CONCEN-TRATION IN THE BIOSPHERE RIVER (μCi/mL)	RATIO OF BIOSPHERE RIVER CONCEN-TRATION TO MPC**
99-TC	3.38E+01	2.04E+06	2.56E+06	2.23E+06	1.74E-04	1.96E-07	6.55E-04	1.10E-11	3.68E-08
129-I	1.26E+02	6.01E+05	7.54E+05	6.68E+05	2.22E-03	2.51E-06	4.18E+01*	1.41E-10	2.35E-03
135-CS	8.28E+02	1.11E+06	1.41E+06	1.23E+06	7.74E-03	8.73E-06	8.73E-02	4.91E-10	4.91E-06
240-PU	--------	--------	--------	--------	--------	--------	--------	--------	--------
236-U	5.10E+02	3.01E+06	4.23E+06	3.37E+06	1.69E-03	1.90E-06	6.33E-02	1.07E-10	3.57E-06
232-TH	9.00E-02	3.20E+06	1.02E+08	7.36E+07	1.06E-09	1.19E-12	5.95E-07	6.72E-17	3.36E-11
241-AM	--------	--------	--------	--------	--------	--------	--------	--------	--------
237-NP	4.11E-01	2.35E+07	2.91E+07	2.60E+07	1.70E-07	1.92E-10	6.40E-05	1.08E-14	3.59E-09
233-U	7.38E+00	3.02E+06	2.91E+07	2.46E+07	1.31E-06	1.48E-09	4.93E-05	8.31E-14	2.77E-09
229-TH	1.89E-01	3.03E+06	3.00E+07	2.55E+07	2.81E-08	3.17E-11	4.53E-06	1.78E-15	2.55E-10
242-PU	--------	--------	--------	--------	--------	--------	--------	--------	--------
238-U	4.00E+03	2.97E+06	3.47E+07	3.38E+06	1.34E-02	1.51E-05	3.78E-01	8.50E-10	2.12E-05
234-U	4.07E+03	2.91E+06	3.47E+07	3.38E+06	1.29E-02	1.46E-05	4.87E-01	8.18E-10	2.73E-05
230-TH	1.80E+02	2.93E+06	3.56E+07	3.51E+06	4.43E-04	5.00E-07	2.50E-01	2.81E-11	1.40E-05
226-RA	9.08E+03	2.57E+06	3.62E+07	3.40E+06	2.41E-02	2.72E-05	9.07E+02*	1.53E-09	5.09E-02
243-AM	--------	--------	--------	--------	--------	--------	--------	--------	--------
239-PU	--------	--------	--------	--------	--------	--------	--------	--------	--------
235-U	5.25E+01	3.04E+06	5.77E+06	3.38E+06	1.80E-04	2.03E-07	6.77E-03	1.14E-11	3.81E-07
231-PA	1.76E+00	3.05E+06	7.68E+06	3.40E+06	6.14E-06	6.92E-09	7.69E-03	3.89E-13	4.32E-07

* Indicates that the isotope is above MPC.

** MPC maximum permissible water concentration uncontrolled water radiation standard (μCi/mL).

---------- Indicates that the isotope decayed before reaching biosphere.

TABLE VIII. INFCE SALT REPOSITORY – SUMMARY OF DOSE RECEIVED BY MOST EXPOSED INDIVIDUAL OVER 50 YEARS FOLLOWING 50 YEARS OF BUILDUP IN THE ENVIRONMENT

NUCLIDE	PEAK TIME YEARS	MAX. RATE (Ci/a)	SKIN	BONE	GI-LLI (mrem)	THYROID	BODY
99-TC	2.23E+06	1.74E-04	0,0E-01	6,1E-06	3,0E-04	0,0E-01	2,5E-06
129-I	6,68E+05	2,22E-03	6,9E-04	6,4E-04	4,3E-04	7,4E-01	1,0E-03
135-CS	1.23E+06	7,74E-03	3,9E-08	1,8E-02	4,0E-04	0,0E-01	7,0E-03
240-PU	------	------	------	------	------	------	------
236-U	3,37E+06	1.69E-03	1,2E-05	1,8E-02	1,3E-03	1,4E-08	1,1E-03
232-TH	7,36E+07	1,06E-09	0,0E-01	4,2E-08	0,0E-01	0,0E-01	0,0E-01
241-AM	------	------	------	------	------	------	------
237-NP	2,60E+07	1.70E-07	1,2E-07	4.1E-06	5,4E-07	9,6E-08	2,7E-07
233-U	2,46E+07	1,31E-06	1,5E-06	1,6E-05	2,4E-06	1,3E-06	2,2E-06
229-TH	2,55E+07	2,81E-08	3,0E-08	8,4E-06	7.0E-07	2,5E-08	2,7E-07
242-PU	------	------	------	------	------	------	------
238-U	3,38E+06	1.34E-02	5.7E-06	1.4E-01	3,0E-02	1,7E-07	8,3E-03
234-U	3,38E+06	1,29E-02	1,0E-04	1,5E-01	1,1E-02	3,7E-06	9,5E-03
230-TH	3,51E+06	4,43E-04	1,4E-03	2,1E-02	2,3E-03	1,2E-03	1,7E-03
226-RA	3.40E+06	2.41E-02	6,8E-02	1,8E+02	3,3E-01	6,1E-02	9,7E+01
243-AM	------	------	------	------	------	------	------
239-PU	------	------	------	------	------	------	------
235-U	3,38E+06	1,80E-04	3,6E-04	0,0E-01	4,8E-04	2,9E-04	4,1E-04
231-PA	3.40E+06	6,14E-06	6,6E-06	2,3E-04	7.0E-05	5,6E-06	1,4E-05

River flow = 500.0 cm Groundwater flow = 886.6 m^3/a

----------- Indicates that the isotope did not reach the biosphere.

reaches the river, it will be diluted by a factor of 10^5 to 10^7, making it well below MPC, as indicated in the column farthest to the right in Table VII.

6. BIOSPHERE AND DOSIMETRY ANALYSES

6.1. Biosphere models

The dose models used in this study were derivatives of ARRRG [4] and FOOD [5], codes developed to estimate annual radiation doses and long-term dose commitments to the total body and selected organs of individuals and to population groups, from both internal and external sources of radiation. Although these computer codes were developed specifically for evaluating the potential radiological impact of commercial power reactors, they are usable for any nuclear facility that may release radioactive materials to the environment.

ARRRG calculates annual individual and population doses resulting from radionuclides released with liquid effluents. Various exposure pathways may be selected by the user: (a) consumption of fish, invertebrates, algae, and drinking water; and (b) direct external radiation from shoreline, water immersion (swimming) and surface water (boating). Doses are calculated for skin (external only), total body, GI-LLI, thyroid and bone. Individual contributions to dose by nuclide and pathway are output. At the time of the study ARRRG was able to calculate doses for eight organs and about 200 radionuclides. The user inputs the following variables to ARRRG from data files: (a) name of the facility under investigation; (b) decay between release and point of exposure (hold-up); (c) usages and mixing ratios by pathway; (d) reactor coolant flow; (e) shore-line width factor; and (f) reconcentration factor parameters.

FOOD calculates annual individual doses from the consumption of agricultural foods and animal products contaminated from air deposition or water (sprinkler irrigated). At the time of the study 14 food types could be selected. The input data for FOOD includes: (a) facility name; (b) hold-ups; (c) usages; (d) irrigation rates; (e) air concentration; (f) crop yields and growing periods for 14 food types (for animal products the parameters refer to animal feed); (g) coolant flow and mixing ratio; and (h) reconcentration factor parameters in the case of liquid release.

Because of time constraints for the study, it was not possible to use the complete versions of ARRRG and FOOD; however, shortened versions of ARRRG and FOOD [6] were used to obtain doses from a river with an average flow rate of 500 m^3/s. Dose was calculated by the following:

$$D_{ij} = df_{ij} \, N_i \left(\frac{283.2}{R}\right) \qquad (10)$$

where

D_{ij} = dose im millirems (mrem) to an organ, j, of the most exposed individual[1] resulting from 50 years' accumulation followed by 50 years' exposure for radionuclide species, i;
df_{ij} = dose factor for radionuclide species, i, and body organ, j (skin, total body, GI-LLI, bone and thyroid);
N_i = rate of discharge of radionuclide species i to the river (Ci/a);
R = water flow rate in river of interest (500 m^3/s);
283.2 = flow rate of reference river (m^3/s) used to generate the dose factors df_{ij}.

6.2. Dose model consequences

There are several modes for assessing dose: most exposed individual, average individual, local population, regional population, etc. The most exposed individual (worst case) was used for these analyses. To obtain maximum potential doses, the times of peak isotope concentrations were obtained from the transport model and the contribution to dose by all nuclides was calculated at each peak time.

The results presented in Table VIII summarize, by nuclide, the doses (mrem) to five organs of the most exposed individual (skin, body, GI-LLI, bone, and thyroid) for the example selected for presentation in this document. The doses are based upon 50 years' buildup at the peak concentration followed by 50 years of exposure. The major contribution to dose is from ^{226}Ra. The highest dose of 180 mrem resulted from ^{226}Ra in bone.

Doses from drinking contaminated groundwater were not analysed since salt concentration in the groundwater was well above drinking-water standards and would preclude its use for water supply.

As was the case for transport model consequences, the dose consequences are for a repository containing a year's accumulation of waste from a 100 GW(e) economy. To obtain the dose to the most exposed individual for any other size of repository up to the maximum of about four years of waste from a 100 GW(e) economy, the dose numbers in Table VIII must be multiplied by the number of hundreds of giga-watt (electric) years of accumulated waste. For example, for a repository holding 2.5 years of accumulated waste from a 100 GW(e) economy, the dose numbers in Table VIII must be multiplied by 2.5; for a

[1] The most exposed individual is a person whose location and habits tend to maximize his radiation dose, resulting in a dose higher than that received by other individuals in the general population.

repository holding 20 years of waste from a 4 GW(e) economy, the dose numbers must be multiplied by 0.8.

To assess the significance of the estimated potential doses, they were compared to the 5000 mrem the average individual might receive from natural background radiation during the same 50-year period.

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Appendix B

NETHERLANDS DOMED SALT REPOSITORY

J. Hamstra

1. INTRODUCTION

At the beginning of 1976 the Netherlands Government decided to order an investigation into all phases of treatment of radioactive waste. The Radioactive Wastes Sub-Commission of the Interministerial Nuclear Energy Commission (ICK) was charged with this investigation. It was stated, in particular, that *"the investigation into the possibilities of ultimate disposal of radioactive wastes is to be pursued with vigour, for which purpose an interdepartmental working group, together with experts from the State Geological Service and the Netherlands Energy Research Centre (ECN), is to carry out a study, including test drillings, into the feasibility and the acceptability of the disposal of radioactive wastes into rock salt formations".*

A number of working groups were set up by the ICK Sub-Commission, one of which specifically covered the safety assessment work. A generic safety assessment of underground disposal of high-level wastes (HLW) in a model salt dome was performed in the form of

A geohydrological model study to establish the isolation properties of a salt dome used for waste disposal purposes; and
An analysis of the radiation doses following a postulated future radionuclide release from a salt dome repository along different pathway models.

The report of the working group on safety assessment was published as appendixes to the report of the Interministerial Nuclear Energy Commission on the feasibility of radioactive waste disposal in salt formations in the Netherlands. These appendixes consisted of:

(a) The geological data for drawing up a geohydrological model as established by the State Geological Service;
(b) The geohydrological model calculations as made by the Geohydrological Division of the National Institute for Water Supply (RID); and
(c) The radionuclide release scenarios with their subsequent radiation dose calculations as made by the Institute for Application of Nuclear Sciences in Agriculture (ITAL).

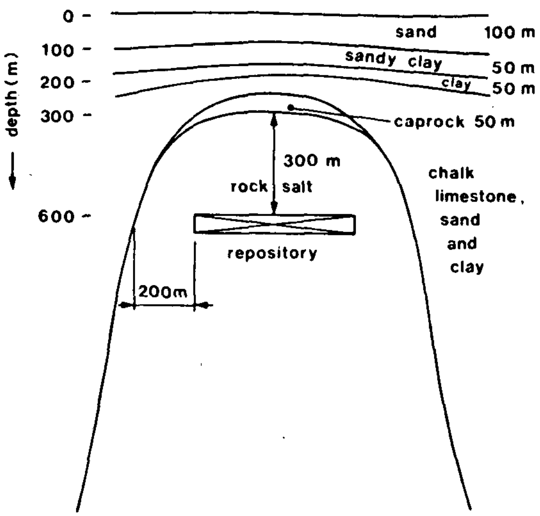


FIG.9. Section through the model salt dome and the adjacent strata.

An English translation of the report and its appendixes was made available by the Ministry of Economic Affairs in April 1979 [1]. Information from this generic safety assessment work relevant to the present document is summarized in the following paragraphs.

2. DESCRIPTION OF HOST ROCK SITE AND REPOSITORY

For the generic safety assessment study a model salt dome, situated in the northeast Netherlands, was selected; its top rock salt is at 300 m depth. Figure 9 shows the cross-section over the reference salt structure and the overlying strata. The repository tunnels were assumed to be situated at 600 m depth and to have an isolation shield of 300 m thickness towards the top and 200 m thickness towards the flanks of the salt dome.

One of the main aims of the Netherlands conceptual design studies for a HLW repository was to limit the thermal loading of the host rock by making an optimal use of the vertical dimension of the domed salt. The conceptual design studies evolved from the disposal of a total of 50 000 canisters of HLW in 50 m deep bore-holes at three consecutive disposal levels at 900, 750 and 600 m depth [2] to the disposal of the same number of HLW canisters in 300 m deep bore-holes

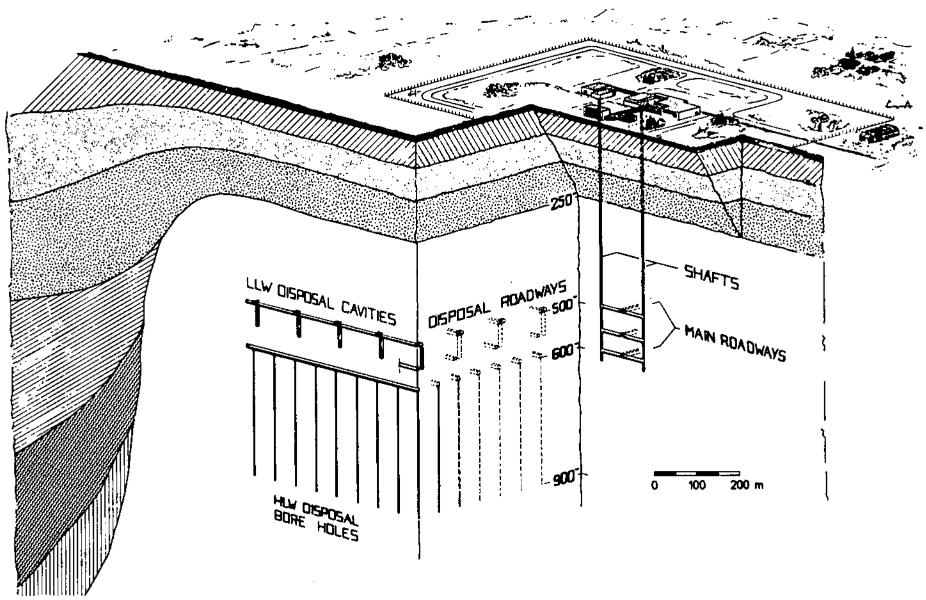


FIG.10. Artist's view of a nuclear waste repository in a salt dome.

from 600 m down to a depth of 900 m [3] as illustrated in Fig. 10. Each HLW canister was assumed to contain 50 litres immobilized high-level waste derived from reprocessing 0.6 t LWR fuel.

The distribution of the HLW canisters over the available rock-salt volume was an important input datum for the safety analysis. The amount of rock salt surrounding each 100 litres HLW was increased from 5000 m^3 for the three-disposal-level concept to 10 000 m^3 for the 300-m-deep bore-hole concept.

For the Netherlands generic safety assessment study, the emplacement of a total of 50 000 canisters of high-level reprocessing waste was conservatively assumed to be realized between a depth of 600 and 680 m. The distribution of the HLW canisters was thus limited and resulted in a rock-salt volume of only 1500 m^3 surrounding each 100 litres HLW.

The radionuclide inventory of the repository was calculated by the ORIGEN code and represented a total nuclear waste production of 1000 GW(e)/a of nuclear power production.

The HLW was assumed to be disposed of 10 years after discharge of the spent fuel from the reactor. The efficiency during reprocessing was assumed at a Pu and U recovery of 99.5%, whereas 100% of the ^{129}I was assumed to remain in the waste.

3. SCENARIO SELECTION

The geohydrological model used for determining, in a generic way, the isolation value of a salt dome used for nuclear waste disposal purposes was established in close co-operation between the State Geological Service and RID. The two main parameters that dominated these model calculations proved to be the groundwater velocity at greater depth and the upward movement of the salt dome.

The scenario used for establishing an average groundwater velocity over and around the top of the salt dome was based on existing geological, geophysical and geohydrological information. The circulation of groundwater in the NE Netherlands takes place for the greater part in the coarse sands of the Upper Tertiary and of the Quaternary. These strata are to be considered as the upper aquifer, which is not a homogeneous medium but is built up mainly of coarse sands, alternating with low permeability clay and loam layers. The bottom part of the aquifer consists of an alternation of fine sand and clay. Consequently, the hydraulic conductivity is low. The base of the aquifer is formed by the Oligocene clay (Rupelian). In some places the thickness of the aquifer is strongly diminished by the presence of glacial erosion valleys, filled up with silty material or clay. These valleys are found to reach depths of more than 300 m. Below the Oligocene clay that forms the base of the upper aquifer, there are water-containing deposits of Lower Tertiary (Eocene and Paleocene) and Mesozoic age. The last-mentioned formations are regarded as the lower aquifer.

The groundwater flow in the northeastern part of the Netherlands in the upper aquifer is well known. The groundwater potential is regularly registered in numerous observation wells. The regional groundwater flow is directed towards the north-northwest. Rain-water reaching the groundwater in the elevated infiltration area near the Federal German border flows underground towards lower areas, where it is drained by open water courses. A schematic section through the geohydrological model of northeastern Netherlands is shown in Fig. 11.

A finite-element model was developed by RID to calculate the rate of groundwater flow in the lower aquifer. The model described the flow in a system of two aquifers, separated by a semi-pervious layer. Aided by this model a flow rate in the lower aquifer of about 0.1 m/a was calculated. In the surroundings of a salt dome the flow lines will bend around the diapir. Owing to these curved flow lines the flow rate might become 0.2 m/a. For the hydraulic conductivity of the semi-pervious clay layer, a value of 2×10^{-9} m/s was used in the model.

The process of salt dome uplift (diapirism) could be very important with respect to the possible dissolution of the isolation shield of rock salt surrounding the repository. Substantial uplift could lead to contact with the upper aquifer where, in comparison with the lower aquifer, relatively high flow velocities occur. In an extreme situation, diapirism would lead to the rise of the dome above ground level.

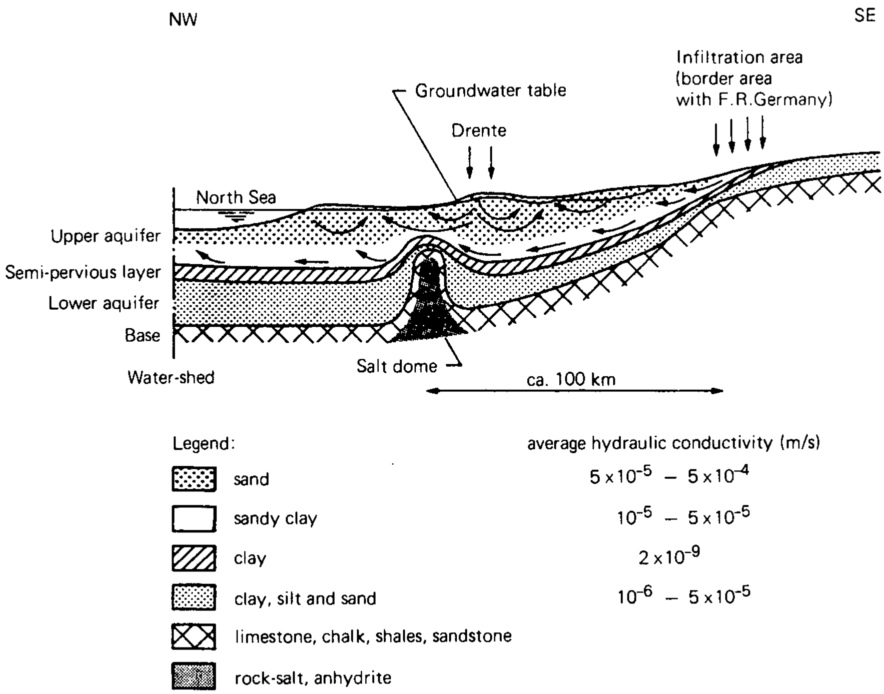


FIG.11. Schematic section through the geohydrological model of northeastern Netherlands.

The average uplift of the salt domes in Groningen and Drenthe from the beginning of the Tertiary period has not been more than 0.05 mm annually.

Since the diapiric movement can be assumed to have a pulsating character, a rate of 0.25 mm annually could be reached. For diapirism elsewhere in the world, figures up to about 2.5 mm annually can be found in the literature. Therefore, the values for upward movement of the salt dome used in the model calculations varied from 0.25 to 2.5 mm annually.

For a future radionuclide release from a salt-dome repository, several scenarios were considered feasible, depending upon the rate of continuous upward movement of the salt dome, the groundwater velocity and the long-term effects of climatological changes.

The most extreme scenario would be a high rate of diapirism of 2.5 mm annually. Supposing that the uplifted covering layers were continuously eroded away, the top of the salt diapir would, once it surfaces, also be eroded and the radioactive waste would ultimately come to the surface and come into contact with water or air. With an annual uplift rate of 2.5 mm approximately 250 000 years would pass before the disposed waste would reach the surface.

The climate at that moment would be very important. It is feasible that climatic changes could lead to a situation whereby the area may be assumed to be covered by sea. If the land were to remain dry, dispersion of the eroded glass, in which the radionuclides were fixed, by wind would be possible in a desert climate. Thus, the analysis of the radiation doses following a future radionuclide release from a salt-dome repository was made, assuming the release to start 250 000 years after disposal, and taking into account the different climatic situations as they may develop in such a time period.

4. REPOSITORY ANALYSIS

For this generic safety assessment no values were attributed to either the repository design or to eventual engineered barriers. Important assumptions for the scenarios were:

The thickness of the isolation shield of rock salt above and around the repository used in the dissolution calculations will set the time period prior to any release of radionuclides; and

The distribution of the HLW canisters over the rock salt volume available for disposal, which in relation to the dissolution rate determines the radionuclide release rate, once the isolation may be assumed to have ended.

The figures used for both of these input data are given in Section 2.

5. GEOSPHERE ANALYSIS

Because of the high rate of the upward movement of the salt dome (2.5 mm annually) assumed for the geohydrological model, the geosphere transport is associated with the rising salt itself and not with dissolved radionuclides moving with water through the rock and/or the sediments.

Because the layers covering the salt dome consist of unconsolidated sand and clay, the arising of a mountain owing to the continuous upward movement of the salt dome was not considered credible, although a hill of some tens of metres might occur. It was, however, assumed that both these uplifted sediments and the surfacing cap-rock would be eroded away continuously. Finally, the surfacing rock salt surrounding the waste canisters was assumed to be dissolved, leading to the waste surfacing almost intact. It is consistent with such a scenario that no value was attributed to the geochemical retardation effects during the transport of radionuclides through the geosphere. Adsorption processes were only considered on soil in the biosphere analysis.

TABLE IX. CALCULATED DISSOLUTION RATE OF A SALT DOME

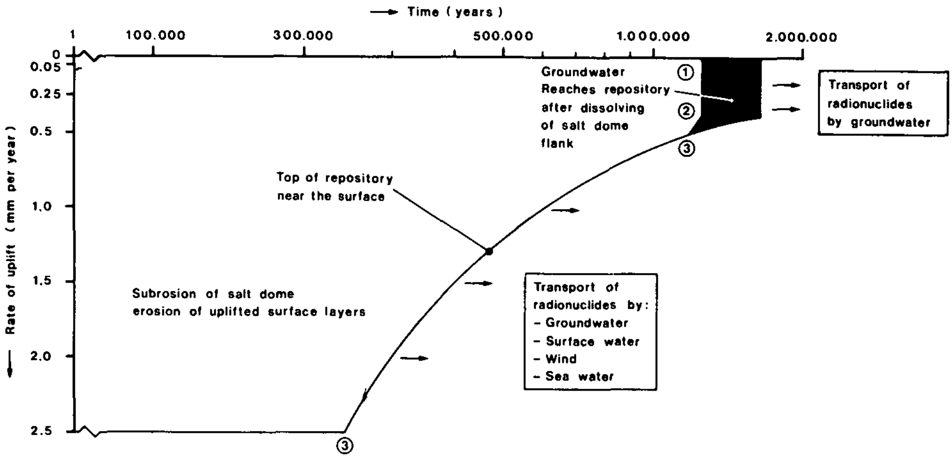
	Situation	Flow velocity (m/a)	Diffusion factor (m^2/a)	Boundary layer (m)	Dispersion factor (m^2/a)	Rate of dissolution (mm/a)
Top of the dome	Slow-flowing groundwater	0.5	0.025	50	0.05–0.5	0.008
Flanks of the dome	Slow-flowing groundwater	0.2	0.025	2	0.025–0.2	0.12–0.16
	Moderate/ fast-flowing	2	0.025	2	0.2–2	0.18–0.19
	groundwater	20	0.025	2	2–20	0.19

Prior to any future release of radionuclides from the disposed nuclear wastes is the continuous process of dissolution at the top and the flanks of the salt dome by moving groundwater. Although the radiation dose calculations were based on a release of radionuclides after 250 000 years, as a consequence of an extreme continuous diapiric movement of 2.5 mm annually, the dissolution calculations were made on a more realistic base.

Because known figures from the beginning of the Tertiary indicate that the average annual uplift of the salt domes was less than 0.05 mm, RID constructed a computer model to calculate the subrosion rate of the model salt dome at low diapiric movement.

The dissolution process of salt in water is governed by the diffusion and dispersion equations. At the top the salt dome is covered by a cap-rock of 50 m. It is presumed that fissures in the cap-rock are present. In these fissures diffusion of salt would take place. At the flanks of the salt dome there is no, or almost no, cap-rock but, if any salt dissolves, some insoluble substance would remain. In the model it was presumed that diffusion would occur through fissures in the boundary layer and dispersion in flowing groundwater in the surrounding permeable strata.

The thickness of the boundary layer above the salt dome was set equal to the cap-rock thickness (50 m). At the flanks of the salt dome a boundary layer of 2 m was provisionally chosen. This 2 m boundary layer might be the result of the dissolution of rock salt and precipitation of insoluble substances. The model (SAL) calculates in a rectangular grid the transport of salt by groundwater under steady flow conditions.



Explanation : ① Dissolution of salt dome flank by slowly flowing groundwater

② " " " " " " (moderately) fast flowing groundwater after 450 m upward movement

③ Repository at surface while the flanks are still (partly) intact

Remark: the width of the shaded area is reflecting the variation in hydrological input data

FIG.12. Time required for the release of stored radioactive substances as a function of the dome uplift rate.

During the subrosion process the diapiric movement continues. For this reason contact with more permeable layers was assumed from the moment the salt dome breaks through the covering clay layer. Results are presented for the subrosion rate of the model salt dome in Table IX. The result of the calculation is that after about 1.5 million years groundwater might have dissolved 200 m rock salt and thus might reach the disposal area. The different combinations of the rate of diapirism and subrosion are presented in Fig. 12.

Although it is more realistic to assume that groundwater would dissolve the salt dome, thus exposing the waste after more than a million years, the worst case was assumed to be that all waste would be transported by the rising salt to the surface. In this case geosphere transport would not be with water through rock and/or sediment, but within the rising salt. The waste might reach the surface almost intact and then the release of radionuclides from the waste would start after about 250 000 years directly within the biosphere. Therefore, retardation of radionuclides by adsorption processes during geosphere transport would not occur in this evaluation.

6. BIOSPHERE ANALYSIS

For the scenarios described above, the following models were used to calculate the consequences of postulated releases in the form of radiation doses to a future population living near the repository area:

(a) A drinking water model, in which the saline groundwater contaminated with radionuclides was assumed to be diluted to maximum chlorine content of 600 mg/L, as recommended by the World Health Organization;
(b) An inhalation model, in which the surfaced glass particles were assumed to be eroded to re-suspendable dust particles;
(c) Two agricultural models, in which the radionuclides from the glass particles in the soil are taken up by plants and reach humans either because they are eaten directly, or because they are eaten by cattle and pass through the milk and meat chain; here sorption processes of radionuclides on soil are taken into account;
(d) A radiation model, in which the external radiation exposure comes from surface glass particles;
(e) A fishery model, in which the surfacing of the glass particles was assumed to develop after the area became sea bottom owing to an excessive relative rise in seawater level.

The radiation doses calculated for the different models were assessed by comparing the results with the maximum permissible doses for individuals of a population according to the ICRP Publications Nos 2 and 6. A review of the main assumptions made for the different radiation dose models and the results are given in Table X.

7. OTHER ANALYSES

Sensitivity analyses were made with respect to the influence of some input data in the model calculations. For instance, the influence of both the thickness of the boundary layer at the flank of the salt dome and the influence of the size of the contact area between water and salt on the dissolution rate were established.

Comparisons with radiation burdens from natural radiation were made to put the radiation burdens calculated for the different uptake models into perspective. For instance, the comparison was made between the amounts of ^{238}U and ^{226}Ra in a soil contaminated with radioactive waste glass and other natural rocks. A comparison was also made between the dose rates above a soil contaminated with vitrified waste particles and those above soils with known elevated natural backgrounds.

TABLE X. SUMMARY OF THE RESULTS OF THE NETHERLANDS DOSE MODELS

Critical pathway	Vitrified waste in soil or rock (fraction)	Contaminated area (km^2)	Inhabitants concerned	Estimated maximum radiation dose divided by ICRP limit
Drinking water	7×10^{-4}	0.5	10^3	< 0.5
Inhalation:	7×10^{-5}	5	10^1	< 0.1
External				< 1
Agriculture 1:	5×10^{-5}	100		
Potatoes			10^5	< 3
Milk			10^4	< 1
Meat			10^3	< 1
External			$10^3 - 10^5$	< 1
Agriculture 2:	7×10^{-4}	0.5		
Milk			10^2	< 1
Meat			10^1	< 1
External			$10^1 - 10^2$	< 10
Marine		25	no estimation	< 0.1

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Appendix C

INFCE HARD ROCK (GRANITE) REPOSITORY

C. Cole

1. INTRODUCTION

The INFCE studies evaluated the relative safety impacts of seven different fuel cycles, including disposal of spent fuel in some cycles. For simplicity of presentation, and because it is sufficient for the purposes of this document, the example described here is for only one of the fuel cycles, a LWR fuel cycle with plutonium recycle.

An integrated modelling system was used in this study to examine the potential consequences of a postulated release of nuclides from a hypothetical geological repository located in a hard rock (granite) formation. For the purposes of the INFCE studies, all radioactive wastes from the fuel cycle (except mining and milling) were placed in the geological repository.

The reference repository for this study is for granitic rock or gneiss as the host rock. These rocks are in many respects representative of a broad class of hard crystalline silicate rocks. The descriptions of waste packages and repository facilities used in this study represent only one of many possible designs based on the multiple barrier concept. Actual designs could contain more barriers or fewer, depending on the actual hydrological conditions of the site and the particulars of the nuclear economy and local regulatory requirements.

The objective of the modelling efforts presented in this study was to predict the rate of transport of radioactive contaminants from the repository through the geosphere to the biosphere and thus determine an estimate of the potential dose to humans so that the release consequence impacts could be evaluated [1]. Currently available hydrological, leach, transport and dose models were used in this study. The hydrological model defines water-flow tubes and travel times from input describing the hydrological system and the disruptive event to be analysed. The transport model uses the output from the hydrological model and radioactive release source terms from the leach model to describe the movement of the contaminants through the geosphere. The transport model thus provides release rates and concentrations of radionuclides in the fluids released to the biosphere (in this case, to a surface-water body). This output then serves as an input to the dose model, which provides the estimate of the environmental dose resulting from the radioactive release.

The data on hard crystalline rock indicate that, except for highly improbable violent events such as meteorite strikes and volcanism, little else in the way of more probable geological activity (tectonics, erosion, etc.) is believed to result in disruption of the repository. If appropriate site selection criteria are applied, the probability of deliberate or inadvertent intrusion of a repository, or its immediate environment, by humans should be negligible. Thus, man-caused disruptive scenarios were not considered. As a result, the only release studied represents the normal scenario in which wastes are moved by the small amounts of water (normally present in hard crystalline rocks at depth) out of the repository areas after the waste canisters have failed.

2. DESCRIPTION OF HOST ROCK SITE AND REPOSITORY

2.1. Geological properties

Granitic rocks and gneisses are the most abundant rock types in the upper 10 km of the earth's continental crust. They occur at the surface in stable platform areas, in the cores of many mountain ranges and regional uplifts, and in the subsurface beneath most of the younger sedimentary cover. They are attractive for waste repositories because of their strength, structural and chemical stability, abundance and low economic value.

Uplift, faulting and fracturing are the most significant effects of tectonic activity in the present context, and the rate at which they take place is the most important aspect in their evaluation. The distribution of mobile zones and stable areas is generally well known on the basis of seismic records and observations and structural evidence. Geological observations and isotopic age determinations have further demonstrated that the present deformational pattern has persisted for tens, even hundreds, of millions of years, and that changes in this pattern are extremely slow. This persistence also applies to individual tectonic features, such as major faults, which remain active for many million years. Their long-time activity probably corresponds to an equally long history of growth and propagation. Therefore, the possible role of tectonic activity must be evaluated on both regional and local evidence. Regarding the generic aspects of granitic rocks and gneisses it is found, however, that a subsurface repository in this type of rock is not endangered by seismic shaking.

Gneisses and granite often occur in vast masses. As a rule they constitute rather monotonous and uniform geological units of little intrinsic value. Ore deposits in these rocks are rather rare. A repository in such a rock, therefore, would not normally impede the development of natural resources or present a significant risk of intrusion by future human activities. Dispersal of the waste in a repository could take place if the rock cover were removed by erosion, either acting alone, or in combination with tectonic activity. This again is highly site-specific and must be judged on the basis of geological and geomorphological evidence at the site and the surrounding region. Very often it is found, however, that the areal exposure of granites and gneisses, which originally formed at great depth, in itself indicates that erosion has reached a mature stage in removing the overlying formations. The potential for renewed erosion, particularly in flat and low-lying areas outside recent orogenetic regions and away from major rivers, is therefore very small.

2.2. Geochemical properties

Geochemical dispersal involves the dissolution and transport of the waste in water pathways. The host rock of a repository will act as a geochemical containment if its minerals react with the water and the individual radioactive elements in such a way that they effectively decay during groundwater transport. Granitic rocks, here taken to cover the granite-syenite-tonalite-diorite range, are mainly composed of feldspars, i.e. aluminosilicates of K, Na and Ca. The content of quartz, SiO_2 and dark minerals, representing silicates of Fe, Mg and Ca, may vary. Muscovite occurs occasionally. Gneisses cover about the same compositional range. Fractures and pores in these rocks as a rule contain quartz, calcite and various phillosilicates, such as chlorite, illite and smectite. Although these minerals represent only a small fraction of the rock by volume, they are geochemically important because they line openings where most of the groundwater flow takes place.

The groundwater in granitic rocks and gneisses in areas with a humid climate is normally of the calcium bicarbonate-sodium chloride type. The pH of the groundwater largely reflects the reactions of dissolved carbon dioxide, which is mainly derived from biological activity in the soil. Here it may reach a partial pressure of the order of 10^4 Pa (10^{-1} bar). If the carbon dioxide of the soil-atmosphere equilibrates with calcite and water, the pH will be near 7.7, and the concentration of Ca^{2+} and HCO_3^- in the groundwater will be of the order of 10^{-3} mol/L, with a molar proportion of HCO_3^-/Ca^{2+} near 2. If equilibration takes place below the water table, the pH will be near 10.4 and the contents of calcium and bicarbonate will be about ten times lower. The general correctness of these calculations is verified by a large number of analyses of groundwater from between 200 and 500 m depth. In many cases the HCO_3^-/Ca^{2+} molar proportion is larger than 2, indicating reactions of carbon dioxide with Na minerals as well. In cases of deeply weathered rock, where most calcite has been removed, silicate/carbon dioxide equilibria may lead to a pH of around 8.0 in the groundwater.

The redox potential, Eh, of the groundwater will normally be buffered around -0.2 ± 0.1 V by reactions with iron-bearing phillosilicates. For groundwater in contact with limonite, FeO(OH), as in oxidized rock or in limonite-stained fracture zones, positive Eh values may be expected. The ionic strength, I, often is of the order of 10^{-2} molar. The content of solutes generally increases with depth. The overall conclusion to be drawn from these studies is that the composition of groundwater in granitic rocks and gneisses is governed to a large extent by a set of reactions with the available minerals. Since these are about the same in all granitic host rocks, the chemical characteristics of the groundwater can be expected to show rather limited variations.

At the pH and Eh of the groundwater in granites and gneisses at depth, some of the radionuclides, i.e. the actinides, will be immobilized by reductive precipitation and thus remain fixed in the bedrock. Both theoretical calculations and actual measurements show, for instance, that the maximum concentration of uranium in bicarbonate-bearing groundwater of this type is at most about 3 μg U per litre. This concentration, furthermore, often already prevails in the oxygenated shallow groundwater zones. The deeper, reducing groundwater is therefore naturally saturated in uranium, and unable to dissolve and transport any additional amounts of this element. Other nuclides react with the host rock by ion exchange and adsorption. These processes are often considered together and described as sorption. They lead to a retardation of the nuclide in relation to the flow of the groundwater. Sorption effects may be calculated by means of distribution coefficients – K_d values. Such values pertaining to granite and gneiss and water of groundwater-like composition have been determined in the laboratory for a number of elements. The retardation effect has further been verified for some of them, e.g. Sr, by field experiments, as mentioned in Section 5.2.2 of Appendix D. It should be noted that iodine shows no effects of sorption and hence must be expected to migrate with the velocity of the groundwater.

2.3. Hydrogeological aspects

Groundwater would be the carrier of radioactive nuclides in geochemical dispersal; therefore, hydraulic considerations and the hydraulic properties of the host rock are of great importance in the present context. The groundwater moves in response to differences in hydraulic potential, i.e. the hydraulic gradient. Disregarding artesian conditions, which might occur if the hydraulic conditions are determined by a sedimentary cover on top of the host rock, such differences reflect the slope of the water table above the repository and in its surroundings. These differences in hydraulic potential decrease exponentially with depth. If the hydraulic properties of the host rock were uniform, this would also imply an exponential decrease in the velocity and volume of the groundwater flow.

From considerations of the hydraulic gradient and the general pattern of groundwater flow, it would appear advantageous to locate a repository beneath a groundwater divide. This would ensure an initially downward direction of the groundwater flow and thereby provide long transit times before the groundwater again approaches the surface. The downward flow would also counteract any tendency of the groundwater to move upward, as might otherwise be induced by heat effects from the waste. It should be noted, however, that the position of a groundwater divide, in a geological sense, might not be permanent.

The hydraulic properties of granitic rocks and gneisses are largely determined by their fractures. Therefore, they are site-specific. Nevertheless, each site may show a combination of the following zones:

(a) A zone of *pervasive fracturing* prevails at shallow depth and near larger fracture zones. Here groundwater flow takes place through a three-dimensional network of interconnected fractures which permeates the bulk of the rock. The hydraulic conductivity of such zones statistically decreases with depth. The effective (cinematic) porosity decreases in an orderly way with decreasing conductivity. The groundwater flow in this zone may be described by equations developed for flow in porous media.
(b) This zone gives way in depth and away from larger fracture zones to a realm of *discrete fracture zones* where water-bearing zones are separated by intervening volumes of rock, which will have low porosity and show very low hydraulic conductivity. The water-bearing zones here generally represent systems of more or less interlacing minor fractures throughout a certain width of rock. Underground studies at the Henderson mine (Colorado, USA) have shown that the individual openings along such fractures rarely exceed 3 m in length, although continuous open fractures occasionally may be traced for many tens of metres. A large-scale tracer test in the vicinity of the Savannah River Plant (South Carolina, USA) indicates that flow in such fracture zones may also be described by equations derived for porous media.
(c) The *intervening volumes of rock* with very low hydraulic conductivity are also fractured, but the width of the fractures may be so small and the hydraulic connection between the fractures missing or incomplete, so that the water flow is severely limited or virtually inhibited. Very little is known of the mechanism of groundwater movements in such rocks, but it appears plausible that equations for porous media may provide an upper limit for groundwater flow in these parts of the bedrock.
(d) Finally, at some distance from the repository, *large fracture zones* may occur. These often indicate intense tectonic deformation. Such zones, in the form of eroded valleys, may represent important loci of groundwater discharge. They thereby mark natural boundaries of the wider area surrounding the repository site.

These various zones, in combination, define the hydraulic environment of the waste repository. Most of the groundwater will move through the shallow zone of pervasive fracturing. Some part of it, however, will percolate deeper and reach the realm of discrete fracture zones. Driven by some smaller gradient here, it will move along these zones towards the points of discharge, where it will again meet and be diluted by the more shallow groundwater. A still smaller fraction of the water will reach the waste emplaced in rock of very low hydraulic conductivity (i.e. 10^{-9} cm/s or less). After passing through some length of this rock, it will again reach one of the discrete fracture zones and end up at last at the same points of groundwater discharge as the major part of the groundwater.

2.4. Inventory

For this study the repository size was based on disposal of wastes resulting from a nuclear economy of 100 GW(e)· a. Its underground projected area would be about 90 ha.

The waste packages assumed for this repository are, with one exception, the same as those assumed for the INFCE salt repository. (See Appendix A, Section 2.2). Because of thermal considerations the HLW canisters are only 20 cm i.d.,instead of 30 cm i.d.; thus, there are 6700 instead of 2900. The HLW canisters are emplaced inside compacted bentonite cylinders that line the holes to provide protection for the canisters from the groundwater in the granite. Backfilling of the repository is done with a mixture of sand and bentonite, rather than excavated granite, for the same purpose.

3. SCENARIO SELECTIONS

The data on hard crystalline rock indicate that, except for highly improbable events like meteorite strikes and volcanic eruptions, no geological activity such as tectonics or erosion could be expected to disrupt the repository significantly. Also, if appropriate site-selection criteria and protective measures are applied, as discussed previously, the probability of deliberate or inadvertent human intrusion into the repository or its immediate environs should be negligible. For these reasons, human-caused disruptive scenarios were not considered in this study. As a result, the only release studied arises from a scenario in which the radionuclides are leached from the waste after the waste canisters have failed. The radionuclides are then transported from the repository by the small amounts of water normally present in crystalline rocks at depth.

Among the prerequisites for use of the GETOUT code (see Section 5.2) in determining radionuclide geohydrological transport are the estimations of two release parameters: the time elapsed after emplacement before release begins and the release duration (a constant release rate is implied). In this section these estimates are made for the waste forms, based on the repository description presented in Section 2.

In the INFCE hard-rock repository [2] containing vitrified waste in canisters and concreted waste in drums, release of radionuclides is from the waste forms

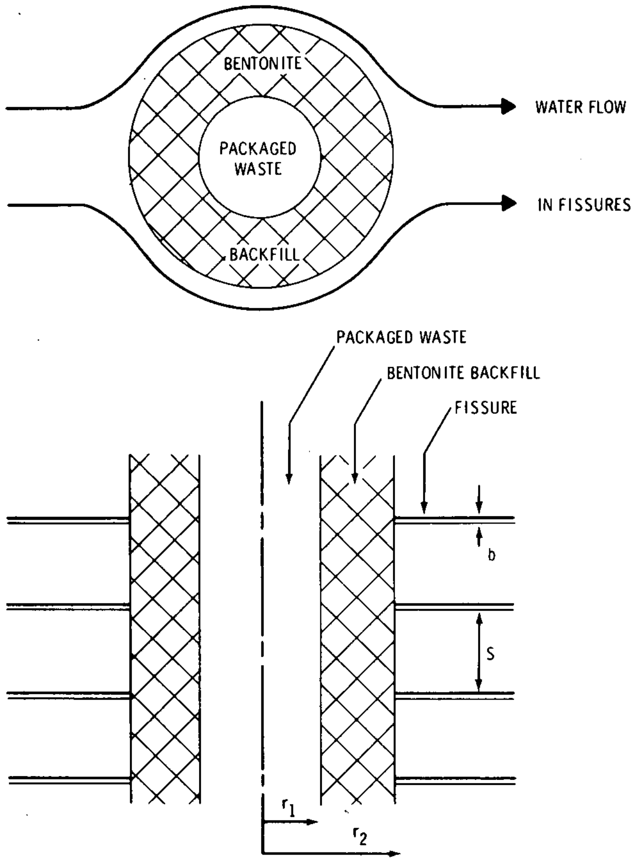

FIG.13. Schematic of packaged waste and bentonite backfill (dimensions used for estimating mass transfer resistance).

through a compacted bentonite backfill, and into groundwater within rock fissures. Canisters of vitrified waste are emplaced in individual holes; for this case the release rate can be estimated by taking into account the resistance to mass transfer of the bentonite buffer.

Neretnieks [3] has treated the estimation of mass transfer resistance of homogeneous bentonite annuli fitted inside a hole in fractured rock. On absorbing water, bentonite swells and exerts pressure on its surroundings. Confined in a hole, the bentonite fills gaps and interstices, so that the assumption of homogeneity is reasonable. The canisters are designed to resist the pressure. It is assumed that the hole intercepts a set of parallel, horizontal fissures in which groundwater flows around the bentonite annulus, as illustrated in Fig.13.

Because of the very low permeability of compacted bentonite, it was assumed that water flow in the bentonite would be too small to increase mass transfer significantly beyond that due to diffusion alone. It is likely that under its swelling pressure bentonite would intrude into the fissures, thereby extending the buffer region and its attending mass transfer resistance; for purposes of this study it was assumed that no intrusion would occur.

The ion-exchange capacity of bentonite can further inhibit release by retaining the shorter-lived or more strongly adsorbed radionuclides long enough for significant radioactive decay to occur. However, any such adsorptive retention must necessarily be preceded by release of canister corrosion products to the bentonite, which might become adsorbed and thereby reduce the exchange capacity. For this reason, no ion-exchange capacity for the bentonite was assumed; however, any retention of canister corrosion products would not be expected to affect the mass transfer resistance described above.

For the purpose of this study it was assumed that stainless steel canisters and drums are used that last 100 years. This fixed the time elapsed before release begins. Realistically, failures of the "100-year" canisters assumed for this study would be distributed over many years, but for the purpose of the study, simultaneous failures were assumed.

4. REPOSITORY ANALYSIS

As discussed above, the bentonite backfill around emplaced waste creates a barrier to mass transport. The mass transport can be controlled in three basic ways: by a limiting release rate from the waste form to the backfill, by a limiting concentration difference across the backfill, or by a limiting flow of water past the waste and surrounding backfill. Because of the bentonite diffusion barrier, it was assumed that limiting concentration differences control the release for individually emplaced canisters, as discussed below.

The release rate, N, through the annular backfill surrounding the vitrified waste and into groundwater flowing in the fissure is given by equation

$$N = 2\pi L \left[\frac{b/S}{\frac{\ell n(r_2/r_1)}{D_1}\,\frac{b}{\delta} + \frac{1}{k_v r_2}} \right] \Delta_c \qquad (1)$$

where the dimensions S, b, r_1, and r_2 are identified in Fig.13, and

δ $= (S-b)/(\ell n\, S/b)$

L is the length of the hole

D_1 is the diffusivity for mass transport in the bentonite

Δc is a concentration difference, and

k_v is a "film coefficient" for mass transfer across the concentration boundary layer in the flowing groundwater adjacent to the bentonite annulus.

Appropriate values for D_1 have been determined experimentally by Neretnieks [3].

To a good approximation [3],

$$K_v = \frac{2}{\pi}\sqrt{\frac{D_v U_p}{r_2}} \tag{2}$$

where D_v is the mass diffusivity in water and U_p the water velocity in the fissure.

The concentration difference, Δc, is taken to be the difference between an appropriate solubility concentration inside the annular backfill and the concentration in flowing groundwater. Neglecting the latter, Δc is equal to the solubility. The release rate, N, can be used to determine the dissolution rate of the waste form.

The dissolution of vitrified waste within a diffusion barrier is not straightforward to analyse because no constituent dominates. Leaching of silicate glasses typically proceeds by ion exchange and slow hydrolysis of silica, although the scenario for complicated waste glasses is poorly understood. In the absence of a diffusion barrier, vitrified waste initially releases its radionuclides disproportionately to their concentration in the waste; the ratios of release rates relative to concentrations in the waste can differ by several orders of magnitude among the various radionuclides. With a diffusion barrier radionuclides would probably also be released disproportionately, but the differences among radionuclides would be difficult to predict. Since silica is usually a major constituent of vitrified waste, and its hydrolysis is closely related to glass leaching, it was assumed in this analysis that release from vitrified waste within a diffusion barrier is that corresponding to the diffusion of amorphous hydrated silica, with other species being released in proportion to their concentration relative to silica.

The release rate from vitrified waste will decrease rapidly from an initial maximum as the temperature decreases; the later release rate for the subsequent long time period is the appropriate parameter for the GETOUT calculations. The leach duration, based on the above considerations and appropriate quantitative data, was calculated to be 8×10^5 years.

It was further envisioned that medium- and low-level wastes would be emplaced in the repository in steel drums. The geometry of drums stacked within sand-bentonite backfill is rather complex. As the mass transfer resistance is difficult to estimate for that waste, it was assumed that a limiting flow of water would control the release. Uranium is the major waste constituent. The release rate is estimated to be controlled by uranium solubility and the water flow intercepted by the drums. Based on an assumed water velocity in the sand bentonite of twice the calculated bulk groundwater velocity ($0.6\ m^3 \cdot m^{-2} \cdot a^{-1}$),

TABLE XI. CALCULATED DURATIONS OF LEACHING FOR DRUMMED WASTES AND RELEASE RATES FROM REPOSITORY
(Basis: 100 GW(e) · a)

Waste type	Duration (years)	Release rates (Ci/a)
Conversion	4.7×10^4	4.1×10^{-4}
Enrichment tails	6.7×10^6	5.3×10^{-4}
UO_2 fabrication waste	9.6×10^3	6.2×10^{-4}
MOX fabrication waste – U	5.4×10^2	6.4×10^{-4}
– Pu	2.7×10^1	6.7×10^1
Depleted U from reprocessing	6.6×10^4	5.6×10^{-5}

the maximum amount of water available was calculated to be 0.12 L per drum per year. By multiplying this amount by the solubility of uranium (1000 mg/L) a leach rate was estimated. Though probably conservative, the same rate was used for plutonium in the waste from mixed-oxide fuel fabrication. The duration of the leaching was then obtained by dividing the uranium and plutonium contents of the drums by the leach rates. No account was taken for retardation or dispersion in the sand-bentonite barrier. Without inclusion of these latter factors, because of limitations of time for the study, the study yielded unrealistically short leach durations for some of the waste categories. The release rates and leach durations for the various waste categories are presented in Table XI.

According to INFCE assumptions, iodine released during reprocessing is not incorporated into vitrified waste but instead is adsorbed on special filters. It is assumed that iodine is immobilized on silver zeolite and emplaced in concrete in drums. For the given iodine inventory the number of drums can be estimated from the capacity (1.46 g I/kg) and density (1200 kg/m^3) of the silver zeolite, so that the appropriate absorption area can be calculated. The release rate is taken to be the product of the solubility of silver iodide (1.5×10^{-6} g/L) and the water flow rate intercepted by the drums (0.12 L/drum per year). The leach duration, based on the above considerations, was calculated to be about 2×10^9 years.

5. GEOSPHERE ANALYSIS

5.1. Hydrological models

5.1.1. Three-dimensional finite-element groundwater flow model

For this study only normal slow water movement through the low permeability rock was considered. Consideration of the effects of highly fractured zones on regional water flow and the decrease in permeability and porosity of the rock masses and fracture zones with depth requires the use of a three-dimensional numerical model to study water movement. To model the multilayered geohydrological system with discrete highly fractured zones described in the following sections, the three-dimensional finite-element groundwater flow model, described in Section 5.11 of Appendix A, was used. The previous discussion is applicable here.

5.1.2. Hydrological model of the conceptual waste repository

The first stage of the modelling effort involved examining the geohydrological data and developing a conceptual model of the system. Much of the information presented here was extracted directly from Appendix 2 of Ref.[1], but additions and adjustments were made as required to define the system for the modelling effort. The generic geohydrological description of the granite waste repository site, presented in Section 2, is expanded in the next paragraphs and is followed by the translation to a conceptual model and finally to an input data set for use with the groundwater flow model.

Figure 14 illustrates the boundaries of the conceptual repository site in granite along with the regional water-table configuration and assumed boundary definitions. Although an actual site might contain some surface layers of till and clay, these layers would tend to retard deeper circulation patterns and were ignored in the description so as to favour conservatism. The stratigraphy was thus assumed to be granite from the surface down. Recharge calculations are avoided by holding the water surface according to a subdued version of the actual topography. The reference topography assumed was a version of an actual granite area modified slightly to be consistent with the assumed boundary conditions. Regional tilt is of an order of 10^{-3} m/m, and local topographic variations are from near zero to 10^{-1} m/m. The site shows a topography common for glaciated hard-rock areas in Precambrian shields. The regional discharge site was assumed to be both large fresh and salt water bodies. The boundary conditions include: no-flow boundaries on the east and west, a vertical no-flow boundary to the south since this is assumed to be the regional groundwater divide, and a lake or ocean boundary to the north which is held at the lake or ocean elevation on the lake or ocean surface

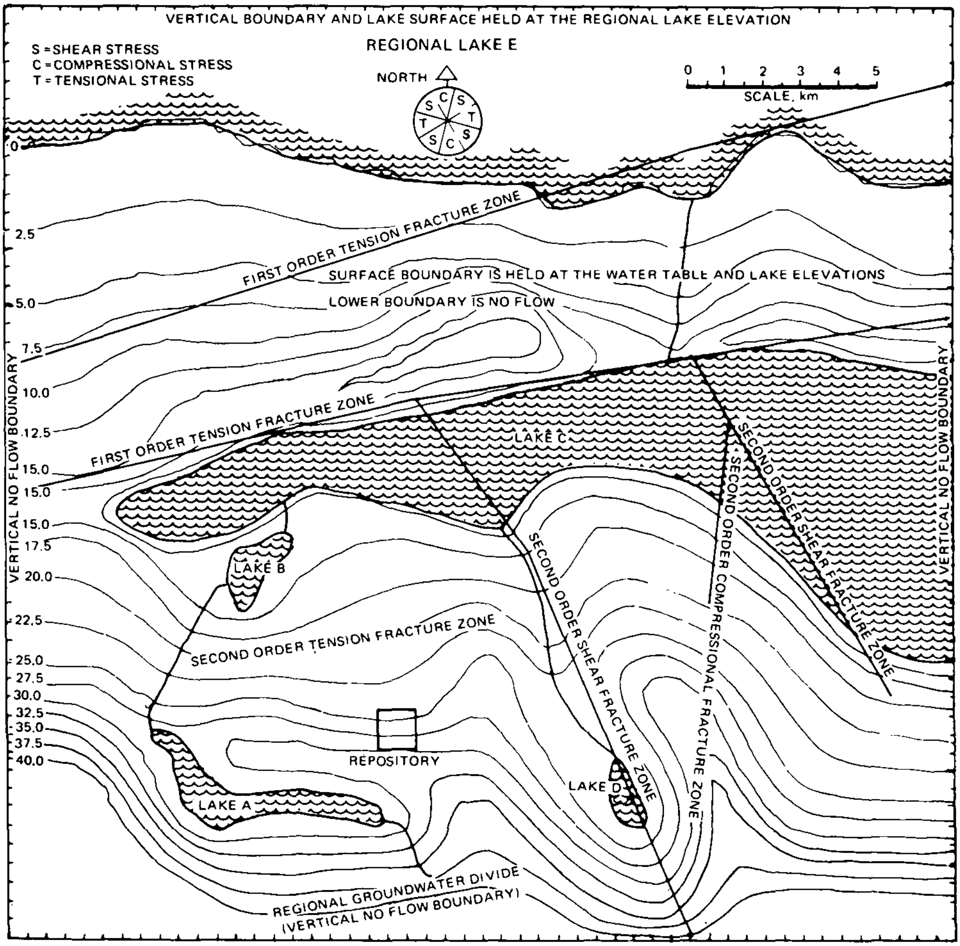

FIG.14. Reference repository site in granite, illustrating boundaries, fractures and water table.

and along the vertical northern boundary; consistent with data observed for hard-rock geologies, permeabilities and porosities decrease with depth. For modelling purposes, the upper 1.5 km is modelled in detail and the lowest layer is of sufficient thickness to avoid interference by the assumed no-flow boundary along the lower surface.

Major shear or fracture zones in patterns like those observed in an actual granite area have been included. These fracture patterns at an actual site should be spaced so that blocks (undisturbed by major fracture or shear zones) of the appropriate size for repository siting are available. All major fracture zones are assumed to be vertical. In accordance with a generic stress distribution in the rock, the fracture zones are divided into tension, shear and compression zones.

As discussed in Section 2, the flow in the hard-rock masses of the reference repository site is governed by all the properties of fracture – orientation, spacing, interconnection and aperture – including the three-dimensional effect of the stress field on aperture size. The equivalent porous medium approach has been used to describe the regional water movement.

The regions of major fracture or shear zones as determined from geophysical methods are handled as discrete features. The equivalent porous medium permeability and porosity were chosen to reflect the higher degree of connectivity between fractures in these zones, the greater fracture permeabilities and aperture sizes (as they relate to porosity), the closer spacing of fractures, and the presence or absence of clay in the fractures of these zones. It should be noted that, when the equivalent porous medium parameters are developed from an inappropriate data base, errors can be introduced in calculating the velocity field.

5.1.3. Model input parameters

The extent of the area used in the hydrological model is 25 000 m by 25 000 m. The finite-element grid representation of the area is shown in Fig.15. Element size, shape and orientation were chosen so as to represent the actual topography and structural properties of the granite mass within the modelled region in the best manner. The regional groundwater divide to the south has an altitude of 40 to 45 m above the level of the regional lake (or sea) elevation. The vertical boundary of the regional lake or sea is placed about 4000 m from the shore-line, and it is a held potential. The major fracture zones are represented by discrete elements. The widths of these discrete elements were assigned according to the type of fracture zone. First- and second-order tension zones were assigned widths of 50 and 10 m, respectively. Shear zones and compression zones were assigned widths of 20 and 5 m, respectively. The model works with discrete layers in the vertical direction in which the permeability and porosity are set constant. Typical values of the hydraulic properties were used for the different geohydrological structures at the ground surface and at various depths. The same slope was assumed for the different geohydrological units, but they started at different surface values.

5.1.4. Geohydrological model results

The output from the hydrological model is the groundwater potential distribution throughout the modelled region. An auxiliary programme for the model calculates the travel time, travel path, and travel distance along any streamline within the region. These values are calculated from the predicted potentials and from the input values used for permeability and porosity.

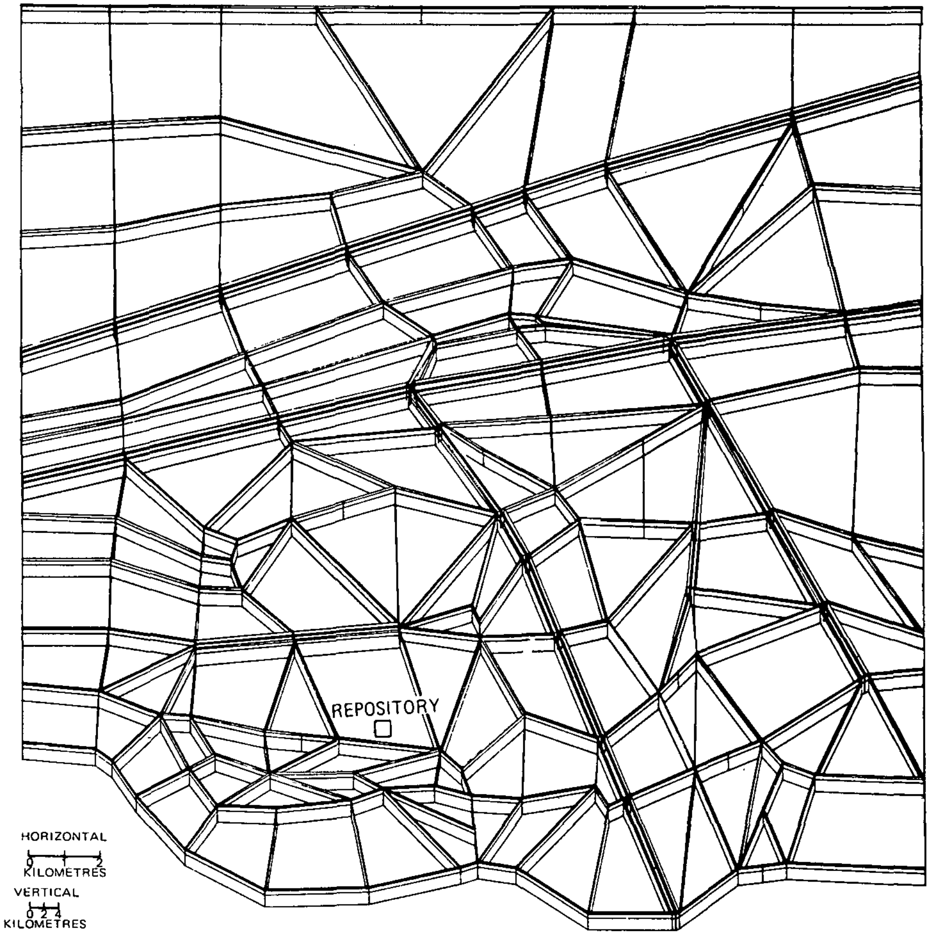

FIG.15. Finite-element grid representation of the model region.

Figures 16 and 17 illustrate the X-Y paths that water, entering the four corners and middle of the generic repository, would take as it proceeds to the discharge site in Lake C. Figure 18 illustrates the Y-Z paths for the same five streamlines. The dots along the streamlines are placed 1000 years apart in time. Notice that, as the streamlines encounter the second-order tension zone en route to Lake C, they move upwards and westwards because of the higher permeability of the fracture and the disjuncture in gradients. The total travel time and distance for

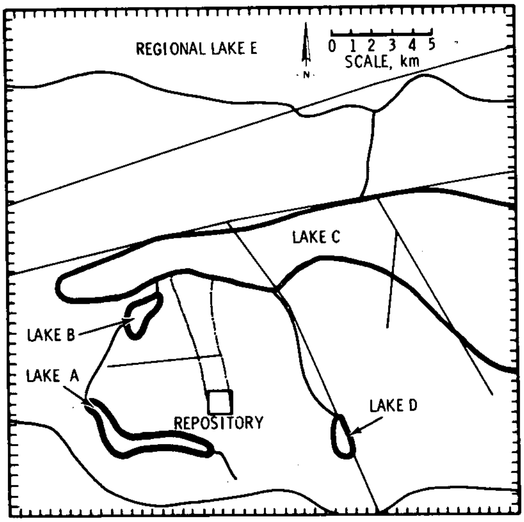

FIG.16. X-Y paths for streamlines 1 and 2 starting at the lower left and right corners of the repository. The dots on the streamline paths are places 1000 years apart in time.

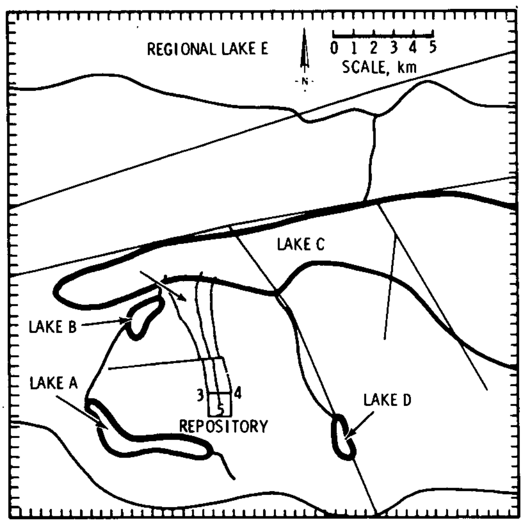

FIG.17. X-Y paths for streamlines 3-5 starting at the upper left and upper right corners of the repository as well as the middle. The dots along the streamline paths are placed 1000 years apart in time.

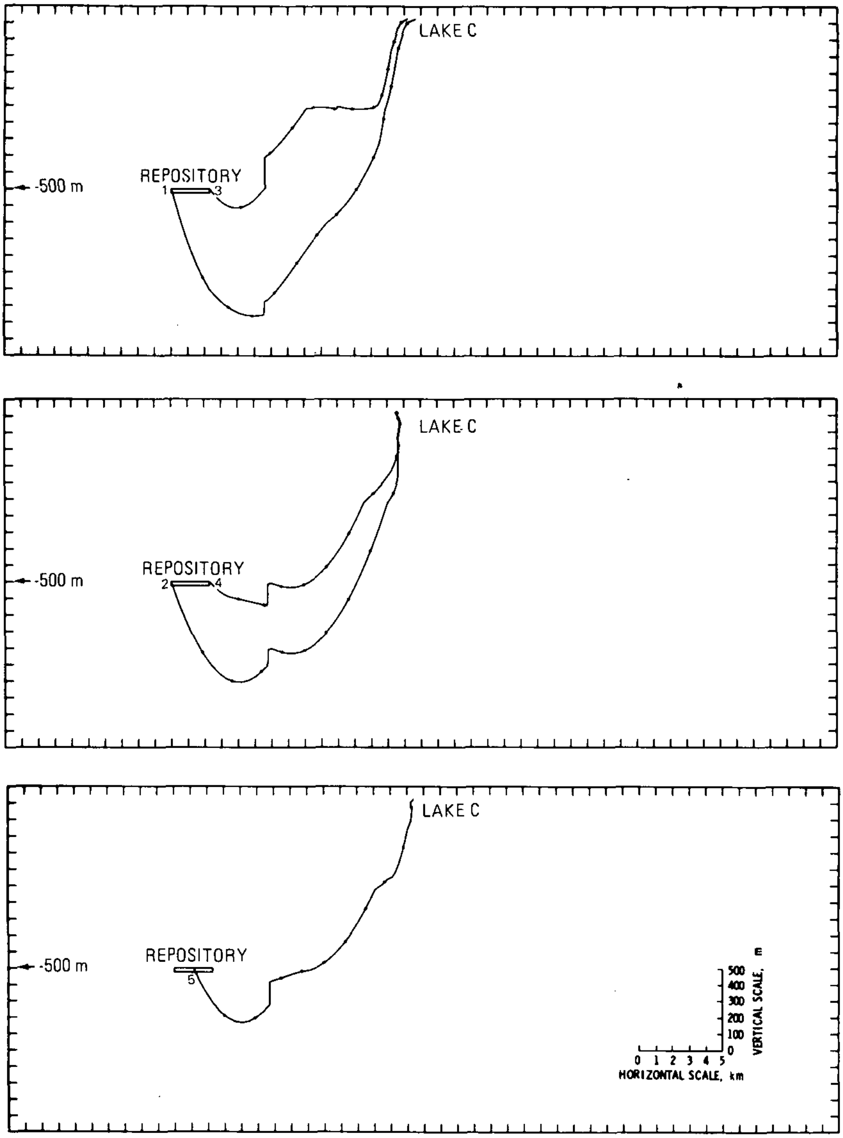

FIG.18. Y-Z paths for the five streamlines starting at the four corners and middle of the repository. The dots along the streamline paths are placed 1000 years apart in time.

each of the five streamlines are illustrated in the figures. The average streamline parameters are:

Average distance	–	7100 m
Average travel time	–	11 700 years ± 1300 years
Average velocity	–	0.61 m/year

Since the transport is to be simulated with a one-dimensional model, the hydrological model results must be reduced to an equivalent, one-dimensional data set. The one-dimensional model requires a column length, pore water velocity, dispersion length, column width, column porosity, column height and column flow. The one-dimensional column parameters are related as follows:

Flow = width X height X pore velocity X porosity.

The flow through the repository was estimated from the average X, Y, Z flux per unit area at the corners and middle of the repository. The resultant one-dimensional flow is thus 2.6 m^3/a.

The one-dimensional column width of 1600 m was estimated from Figs 16 and 17. Pore water velocity of 0.61 m/a was obtained from the quotient of the average streamline distance and travel time. A porosity estimate of 1.9×10^{-4} was taken from the average of the time-weighted porosity along each of the five streamlines. The theoretical column height of 14.2 m was chosen to be consistent with the flow, width, velocity and porosity estimates. A column length of 7100 m ensures the appropriate average travel time for a velocity of 0.61 m/a. The ± 1300 years variation in arrival time due to flow geometry can be accounted for by appropriate choice of the minimum longitudinal dispersion length. For a one-dimensional problem a ± 1300-year spread in arrival time for a conservative contaminant is due to a ± 800 m (± 1300 X 0.61) spread in the contamination plume. The sigma for a one-dimensional transport problem is given by the square root of the product of twice the dispersion length X the velocity X the travel time. Equating 800 m to the transport sigma, the equivalent dispersion length required to yield a ± 1300-year spread is 42 m.

5.2. Transport models

5.2.1. One-dimensional multi-component mass transport model

The model used for the nuclide migration calculations is a one-dimensional transport model that includes axial dispersion, geochemical retardation, and radioactive chain decay. The model is called GETOUT [4, 5]. The version used in this study is mathematically equivalent to that used in Ref.[5], i.e. the dispersion is omitted for those nuclides formed by chain decay during the migration process. However, daughter-product dispersion was accounted for using dispersion factors generated by GETOUT for parent radionuclides. This procedure has been shown to give comparable results to a modified version of GETOUT [6].

The model is based on the analytical solutions of a set of partial differential equations of the form:

$$K_i \frac{\partial N_i}{\partial \theta} = \frac{1}{Pe} \frac{\partial^2 N_i}{\partial \eta^2} - \frac{\partial N_i}{\partial \eta} - K_i R_i N_i + K_{i-1} R_{i-1} N_{i-1} \qquad (3)$$

where:

N_i = number of moles of nuclide i
θ = dimensionless time = $t \times u/L$
 t = time, years
 u = groundwater velocity, m/a
 L = column length, m
η = dimensionless length co-ordinate = Z/L
 Z = length co-ordinate, m
Pe = Peclet's number = $u \times L/D$
 D = dispersion coefficient, m^2/a
K_i = nuclide retentivity = u/u_i
 u_i = nuclide velocity for nuclide i, m/a
R_i = decay number for nuclide i = $\lambda_i \times L/u$
 λ_i = decay constant for nuclide i, $year^{-1}$.

Equation (3) is solved for a constant release rate of the waste. The retardation factor, K_i, is based on the assumption that for trace contaminants the geochemical interactions are reversible ion-exchange or adsorption reactions where the concentration of radionuclides sorbed on the rock is proportional to the concentration in the groundwater. From the laboratory measurements of the distribution coefficient, K_d (m^3/kg), the retention factor for a fissured rock can be calculated:

$$K_i = 1 + K_a \cdot a \frac{(1-\epsilon)}{\epsilon} \qquad (4)$$

where:

$$K_a = \frac{K_d}{a_2} = \text{surface distribution coefficient, m}$$

a_2 = conversion factor equivalent to surface area of the laboratory sample, m^2/kg
ϵ = the porosity of the rock

Equation (4) is based on the assumption that the sorption reactions take place only at the surface of the crack walls. Experimental measurements [7] of the conversion factor, a_2, indicate a significant kinetic effect, possibly diffusion into the microfractures of the rock mass. This would increase considerably the retention factors compared with the values used in this study.

5.2.2. Input parameters

Input data for use in the GETOUT model calculations include the radionuclide inventory, the release scenario results presented in Section 4 and additional geochemical and hydrological data. Whereas in the hydrological model (Section 5.1.2) both the permeability and the porosity can vary with depth, for the GETOUT model a homogeneous migration column is assumed; thus, these values as well as the resulting groundwater flow and velocity must be averaged before being used in the GETOUT model, as discussed in the next paragraph.

The hydrological model yielded an average groundwater travel time of 11 700 years. In the actual transport modelling this has been rounded to 10 000 years. The geometry of the repository gives a spread in the travel time for the five streamlines corresponding to a standard deviation of about 1300 years. If this variability were interpreted in terms of the dispersion model defined by Eq.(3), it would correspond to a dispersion coefficient of 8.1×10^{-7} m^2/s. It has, however, not been verified that this variation of transport times can be interpreted as a dispersion coefficient and applied in the GETOUT model. The travel-time distribution in the dispersion model arises from velocity fluctuations and molecular diffusion within a flow tube. However, the distribution obtained from the hydrological model is due to differences between the flow tubes and, hence, mainly dependent on the topography, permeability and porosity. Because of these uncertainties, a value of 1.5×10^{-9} m^2/s, accounting only for the molecular diffusivity for ions in water, was chosen. This implies, however, that the influence of dispersion on the maximum discharge rate is not fully taken into account. The average permeability and porosity have been evaluated as the time-integrated means; the permeability obtained this way was 6×10^{-9} m/s, and the porosity 2×10^{-4}.

Regarding geochemical input data for the transport model calculations, as discussed previously, chemical interactions between the dissolved waste nuclides in the groundwater and the rock result in a retardation effect quantified by a retention factor that is specific for a given element in a certain chemical and geological environment. The distribution coefficients, K_d (mL/g), used for the calculation of the retardation factors were taken as the best estimate values used in the study described in Appendix D. The retention factors were then calculated from Eq.(4). The factor a_2 for the conversion of the K_d values to the surface distribution coefficients, K_a, was assessed from K_a measurements on a natural crack surface [7]. The a_2 values were obtained by dividing the K_d values, measured for crushed rock, by the measured K_a values. Three different elements were used in the K_a measurements: strontium, caesium, and americium. The a_2 values obtained for these elements were 3, 2, and 10 m^2/kg, respectively. In this study the caesium value was used only for caesium, the strontium value for both strontium and radium, and the americium value for the rest of the waste elements. The differences in the a_2 factor for the various elements reflect their different chemical behaviour in the geological environment. The use of americium value for the bulk of the elements is justified by similarities in the behaviour (e.g. formation of insoluble hydroxide complexes). This assumption also results in the lowest retardation factors, thus yielding the shortest nuclide migration times.

The area of fissure surfaces per unit volume of rock, a (m^{-1}), can be calculated from the average fissure spacing, s (m), assuming that the fissure surfaces are planar and parallel:

$$a = 2/s \tag{5}$$

If the porosity of the rock is interpreted as the result of parallel fissures with planar walls, the fissure spacing can be obtained from:

$$s = \frac{\left(\frac{12\nu}{g} \cdot K_p\right)^{1/2}}{\epsilon^{3/2}} \tag{6}$$

where:

g = gravitational constant, m/s^2
ν = kinetic viscosity, m^2/s
K_p = parallel fissure permeability, m/s

Based on an average porosity of 2×10^{-4} in the flow tube and the average permeability of 6×10^{-9} m/s, the average fissure spacing was calculated as 0.03 m. According to Eq.(5) this spacing yields a value for the parameter, a, of about 67 m^2/m^3 rock mass.

The geochemical input data to the transport calculations are summarized in Table XII. The retention factors were adjusted for the porosity and permeability used in this study.

5.2.3. Transport model consequences

The nuclide discharge rates to the biosphere for high-level waste were calculated with the GETOUT model. The results are given in Table XIII as maximum discharge rates and times of these maximums. The nuclide travel

TABLE XII. DISTRIBUTION COEFFICIENTS, K_d, SURFACE DISTRIBUTION COEFFICIENTS, K_a, AND RETENTION FACTORS, K_i, USED IN TRANSPORT CALCULATIONS

Element	K_d (mL/g)	K_a (m)	K_i
Sr	0.016	0.008	2700
Tc	0.05	0.005	1700
I	0	0	1
Cs	0.064	0.021	7000
Ra	0.50	0.25	84000
Th	2.4	0.24	81000
Pa	0.6	0.06	20000
U	1.2	0.12	41000
Np	1.2	0.12	41000
Pu	0.30	0.03	10000
Am	32	3.2	1080000
Cm	16	1.6	540000

TABLE XIII. SUMMARY OF INFCE HARD ROCK REPOSITORY TRANSPORT MODEL RESULTS FOR HLW

Isotopes	Time of peak discharge (years)	Maximum rate of discharge (Ci/a)
^{135}Cs	7.1×10^{7}	7.2×10^{-11}
^{226}Ra	4.1×10^{8}	1.0×10^{-6}
^{230}Th	4.1×10^{8}	1.1×10^{-6}
^{232}Th	8.1×10^{8}	3.2×10^{-9}
^{231}Pa	4.1×10^{8}	1.1×10^{-6}
^{234}U	4.1×10^{8}	2.2×10^{-6}
^{235}U	4.1×10^{8}	1.7×10^{-6}
^{236}U	4.1×10^{8}	8.4×10^{-11}
^{238}U	4.1×10^{8}	2.2×10^{-6}

TABLE XIV. SUMMARY OF INFCE HARD ROCK REPOSITORY TRANSPORT MODEL RESULTS FOR NON-HLW

Isotopes	Maximum rate of discharge (Ci/a)
^{226}Ra	3.4×10^{-4}
^{230}Th	3.5×10^{-4}
^{231}Pa	1.7×10^{-5}
^{234}U	7.4×10^{-4}
^{235}U	8.4×10^{-6}
^{238}U	7.4×10^{-4}

times are generally around hundreds of million years. An exception is ^{129}I which migrates at the groundwater velocity with a travel time of about 10 000 years; however, as discussed in Section 4, because of its extremely low solubility and consequent long leach duration, its release rate is expected to be insignificant. The long travel times allow most of the nuclides to decay, leaving only those with extremely long half-lives and those arising from long-lived parent nuclides; these are ^{238}U along with its daughter nuclides ^{234}U, ^{230}Th, and ^{226}Ra and ^{235}U with its daughter nuclide ^{231}Pa.

The transport of nuclides released from non-high-level wastes, as discussed in Section 4 was treated separately. The ^{235}U, ^{238}U and ^{239}Pu are the only nuclides released that will result in any significant discharge rates to the biosphere. The other nuclides have either small inventories or will decay during the long transport times. Table XI presented the leach durations and the resulting release rates. The release rates of ^{238}U were used to calculate the discharge rates to the biosphere for the daughter nuclides ^{234}U, ^{230}Th and ^{226}Ra. The discharge rates of ^{231}Pa and ^{235}U were calculated from the release rates of ^{239}Pu and ^{235}U. The maximum discharge rates to the biosphere are presented in Table XIV.

It should be stressed that the calculated leach durations in several of the cases are extremely short and that an analysis of the diffusion resistance in the buffer material would yield much longer durations. It should also be pointed out that the leach durations might be extended by improved waste forms and packages.

In summary, all discharge rates to the biosphere calculated by the transport model are small in magnitude. No fission-product peaks of significance appeared. The peak concentrations for actinides all occur at 400 million years or later.

6. BIOSPHERE ANALYSES

The biosphere environment selected for this generic study is representative of a typical granite site. This environment includes an inland lake with local farmlands and drainage to a larger intermediate lake[1] or to the sea. Lake C (Figs 16 and 17) receives radioactivity from the deep groundwater as defined in Section 5.1 of this appendix. The biosphere model, described in Section 6.1 uses the deep groundwater activity release rate to determine the radioactivity in each compartment as a function of time. The compartment activities are used in the pathway analysis to determine the rate of radionuclide intake for the most exposed individual. The pathways of uptake by people are described in Section 6.2. The radionuclide intake rates are used to calculate the dose received in the fiftieth year of exposure by the most exposed individual. The dosimetry models are described in Section 6.3.

Doses presented in this report are the annual doses for the most exposed individual rather than the 50-year accumulated doses presented for the INFCE salt repository study [8] described in Appendix A. Therefore, the dose results in the two studies cannot be directly compared.

6.1. Biosphere transport model

Transport of radionuclides in the biosphere is described by the multi-compartment model of Bergman et al. [9]. In this model the ecosystem is divided into a number of physically well-defined areas or volumes. In the following discussion these are referred to as compartments. The quantity of radionuclide in each compartment is described by a system of linear first-order differential equations expressed mathematically in vector form as follows.

For parent radionuclides:

$$\dot{Y}_P(t) = K_P Y_P(t) + Q_P(t) - \lambda_P Y_P(t) \qquad (7)$$

For daughter radionuclides:

$$\dot{Y}_D(t) = K_D Y_D(t) + \lambda_D Y_P(t) - \lambda_D Y_D(t) \qquad (8)$$

The vectors Y_P and Y_D refer to activity in the system's compartments at time t, and the vectors $\dot{Y}_P$ and $\dot{Y}_D$ represent changes in activity per unit time. The coefficient matrix K (year^{-1}) describes the constant transfer rates between the compartments, and the vector $Q_P(t)$ describes the production or release within the compartment (activity per year). The daughter activity source strength within

[1] Called a regional lake in Section 5.1.3.

each compartment is a function of the parent activity in the compartment. The constants λ_P and λ_D are the radioactive decay constants for the parent and daughter radionuclides, respectively.

Inherent in the use of this mathematical model are the assumptions:

The radionuclide outflow from a compartment is dependent on the amount of the radionuclide in it and on the compartment's transfer parameters;

The compartment is instantaneously well mixed; and

Each unit of activity has the same probability of leaving the compartment.

The model of the biosphere is divided into three subsystems of progressively increasing size referring to a regional, an intermediate, and a global ecosystem. This subdivision of the environment into three ecosystem zones makes it possible to:

Study extreme exposure situations in limited ecosystems to help define the most exposed individual;

Increase the realism of the dispersal pattern described by the model by considering gradual dispersal on an ever-increasing scale as well as feedback between different zones; and

Apply the model adequately to typical conditions by choosing a large lake or a sea as an intermediate zone.

Figure 19 shows the compartments considered and their pathways of interaction. Radioactivity from the repository enters the regional ecosystem through the deep groundwater in contact with the inland lake from which dispersal in the ecosystems begins. The lake has an area of 350 km^2 and an average depth of 20 m. The area of the sediment layer is the same as the lake's, with an upper layer of sediment 10 cm deep assumed to participate actively in the processes of exchange with overlying water. The regional soil compartment consists of 900 km^2 farm land. The soil compartment in the region is considered to have an average depth of 2 m. Subsurface groundwater includes all soil water and groundwater down to a depth of 2 m and is not a primary recipient for the radionuclides from the repository. The average period of turnover for the subsurface groundwater is assumed to be three years. There is hydrological equilibrium within a precipitation area.

The sea system comprises a surface area of 3.7×10^5 km^2 and an average depth of 60 m. The sediment compartment is the sediment layer at the bottom of the sea. The atmosphere above the regional and sea area is the tropospheric air volume up to an altitude of 1 km.

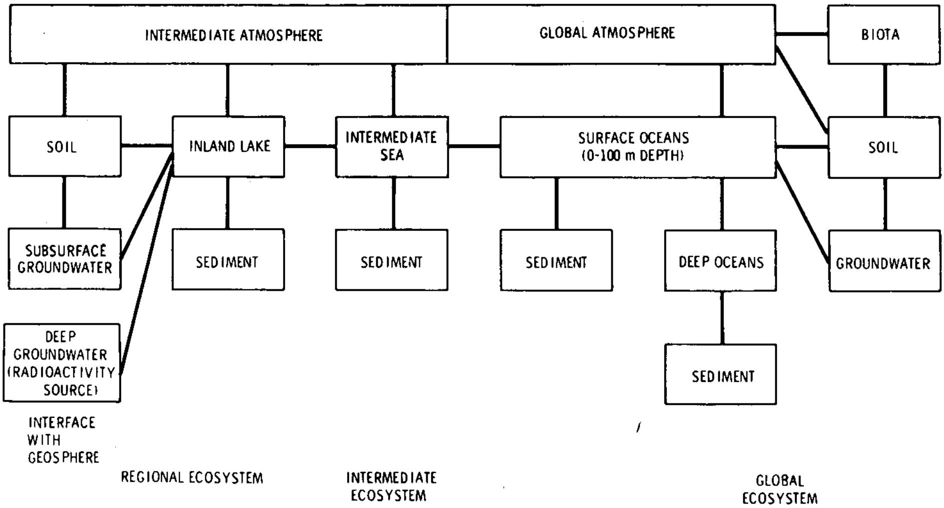

FIG.19. Compartments of the various ecosystems.

The global ecosystem embraces seven compartments[2] that are considered important for the dispersal and turnover of long-lived radionuclides. The oceans are divided into two compartments because mixing and exchange in the seas decrease rapidly with increasing depth. The surface ocean consists of the upper water layer down to a depth of about 100 m. The deep ocean basin is below the surface ocean. These two compartments connect directly with their respective sediment compartments. The uppermost sediment compartment encircles the continents and amounts to about 4% of the total sediment area.

The soil compartment consists of the upper ground layer on the continents down to a depth of 0.5 m. The groundwater compartments, which transports the nuclides to the surface water and back to the soil compartment, connects with the soil compartment. The biota compartment consists of the terrestrial short- and long-lived primary producers, i.e. vegetation that has a short life cycle of up to a few years and vegetation with a life extending over several decades. The biota is particularly important for the turnover of carbon, iodine, and technetium.

The turnover of radioactive elements in the biosphere takes place in relation to the movement of certain carriers in different media. Through irrigation as well as dry and wet deposition, radioactive substances can be transferred to the ground,

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[2] The masses and areas of the various compartments are presented in Ref.[1].

while re-suspension, leaching and runoff are responsible for transport in the opposite direction to the atmosphere, subsurface groundwater and lake water. In the lake the activity settles out and is re-suspended while at the same time it is carried to the sea through water turnover. Exchange of activity between water and sediment occurs there as well. The sea is connected with the global ocean area. Exchange takes place between the global atmosphere and the ocean by means of evaporation, precipitation, and aerosol formation. Radioactive elements are recirculated in the global land area by means of re-suspension, leaching, and runoff.

The structure of the model permits the recirculation of radioactive elements between different parts of the compartment system. The exchange of radionuclides between the compartments is described by transfer coefficients which give turnover per unit time. Water balance calculations and hydrological information concerning water turnover on a regional and global scale are used in cases where groundwater and surface water are carriers. With this as a basis and with the aid of distribution coefficients determined by the mobility of the nuclide in relation to that of water, nuclide-specific coefficients for transfer between soil and water are obtained.

Studies of fallout radioactivity from nuclear weapon tests have provided information on the dispersal and deposition of a number of nuclides in various media. Leaking storage facilities and releases have also contributed to information on how elements migrate in soil and water [10–14]. The distributions of the stable isotopes of the radioactive elements, or of chemically analogous elements in the different compartments, as well as experimental data from field and laboratory studies, have also been used in the model [11, 15–18]. The transfer parameters with derivations are reported in Ref.[9].

6.2. Exposure pathway model

The environmental concentrations generated by the biosphere transport model are used in the exposure pathway analysis to estimate the total intake by the most exposed individual from ingestion and inhalation. External exposure situations are also considered. Previous work [9] has shown the most exposed individual to be located in the regional ecosystem. Pathways that have been shown to be important are internal exposure via inhalation and ingestion of food and drinking water and external exposure from material deposited on the ground. Figure 20 illustrates these pathways in the regional ecosystem. Other pathways of interest for external exposure include bathing, beach activities and exposure to fishing tackle contaminated with lake sediments.

Internal exposure from food results from several ecological transport paths such as uptake by plant roots, by fish, and by grazing animals used for meat and milk production. The food crop and grazing pathways include contributions

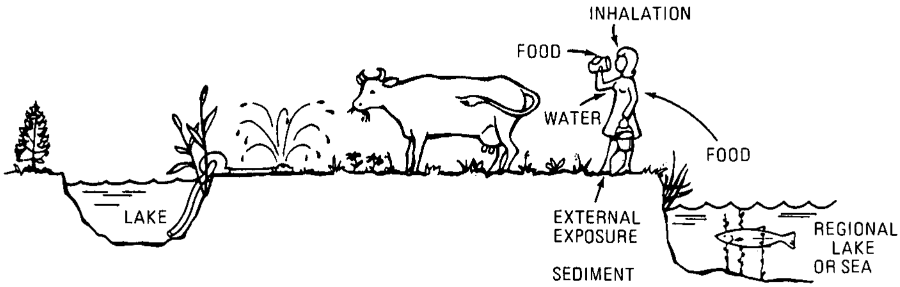

FIG.20. Paths of human exposure in the regional ecosystem.

from irrigation with lake water. Also, groundwater from irrigated areas is used for human drinking water. The exposure pathways considered in this study are detailed in Ref.[1]. The annual intake by the most exposed individual is calculated by the equations given in Ref.[9].

External exposures are calculated directly from compartment activity levels using dose conversion factors described below. The exposure is also dependent on the total time of exposure (hours/year) for each external pathway. All data parameters for the pathway analysis are reported in Ref.[9].

6.3. Dosimetry model

The pathway models consider exposures to the most exposed individual from external irradiation, inhalation of airborne radioactivity and ingestion of contaminated food and water. This section describes the models used to convert environmental concentrations to external radiation dose and to convert radionuclide intakes via inhalation and ingestion to dose.

The weighted whole-body dose is calculated for each radionuclide and each pathway. The results presented in Section 6.4 represent the dose during the fiftieth year following fifty years of chronic intake for the most exposed individual plus any external exposure received during that year.

External radiation exposure may contribute to the dose for the most exposed individual through the following pathways:

Exposure to contaminated ground (soil),
Exposure to contaminated beaches (sediment),
Exposure while bathing (lake water),
Exposure to contaminated fishing tackle (sediment).

The external exposures consider beta and gamma radiations. Gamma radiations contribute to all internal organ doses equally while beta radiations contribute only to skin dose. All external exposures are calculated in units of rem/a, with consideration of the fraction of the year that exposure is received. Details for calculation may be found in Ref.[9].

The weighted whole-body doses for the most exposed individual from ingestion and inhalation are based on the recommendations of ICRP [19]. Weighted whole-body dose conversion factors taken from the work of Adams et al. [20] were used to convert intake to dose. The factors give the doses (rem) to an individual during the fiftieth year of exposure for chronic intake at 1 Ci/a. A quality factor of 20 has been used for high linear energy transfer (LET) radiation and a factor of unity for low LET radiation, e.g. beta and gamma. The conversion factors are based on internal organ doses calculated using the lung model of the ICRP Task Group on Lung Dynamics [21] and the gastrointestinal tract model of Eve [22].

As stated previously, the radiation doses calculated for high-level waste disposal in granite cannot be directly compared with the radiation doses calculated for the INFCE salt repository [8]. Based on only the dose models, the comparison would be invalid for several reasons. Firstly, the salt repository values are doses accumulated over a 50-year period, while the granite values are annual doses in the fiftieth year following 50 years of constant chronic intake. Therefore, the two schemes for accounting for long-term exposure to radionuclides in the environment are considerably different.

Secondly, the dose methodology for the INFCE salt repository was based on ICRP Publication 2 (1959) [23], which identified certain specific organs as "critical organs". The dose methodology used for the INFCE granite site was based upon ICRP Publication 26 (1977) [19], which identifies an entirely new entity called a "weighted whole-body dose equivalent". The weighted dose is obtained by calculating doses to numerous body organs, selecting those identified either as important organs and/or those receiving the highest doses, and then multiplying these selected organ doses by weighting fractions.

A third difference is that a few of the parameters used to calculate the dose effectiveness of certain radionuclides have undergone changes between the two ICRP reports. The principal change was in the value of the Quality Factor, Q, for alpha radiation. The newly recommended value is 20, and the older value is 10.

6.4. Biosphere model consequences

The biosphere and dosimetry models were used to generate the maximum annual dose received by an individual. Table XV presents the annual maximum individual dose (by radionuclide) for the high-level waste.

TABLE XV. INFCE HARD ROCK REPOSITORY: ANNUAL MAXIMUM INDIVIDUAL DOSES FOR HIGH-LEVEL WASTE

Radionuclide	Time of maximum (years)	Dose (rem/a)
^{135}Cs	7.1×10^{7}	1.7×10^{-14}
^{226}Ra [a]	4.1×10^{8}	1.1×10^{-9}
^{230}Th	4.1×10^{8}	3.1×10^{-10}
^{232}Th	8.1×10^{8}	5.0×10^{-11}
^{231}Pa	4.1×10^{8}	2.7×10^{-6}
^{234}U	4.1×10^{8}	1.6×10^{-10}
^{235}U	4.1×10^{8}	5.0×10^{-11}
^{236}U	4.1×10^{8}	1.1×10^{-14}
^{238}U	4.1×10^{8}	2.6×10^{-10}
$^{230}Th/^{226}Ra$ [b]	4.1×10^{8}	2.4×10^{-8}
$^{234}U/^{226}Ra$ [c]	4.1×10^{8}	6.0×10^{-10}
Maximum annual total dose	–	2.7×10^{-6}
Time of maximum total dose (years)	–	4.1×10^{8}

[a] Refers to ^{226}Ra which reaches the biosphere directly from the groundwater.
[b] Refers to ^{226}Ra produced by radioactive decay of ^{230}Th in the biosphere.
[c] Refers to ^{226}Ra produced by radioactive decay of ^{234}U (via ^{230}Th) in the biosphere.

Table XVI presents the dose results for the non-high-level waste category. As discussed above, the dose represents the weighted whole-body dose received by the most exposed individual during the fiftieth year following 50 years of chronic intake.

The timing and magnitude of doses to the most exposed individual parallel the radionuclide discharge rates to the biosphere (Section 5.2.3). The highest doses appear at 400 million years. The calculated total dose of 0.003 mrem per year for high-level waste is far below the average annual background dose rate of 100 mrem per year. The total dose for non-high-level waste was 0.05 mrem. The main contributors to dose were ^{231} Pa and ^{226} Ra through the ingestion pathways.

TABLE XVI. INFCE HARD ROCK REPOSITORY: ANNUAL MAXIMUM INDIVIDUAL DOSES FROM NON-HIGH-LEVEL WASTE CATEGORIES

Radionuclide	Dose: rem/a in 50th year
^{226}Ra[a]	3.5×10^{-7}
^{230}Th	1.0×10^{-7}
^{231}Pa	4.3×10^{-5}
^{234}U	5.3×10^{-8}
^{235}U	2.5×10^{-9}
^{238}U	8.8×10^{-8}
$^{230}Th/^{226}Ra$[b]	7.7×10^{-6}
$^{234}U/^{226}Ra$[c]	2.0×10^{-7}
Maximum annual total dose	5.1×10^{-5}
Time of maximum (years)	4.1×10^{8}

[a] Refers to ^{226}Ra which reaches the biosphere directly from the groundwater.
[b] Refers to ^{226}Ra produced by radioactive decay of ^{230}Th in the biosphere.
[c] Refers to ^{226}Ra produced by radioactive decay of ^{234}U (via ^{230}Th) in the biosphere.

The dose results shown in Tables XV and XVI are for a repository containing waste from a 100 GW(e)/a nuclear economy. To obtain the dose to the most exposed individual for any other sized repository up to the maximum of about four years of waste from a 100 GW(e) economy, the dose numbers must be multiplied by the factor:

$$\text{Factor} = \frac{\text{Economy size} \cdot \text{Years of operation}}{100} \tag{9}$$

Where economy size is in units of GW(e)/a.

For example, for a repository holding 20 years of waste from a 4 GW(e) economy, the dose numbers must be multiplied by 0.8.

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Appendix D

SWEDISH HARD CRYSTALLINE ROCK REPOSITORY

P.E. Ahlström

1. INTRODUCTION AND BACKGROUND

In 1977 the Swedish Parliament passed a Nuclear Stipulation Law which requires that before loading fuel and the operation of any new nuclear power reactor in Sweden the reactor operator shall, among other things, show how and where high-level waste from reprocessing (or spent unreprocessed nuclear fuel) can be finally disposed of in an "absolutely safe" way. The Swedish nuclear power industry responded to the proposed bill by organizing the Nuclear Fuel Safety Project (KBS).

The KBS project investigated potential disposal concepts for both the alternatives (high-level waste from reprocessing and spent unreprocessed fuel) which were mentioned in the Stipulation Law. A report of the handling and final storage of high-level vitrified waste was completed in December 1977 [1]. Later on, this report was supplemented by additional geological investigations [2]. Based on these reports and an extensive review by several Swedish and foreign organizations and individuals, the Swedish Government approved the fuel loading and startup of additional nuclear power reactors in June 1979 and in April 1980. The KBS project also completed and published a study on final storage of unreprocessed spent fuel [3].

This Appendix describes some of the methods and data used for the safety analyses included in the first KBS reports [1]. These methods reflect the status by mid-1977 when the analyses were made. Some additional information and data have been included for completeness and in such case the proper references are given. No attempt has been made to update the models and results in order to reflect the extensive development work which has been performed by KBS and others since 1977.

The Swedish Stipulation Law uses the expression "absolutely safe". In the strictest meaning of the word, no human activity can be considered absolutely safe. The fact that such an interpretation of the wording of the Law was not intended is evident from the formulation of the statements made by the Government in support of the Law, indicating that the storage of waste shall fulfil *"the requirements imposed from a radiation protection point of view and which are intended to provide protection against radiation damage"*. Questions regarding protection against radiation damage are regulated by the Swedish Radiation Protection Act.

This interpretation is supported by the statements made by the Committee of Commerce and Industry in its review of the Law, in which the Parliament also concurred. The Committee thus finds the expression "absolutely safe" to be warranted in view of the very high level of safety required, but considers that a *"purely Draconian interpretation of the safety requirement"* is not intended. ('Draconian' in the purest sense of being excessively severe, inhuman.)

The safety analyses of the KBS studies were carried out with the requirements and the interpretation of the law in mind. No effort was made to do the kind of analyses which would be required for licensing a repository at a specific site.

2. DESCRIPTION OF HOST ROCK SITE AND REPOSITORY

2.1. Host rock characteristics

The geological investigations carried out by the Geological Survey of Sweden on behalf of KBS aimed to establish whether the Swedish Precambrian bed-rock can be used for a waste repository [1, 2]. Field investigations have been performed at five sites, three of which have been selected for further study. The studied areas contain the most common types of rock, namely granite, gneiss and gneiss-granite. The descriptions of the properties of these rocks, given in Appendix C, are applicable from a generic standpoint. Some of the site-specific information found during the Swedish studies is summarized below. This site-specific information is given to enable one to get an idea of the type of studies needed and the data that can be generated from them.

Of the areas studied, the *Karlshamn area* is geologically the best known. The bed-rock in the area is a grey gneiss which contains few fractures and little groundwater. The fracture systems alternate, and the fractures exhibit no pronounced main directions. Core drillings, which were carried out within the area to a depth of 790 m, also show unchanged good conditions at this depth. This is because the gneiss has been a rigid and highly resistant body ever since its folding more than 1300 million years ago. Displacements along fracture zones have been small for a very long period, approximately 0.02 mm per million years. On the basis of obtained data, the groundwater flow at a depth of 500 m can be calculated to be considerably less than 0.2 L/m^2 annually (permeability of the order of 5×10^{-12} m/s on the average below 300 m level).

Geological and geophysical surveys have been conducted in the *Finnsjö area* which is characterized by a very common type of crystalline Precambrian rock in Sweden. The area is composed of primary granite, which is a uniform, weakly gneissified granite. The central parts of the area are characterized by large blocks of little-fractured bed-rock interspersed by fracture zones. On the basis of measured values for water permeability, the groundwater flow at a depth of 500 m can be calculated to be approximately 0.1 L/m^2 annually, although larger flows occur in some fracture zones. The path of the groundwater through the bed-rock has also been studied. In general, the groundwater flows down into the rock in elevated areas and then up towards the surface in valleys. The influence of the topography on the groundwater flow often extends down to a depth of several thousand metres.

The *Kråkemåla area* has been the third site for geological and geophysical surveys where three core bore-holes have been studied. The area is composed of a very uniform, undeformed granite. The groundwater flow in the less pervious sections has been calculated to be about 0.15 L/m^2 annually. Considerably larger flows are found in the crush zone within the area.

Several studies of groundwater composition and movement have been conducted and the results carefully evaluated. Groundwater datings show, for example, that the transit time of the groundwater to the surface of the earth from a rock repository in an inflow area can amount to several thousand years. Changes around a rock repository caused by the blasting work and by the waste heat generated by the waste are local. The risk that new flow paths for the groundwater will be created by such changes is negligible. The studies at Karlshamn, Finnsjö and Kråkemåla show that all three areas possess the fundamental prerequisites for a safe rock repository for high-level waste, providing that the repository is designed with consideration for local conditions.

2.2. Repository design and data

The repository size is based on disposal of only HLW from thirty years' operation of thirteen LWRs, equivalent to about 300 GW(e) · a without plutonium recycle. The area of the filled repository is about 1 km^2.

The HLW is vitrified as borosilicate glass and enclosed in stainless-steel canisters (400 mm i.d. and 1500 mm long); the HLW canisters are encapsulated in canisters of titanium (6 mm thick) with annular spaces of 100 mm thickness filled with lead. It was assumed that about 9000 canisters would be disposed of in the repository. The canisters are emplaced in the repository with a backfill, nominally 85% quartz sand and 15% bentonite.

The repository consists of a system of tunnels at 500 m depth. The tunnels are 3.5 m wide and high and spaced at about 25 m. The waste is emplaced in 1-m-diameter and 5-m-deep holes which are drilled at 4 m spacing from the tunnel floors.

The inventory of radionuclides was calculated using the ORIGEN code [4]. Data for PWR fuel with 33 000 MW · d/t U were used when performing the calculations. Later calculations using more detailed reactor physics codes [5] have shown that the ORIGEN code used probably underestimated the amount of heavy nuclide formed.[1]

The inventories calculated do, however, conservatively represent the mix of 75% BWR fuel (27 600 MW · d/t U) and 25% PWR fuel (33 000 MW · d/t U) which are the basis for the analyses (see Vol.II, Section 8.2 of Ref. [3]).

3. SCENARIO SELECTIONS

The radioactive waste is isolated from the biosphere by a number of barriers. The degradation of these barriers by both slow natural processes and by extreme and sudden events was evaluated. The release scenario of primary significance is that in which radionuclides are leached from the waste, after the waste canisters have failed, and transported from the repository by small amounts of groundwater normally present in crystalline rocks at depth.

The activity release caused by natural processes is determined by the minimum canister service life of 1000 years, a glass leaching period of 30 000 years and a nuclide specific retention in the geosphere transport based on assumed oxidizing conditions of the groundwaters all the way down to the repository. The parameters used in the analyses were selected to be conservative, that is to yield consequences to the biosphere more unfavourable than would be expected. The intention was to calculate an upper limit of the consequences to show that it is possible to provide a safe disposal of the high-level waste according to requirements in the Stipulation Law (see Section 1).

The quantitative discussion on extreme events was concentrated on bed-rock movements. Owing to the stability of the Fennoscandian shield such occurrences have very low probabilities. The consequences of various rock displacements through the repository were analysed.

The effects caused by acts of war, sabotage and future disturbance by man were discussed in a qualitative way.

4. REPOSITORY ANALYSES

As a reference case for safety analysis, a titanium/lead/stainless-steel canister system was devised for the glass waste form. This combination was estimated to withstand 1000 years in the repository without loss of integrity [6]. Earlier penetration of canisters (e.g. 100 and 500 years) was also considered. In a variation analysis, the consequences of one canister having a damaged seal at

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[1] The ORIGEN code has since been modified – see Ref. [31] of Chapter 7 of this Safety Series.

the time of being emplaced was calculated. With the waste form as one barrier, the canisters are another effective barrier during this period when the total radioactivity level is lowered through decay by three orders of magnitude. The retardation potential of clay in cracks is a third barrier. Radiolytic effects on the groundwater will be reduced to insignificant levels by the shielding offered by the lead.

In the reference scenario, a basic leach rate for vitrified waste of 2×10^{-7} g/cm^2 of surface area per day at 25°C was chosen, based on laboratory experiments with short-term interval leachant replacements. This leach rate was corrected for the effect of temperature, being for example 10-fold higher at 70°C. The surface area assumed for leaching was assumed to be five-fold higher than the geometrical surfaces of the waste forms to allow for cracks in the glass; this factor is about twice that indicated by experimental measurements. Based on experimental evidence and analysis of the groundwater situation, it was concluded that the pH of the groundwater would stabilize at a value between 8 and 9. The effect of pH on the leach rate could thus be neglected.

On the basis of these assumptions, the leach rate corresponds to a dissolution time of about 30 000 years after the canisters fail.

In reality, leaching of the waste glass will be controlled, because of the low solubility of silicic acid, by the supply of water which is limited owing to the low permeability of the host rock and the bentonite clay barrier. This subject is discussed in Appendix C, Section 4. Complete dissolution of the glass was estimated to take approximately 3 000 000 years when these factors were considered. Nevertheless, the shorter duration dissolution time was used for most of the safety analyses, although the effects of longer durations were evaluated.

5. GEOSPHERE ANALYSES

5.1. Hydrogeological studies and modelling

As described in Section 2.1 numerous geological and geophysical investigations have been carried out in three areas of Sweden, including measurements of groundwater flow. Groundwater datings (using the ^{14}C method) also showed that the transit times of groundwater to the surface of the earth from a repository in an inflow area can amount to several thousand years (e.g. in Kråkemåla from 4000 to 11 000 years). In addition, comprehensive theoretical studies have been carried out within the KBS project to shed light upon the flow pattern of the groundwater in rock at various depths. Thus, the repository safety analyses rest on experimental and theoretical data for actual potential sites.

To calculate the travel time of the groundwater, i.e. the time it takes for the water to migrate from a repository to a receiving body of water, such as a lake or a well, requires knowledge of the local pattern of groundwater flow as well as of the permeability and porosity properties of the rock. Although these data are not yet fully known, calculations were carried out for a number of reference cases employing hydrological models like those described previously and using various input data.

The choice of input data to the nuclide transport calculations was based on two-dimensional and axially symmetric one-dimensional models for the sites of Finnsjö [7] and Karlshamn [3, 8]. These models were used to calculate the groundwater transport time from the repository to the biosphere for a number of simplified cases. The results exhibit considerable variations in calculated groundwater travel times owing to the choice of input parameters.

Since it cannot be demonstrated with certainty at this time that the travel time of the groundwater from depths of around 500 m generally amounts to several thousand years, the very conservative value of 400 years was used in the consequence analysis. Both the theoretical calculations and the age determinations show that actual travel times from a suitably situated final repository are considerably longer.

5.2. Radionuclide transport models

5.2.1. Mathematical model

The model used for the nuclide migration calculations was the one-dimensional transport model, called GETOUT [9], adapted to Swedish conditions by the KBS project. This model is for a homogeneous medium and takes into account hydraulic convection and dispersion as well as chain decay and geochemical retardation for the various nuclides. GETOUT is based on analytical solutions of a set of first-order differential equations:

$$D \frac{\partial^2 N_i}{\partial Z^2} - V \frac{\partial N_i}{\partial Z} - K_i \frac{\partial N_i}{\partial t} - K_i \lambda_i N_i + K_{i-1} \lambda_{i-1} N_{i-1} = 0 \qquad (1)$$

where:

D = dispersion coefficient (m^2/s)
V = groundwater velocity (m/s)
K_i = retardation factor for nuclide i
λ_i = decay constant (s^{-1})

Z = distance of migration (m)
t = time (s)
N_i = discharge rate of nuclide i at Z and t (mol/s)

The leach rate is assumed to be constant. If the dispersion can be neglected, N_i is independent of specific values of V and Z. Instead the ratio Z/V, i.e. the groundwater transport time, will be the controlling parameter. Other parameters are

Time to canister failure
Dissolution time for glass or fuel
Retardation factors.

5.2.2. *Input parameters*

Input data for use in the GETOUT calculations were the radionuclide inventory in the repository, the release rates as determined by the repository analyses (Section 4) and the groundwater travel time as derived from the hydrogeological models (Section 5.1). In addition, geochemical data were derived from experimental data as described below.

Various chemical reactions are responsible for retention of nuclides in the geosphere, primarily ion-exchange processes, ion adsorption, reversible precipitation and mineralization. These processes are collectively referred to below by the term "sorption". Mineralization and precipitation are the most favourable processes from the viewpoint of safety, since they result in very low residual levels in the groundwater and thereby high retentions. It can be assumed on good grounds that many of the elements in the waste participate in mineralization and precipitation reactions, for example caesium (mineralization), protactinium and americium (precipitation as hydroxide compounds). Available experimental data indicate that a safety margin can be obtained with respect to retention by treating sorption as ion exchange. Consequently, in the safety analysis, all sorption is considered to be reversible processes.

Retention is expressed as a retardation factor, defined as the ratio of the groundwater velocity to the nuclide velocity. The size of the retardation factor is dependent on a chemical equilibrium constant and a quantity which characterizes the available amount of ion-exchange material.

Within the KBS project, Allard and Neretnieks [10, 11] carried out determinations of the equilibrium constants for the buffer material (10% bentonite clay and 90% quartz sand), granite and various zeolites. The data encompass 14 elements. Burkholder [9] has specified retardation factors for a large number of elements in a type of soil called Western US Desert Subsoil. Landström et al. [12] carried out in-situ measurements of retardation in rock fissures at Studsvik.

Retention in the buffer material immediately surrounding the waste canisters was neglected; Neretnieks [11] has shown that diffusion through 20 cm of buffer mass is relatively rapid. However, the buffer material is of vital importance in preventing radioactive elements which have been dissolved from the glass from dispersing via tunnels and shafts. Häggblom [13] has shown that diffusion over the distances in question is extremely low.

The retardation factor in rock can be written as follows:

$$K_i = 1 + K_a \cdot a_1 \tag{2}$$

K_i = retardation factor

K_a = surface-based equilibrium constant $\left(\dfrac{\text{Ci/m}^2\ \text{rock}}{\text{Ci/m}^3\ \text{solution}}\right)$

a_1 = accessible surface area for ion $\left(\dfrac{\text{m}^2\ \text{rock}}{\text{m}^3\ \text{solution}}\right)$

The surface-based equilibrium constant can be calculated from a mass-based equilibrium constant as follows:

$$K_a = \frac{K_d}{a_2} \tag{3}$$

K_d = mass-based equilibrium constant $\dfrac{\text{Ci/kg rock}}{\text{Ci/m}^3\ \text{solution}}$

a_2 = specific area for laboratory specimen $\dfrac{\text{m}^2\ \text{rock}}{\text{kg rock}}$

Allard [10] was able to determine the K_d values with reasonable accuracy. But there is a degree of uncertainty involved in the determination of the surface areas a_1 and a_2. If the particles in the crushed rock specimen used in Allard's measurement are regarded as solid spheres, a value of approximately 30 m^2/kg is obtained for a_2. A measurement according to the BET method (adsorption of nitrogen gas on the rock specimen) gave a specific surface area of 12 000 m^2/kg, however. The large difference between the measurement results and the calculated external surface area of the particles shows that a large portion of the area is in pores in the particles. If the pores are sufficiently large for the waste nuclides to enter, a_2 should be set at 12 000 m^2/kg in calculations of K_a. In this case, however, a_1 should be estimated on the basis of the assumption that the rock is porous, i.e. that the waste nuclide can diffuse into the rock from the fissures in which transport normally takes place.

In the calculations, the rock was assumed to be solid, i.e. the value of a_2 was set at 30 m^2/kg. The accessible surface area for ion exchange, a_1, was calculated as the geometric surface area of the fissures, assuming that the walls

TABLE XVII. RETARDATION FACTORS

Element	Oxidizing environment	Reducing environment with conservative concentration values and short contact time	Best estimate for reducing environment and slow groundwater transport
Ni	–	–	6 100
Sr	51	120	1 500
Zr	8 000	4 800	61 000
Tc	1	1	950
I	1	1	1
Cs	800	1 200	4 000
Ce	80 000	19 000	200 000
Nd	25 000	3 800	200 000
Eu	50 000	30 000	200 000
Ra	670	1 200	48 000
Th	5 100	1 900	46 000
Pa	37	37	11 400
U	41	1 900	23 000
Np	260	1 900	23 000
Pu	1 100	2 800	5 700
Am	80 000	19 000	610 000
Cm	40 000	9 500	305 000

of the fissures are flat and parallel. The retardation factors were calculated for three different fissure sizes corresponding to the permeability (k) span of 10^{-9} to 10^{-5} m/s. Subsequently, supplementary measurements indicated that the model assuming solid rock and plane-parallel fissure walls may underestimate the retardation factors by at least a factor of 10. Furthermore, Allard's early measurements [14] were performed under oxidizing conditions which entailed high valence states for the elements neptunium and plutonium. Owing to the presence of iron (II) in the type of rock selected for the repository, the chemical conditions will be reducing. This means that the elements mentioned will be present primarily in the form of tetravalent ions.

Allard's later experiments [10] showed that many of the retardation factors will be considerably larger in a reducing than in an oxidizing environment. The experiments also showed that the retardation increases with increasing contact

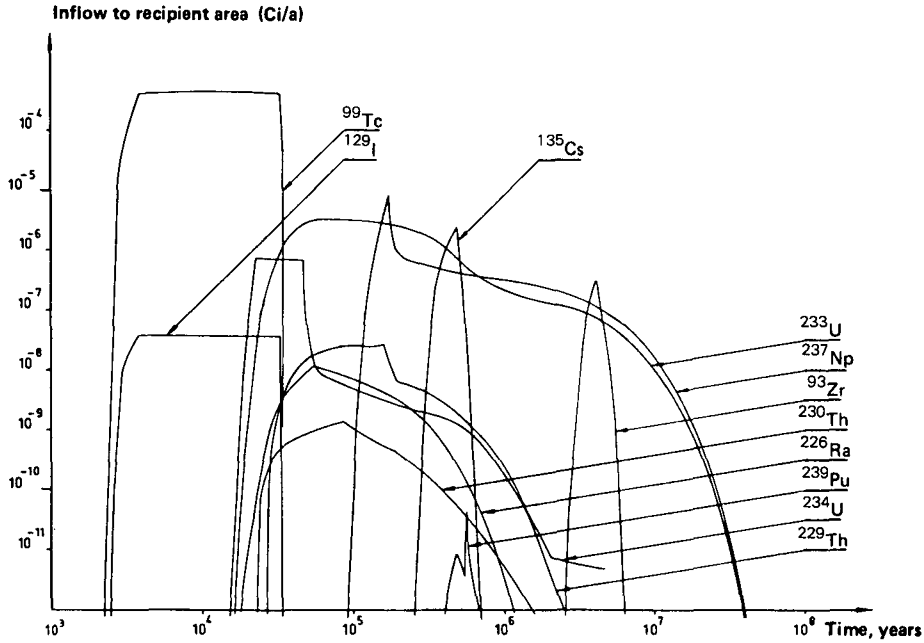

FIG.21. Example of calculation of inflow to recipient area at various points in time, carried out using GETOUT computer program.

time between the solution and the solid. The influence of these parameters are illustrated by Table XVII, which was derived from Allard's measured K_d and K_a values [15]. The values were calculated for a rock with a hydraulic conductivity of 10^{-9} m/s and an average fissure spacing of 1 m, assuming a surface reaction mechanism.

The nuclide transport calculations for the vitrified high-level waste study [1] were made using only the values in "Set a" of Table XVII. These values were the only data available at the time the calculations were made.

Other phenomena involved in the nuclide migration are the possibility of the formation of colloids and complexing with organic species in the clay. These phenomena were not fully explored but later studies [3, Vol. II, Section 7.2.5] have shown that such transport mechanisms are small compared with other transport mechanisms.

The dispersion is treated as an axial diffusion mechanism in GETOUT and will only give a slight effect of annual inflow. Neretnieks [11] has shown that the dispersion due to the occurrence of different crack widths and corresponding water flow is more important. The latter effect has been taken care of by manual corrections.

5.2.3. *Example of results from geosphere analyses*

Figure 21 is an example that illustrates the type of results obtained from calculations using GETOUT. The curves in the graph describe the inflow of important radionuclides to the primary recipient area as a function of time. The example is based on the following assumptions:

HLW from 300 GW(e) years;
1% iodine-129, 0.1% uranium and 0.5% plutonium loss to waste;
All canisters fail after 1000 years;
Dissolution time for the glass is 30 000 years;
Groundwater travel time is 400 years;
Retardation factors according to "Set a" in Table XVII.

Nuclides of little radiological importance have been omitted from the figure.

6. BIOSPHERE AND DOSIMETRY ANALYSES

6.1. Biosphere transport models

The interfaces between the geosphere and the biosphere are where the groundwater comes into contact with receiving bodies, such as a lake or a well; these are called "recipients" in the following text.

The ecosystems are described by a model system which encompasses interconnected areas or volumes, as illustrated by Fig. 22. Within and between these areas, radionuclides can be transferred to various reservoirs, known as "compartments", such as water, sediment, earth, biota and atmosphere. Backflows can also occur between the compartments in an interconnected system. The concentration of radionuclides within the compartments is assumed to be homogeneous. (Figure 22 is somewhat simplified and includes the separate reservoir systems for the local area, considered in the model applications, within the regional area.)

The models for the intermediate and global ecosystems apply to all main types of outflow from the geosphere to the biosphere. However, the models for the local and regional ecosystems vary, depending on whether the outflow from the geosphere takes place in an inland area, such as in a valley or under a lake, or to the Baltic Sea (Intermediate Sea). Details of these ecosystems are described below.

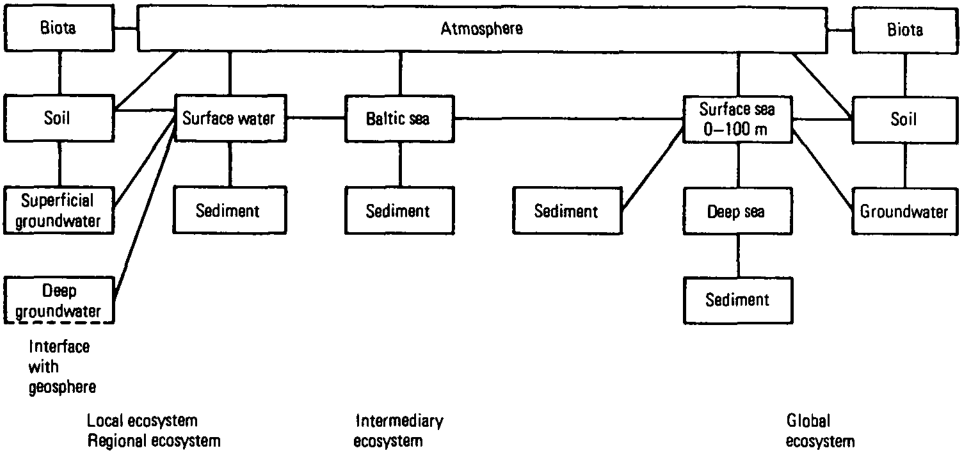

FIG.22. Reservoirs for the various ecosystems.

6.1.1. Local ecosystems

Three main alternatives for inflow of radionuclides to the biosphere were studied:

The well alternative: If the groundwater discharges into a valley, the wells and the nearest lake in the catchment area comprise the primary paths of inflow for the radionuclides into the biosphere. The inflow of radionuclides is divided equally between a valley and a nearby lake. Half the inflow was thus assumed to be diluted in the percolated rain-water (5×10^5 m^3/year) from a 2 km^2 area.

The lake alternative: If the groundwater flows out into a lake the lake is the path of inflow. The inflow is divided equally between a nearby lake and its downstream lake system. Dilution was assumed to occur in 2.5×10^7 m^3/year.

The Baltic Sea alternative: If the groundwater containing the radionuclide discharges into the Baltic Sea, a water area in the proximity of the coast is the primary recipient of activity. The inflow occurs into a coastal zone of the Baltic Sea with a volume of 1 km^3. This volume is exchanged 10 times per year.

The local ecosystem in the well and lake alternatives is assumed to consist of a 0.25 km^2 area of farm land. The area is regarded as a system of reservoirs for the radionuclides. The depth of the reservoirs for soil and superficial groundwater is 2 m.

In the Baltic Sea alternative, the local ecosystem consists of 1 km^3 sea-water and underlying sediment within a coastal belt 2 km wide and 30 km long.

6.1.2. *Regional ecosystems*

In the well and lake alternatives, the regional ecosystem is assumed to consist of a land area which is 30 X 30 km and 2 m deep. The groundwater in the system is the superficial groundwater down to a depth of 2 m underneath the land area. The surface water in the regional ecosystem consists of the volume of the lake. The ecosystem also includes the surface sediment layer at the bottom of the lake.

In the Baltic Sea alternative, the regional ecosystem is the same as the local ecosystem.

6.1.3. *Intermediate ecosystem*

The intermediate ecosystem consists of the Baltic Sea and its coastal region. The Baltic Sea reservoir comprises a water volume of 3.7×10^5 km^2 with an average depth of 60 m. The system also includes the sediment at the bottom of the Baltic Sea and the volume of air in the atmosphere up to an altitude of 1 km above the region and the Baltic Sea area.

6.1.4. *Global ecosystem*

The global ecosystem encompasses a number of different reservoirs.

The global atmosphere up to an altitude of 1 km.

The surface sea, which comprises the upper 100 m of the open sea. It mixes relatively rapidly, but has a relatively slow rate of exchange with the deep sea.

The deep sea, which consists of the sea-water below a depth of 100 m.

The sediments at the bottom of the sea. These include sediments on the bottom down to a depth of 150 m around the sea coasts (littoral sediments) and sediments on the deep sea bottom (abyssal sediments). The total surface area of the sediments is approx. 2×10^8 km^2. The littoral sediments comprise 4% of the total surface area.

Soil, which comprises an upper soil layer to a depth of 0.5 m.

Groundwater below the surface of the ground. This comprises 4×10^{18} kg water which transports the radionuclides to the surface seas and the upper soil layer.

The biomass of the global land area – the biota. This constitutes an important reservoir in the global ecosystem for some radionuclides with long half-lives, e.g. ^{14}C, ^{99}Tc, and ^{129}I.

6.1.5. *Transfer of radionuclides*

Within these regional and intermediate ecosystems there is a turnover of radionuclides in relation to the movements of the air and water masses which transport the activity. The nuclides are then transferred to the land area through irrigation from the lake or via the atmosphere by precipitation which entrains particles which come from the global land areas. The nuclides are then recirculated in various natural cycles until they reach the superficial groundwater or run off into the lake again via the groundwater and surface water.

The lake and its sediments exchange nuclides by means of sedimentation, resuspension and dissolution. Surface water runs via the lake to the Baltic Sea, where there is an exchange between water and sediment. The Baltic Sea is connected via Öresund Sound and The Belts with the oceans in the global ecosystem. By means of mechanisms such as evaporation, precipitation and foaming the radioactivity can be exchanged between the air and water in the Baltic Sea as well as the global area.

The global ecosystem is connected with the regional ecosystem by exchanges via air and water in the Baltic Sea area. The global system of carriers is basically identical to the system for the regional and intermediate areas.

The transfer of radionuclides from one reservoir (compartment) to another can be calculated with the aid of coefficients of transfer. These have been determined by a review of the results from many different studies [16], mainly of the distribution of the various nuclides between the reservoirs, nuclide balances, the migration of the radionuclides from the atmospheric testing of nuclear weapons, the escape of nuclides from leaking storage facilities and a number of laboratory experiments with ecosystems on land and in water. Data on the turnover of air and water in the Baltic Sea area and in the global system are available from meteorological and hydrological studies.

With the aid of the mathematical model it is possible to calculate the concentration of various radionuclides in the reservoirs when the inflow of radioactivity to the primary recipient and the coefficients of transfer between the reservoirs are known.

TABLE XVIII. EXPOSURE PATHWAYS IN THE LOCAL ECOSYSTEM AND IMPORTANT NUCLIDES

Exposure pathways	Primary recipient[a]	Some important nuclides
Internal exposure:		
Inhalation	W, L	–
Soil – grain	W, L	–
Soil – green vegetables	W, L	^{237}Np, ^{229}Th, ^{231}Pa
Soil – root vegetables	W, L	–
Grass – milk	W, L	^{99}Tc, ^{129}I, ^{226}Ra
Grass – eggs	W, L	^{129}I, U(all), ^{14}C
Grain – eggs	W, L	–
Drinking water	W, L	^{237}Np, ^{226}Ra, U(all), Th(all), ^{242}Pu, ^{231}Pa
Water – fish (fresh and salt water fish)	W, L, B	^{135}Cs, ^{226}Ra, U(all) ^{14}Cs
External exposure:		
Ground contamination	W, L	–
Beach activities	L, B	^{226}Ra, ^{229}Th
Swimming	L, B	–
Fishing tackle	L, B	^{226}Ra, ^{229}Th

[a] W = well; L = lake; B = Baltic Sea.

6.2. Exposure pathways

When the radionuclides have arrived at the reservoirs in the biosphere, they can reach man in basically two different ways. They can be ingested into the body either through food and water or through inhalation. As long as they remain in the body they cause internal irradiation. Knowledge concerning the transport and enrichment of the radionuclides in the food chains is therefore of great importance for being able to calculate the dose to man. Human beings can also be irradiated by radionuclides outside the body – "external irradiation".

The different pathways accounted for in the dosimetry analyses are listed in Table XVIII. In the table W = well; L = lake; and B = Baltic Sea. The table also lists some important nuclides for the pathways.

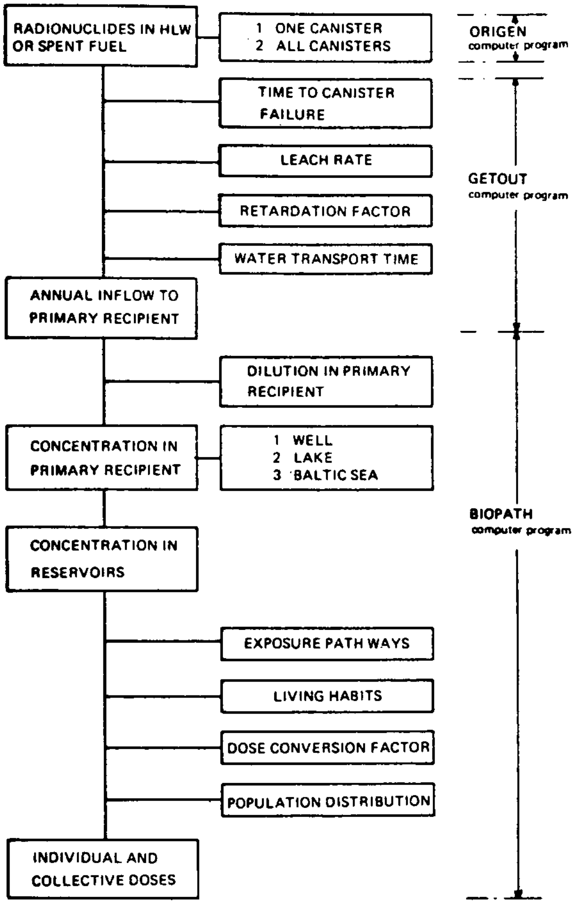

FIG.23. Scheme of consequence calculation.

6.3. Dosimetry models

The principle of the dose calculations starting with the source of the radionuclides and ending with radiation doses or consequences is shown in Fig. 23. The use of the ORIGEN and GETOUT computer programs was introduced previously. The BIOPATH computer program [16], briefly introduced here, was developed at Studsvik, Sweden, for the calculations of individual and collective doses arising from releases of radionuclides into the biosphere. The mathematical treatment of ecological cycling is based on compartment theory, illustrated in Fig. 22. The transport of nuclides between these compartments or

TABLE XIX. DOSE CONVERSION FACTORS FOR INTAKE WITH FOOD AND WATER OR THROUGH INHALATION OF 1 Ci IMPORTANT NUCLIDES

Nuclide	Weighted whole-body dose commitment (rem/Ci)	
	Intake by food and water	Inhalation
^{14}C	9.9×10^{2}	6.6×10^{2}
^{90}Sr	1.5×10^{6}	2.3×10^{6}
^{93}Zr	1.7×10^{2}	1.8×10^{4}
^{99}Tc	5.5×10^{2}	3.6×10^{2}
^{129}I	3.4×10^{5}	1.9×10^{5}
^{135}Cs	7.3×10^{3}	5.7×10^{3}
^{137}Cs	5.5×10^{4}	3.8×10^{4}
^{226}Ra	2.8×10^{6}	3.8×10^{6}
^{229}Th	1.8×10^{6}	4.9×10^{9}
^{230}Th	3.4×10^{5}	9.0×10^{8}
^{231}Pa	6.6×10^{5}	2.4×10^{9}
^{233}U	1.1×10^{5}	2.7×10^{6}
^{234}U	1.1×10^{5}	2.7×10^{6}
^{235}U	1.1×10^{5}	2.7×10^{6}
^{236}U	1.1×10^{5}	2.7×10^{6}
^{238}U	1.1×10^{5}	2.7×10^{6}
^{237}Np	2.0×10^{5}	5.0×10^{8}
^{239}Pu	1.6×10^{5}	9.5×10^{8}
^{240}Pu	1.6×10^{5}	9.5×10^{8}
^{242}Pu	1.6×10^{5}	9.5×10^{8}
^{241}Am	2.2×10^{5}	4.1×10^{8}
^{243}Am	2.2×10^{5}	4.1×10^{8}

reservoirs is described by a set of first-order differential equations with constant coefficients. The mathematical analysis also includes products in decay chains, i.e. daughter nuclides generated by decay of nuclides during ecological cycling. The equations are written as follows:

For the parent nuclide

$$\dot{Y}_M(t) = K_M Y_M(t) + Q_M(t) - \lambda_M Y_M(t) \tag{4}$$

For the daughter

$$\dot{Y}_D(t) = K_D Y_D(t) + \lambda_D Y_M(t) - \lambda_D Y_D(t) \tag{5}$$

where,

Y = amount of activity in compartment at time t
$\dot{Y}$ = change of activity per unit time
K = transfer coefficient
Q = source strength within the compartment
λ = decay constant

The further dispersion and turnover of the nuclides take place in relation to the movement of certain carriers in different media. Uptake in food is described by use of concentration and distribution factors. The exposure pathways which have been considered here are those which experience has shown to cover the most significant possibilities.

The individual radiation doses calculated are weighted whole-body annual dose rates as a function of time with weight factors according to ICRP 26 [17]. The collective doses are weighted whole-body annual global collective dose rates. The dose conversion factors used are given in Table XIX. It should be pointed out that some of the dose conversion factors given in Table XIX have been changed considerably in the recent publication, ICRP 30 [18], e.g. for the actinide elements in particular.

6.4. Results from biosphere and dosimetry analyses

As stated previously, apart from the extremely unlikely events such as a large meteorite hitting the repository area or a volcanic explosion, transport of radionuclides from the repository to the biosphere can only occur by groundwater flow. Initiating events or processes of release may be:

Initial failure of one or a few canisters;
Long-term degradation of the canisters as a result of corrosion,
Breakage of canisters due to substantial rock displacement as a result of faulting.

In the initial stages of the safety analysis, efforts were made to cover the two-dimensional risk spectrum by treating consequences *and* probabilities. Lack of probability data and time made it necessary to concentrate on consequences of the most important release scenarios, while keeping the axis of probability in mind. Thus, for example, the many possible modes of canister failures were treated in a simple but realistic way, by calculating the consequences of an initial failure of one canister at the time of emplacement as one main case in parallel with the case of a failure of all canisters during a certain time interval as a result of long-term degradation. Other cases, e.g. initial failures of several canisters, could easily be evaluated by comparison.

About 160 runs were made with GETOUT, covering about 30 nuclides. The output data from GETOUT, illustrated by the example in Fig.21, are the annual inflows of activity to a primary recipient as a function of time. These data are used as input to BIOPATH, where the three relevant types of primary recipient are also defined by a certain volume for dilution. The concentrations in the primary recipients are calculated as a first step. About 60 of the GETOUT runs were followed by BIOPATH runs, each one treating only one single nuclide. Thus, with an emphasis on the 5–15 most important nuclides, about 70 runs with BIOPATH were made [19].

The consequences of releases of radionuclides from the waste canisters into the groundwater, calculated during the above BIOPATH runs, are summarized below. Consequences for the reference scenario (i.e. canister life: 1000 years; glass dissolution time: 30 000 years; groundwater travel time: 400 years; and retardation factors, "set a" of Table XVII are presented first, and the effects of variations from the reference parameters are discussed subsequently.

To limit the scope and content of this document, there is no detailed analysis or discussion of the evidence for the choice of certain values or range of variation of parameter data. These can be found in various KBS reports [1, 19].

6.4.1. *Reference scenario*

Since calculations for the reference scenario were based on the very conservative assumptions described previously, they reflect upper limits for the consequences of radionuclide releases to the biosphere. The maximum radiation dose rates to individuals in the critical group for the reference scenario are shown in Table XX.

The dose to individuals far in the future, who may use water from a nearby well, will remain at approximately 10 mrem/a and will relate to only a small group of people. As can be seen, the predominant nuclides for the well case are ^{237}Np, ^{99}Tc, ^{226}Ra, ^{233}U and ^{135}Cs. The use of water from the lake will limit the maximum individual dose rate to 1 mrem/a in the reference scenario. If the inflow goes to the Baltic the doses will be considerably lower.

TABLE XX. MAXIMUM ANNUAL DOSE RATES TO INDIVIDUALS

Nuclide	Maximum inflow to recipient		Maximum dose rate (rem/a)		
	Time (years)	Activity (Ci/a)	Well	Lake	Baltic
^{93}Zr	4×10^{6}	3×10^{-3}	2×10^{-7}	2×10^{-7}	2×10^{-9}
^{99}Tc	6×10^{3}	5×10^{0}	2×10^{-3}	9×10^{-5}	7×10^{-7}
^{129}I	6×10^{3}	1×10^{-4}	7×10^{-5}	3×10^{-6}	2×10^{-8}
^{135}Cs	4×10^{5}	2×10^{-2}	4×10^{-4}	3×10^{-4}	1×10^{-6}
^{226}Ra	5×10^{4}	1×10^{-4}	6×10^{-4}	4×10^{-4}	1×10^{-6}
^{229}Th	9×10^{4}	3×10^{-4}	6×10^{-4}	6×10^{-4}	2×10^{-6}
^{230}Th	5×10^{4}	1×10^{-5}	2×10^{-6}	3×10^{-7}	2×10^{-9}
^{233}U	5×10^{4}	3×10^{-2}	2×10^{-3}	9×10^{-5}	7×10^{-7}
^{234}U	3×10^{4}	7×10^{-3}	3×10^{-4}	2×10^{-5}	2×10^{-7}
^{237}Np	2×10^{5}	9×10^{-2}	9×10^{-3}	4×10^{-4}	3×10^{-6}
^{239}Pu	6×10^{5}	5×10^{-7}	4×10^{-8}	4×10^{-8}	7×10^{-12}
Maximum total dose rate			1×10^{-2}	1×10^{-3}	5×10^{-6}
Time to maximum total dose				200 000 years	

6.4.2. *Effect of the content of radionuclides*

With other conditions constant, all calculation results for a given nuclide are proportional to the level of the nuclide in the waste. This level is affected by two factors which are important in this connection, namely:

(i) Degree of separation of uranium and plutonium as well as certain fission products in reprocessing; and
(ii) Time from discharge of fuel from reactor to reprocessing.

It has been assumed in the calculations that 0.1% uranium, 0.5% plutonium and 1% ^{129}I end up in the high-level waste. Some consider that the value for uranium is probably slightly too low, but this is of little importance for the results. It is estimated that an increase to 0.5% uranium would increase the level of (and the dose from) ^{234}U and ^{226}Ra by about 30%. The value of 0.5% chosen here for the plutonium content is above the expected or actual values for fuel cycles without plutonium recycle. Radionuclide inventory values for a simple plutonium recycle scheme or more complicated uranium and plutonium recycling schemes at equilibrium are not greatly changed apart from ^{240}Pu and ^{243}Am with its daughter ^{229}Pu. Because of the high retardation factors for these nuclides, these changes are of little importance.

It was assumed that reprocessing takes place 10 years after discharge of the fuel from the reactor. Of the important nuclides, the level of ^{237}Np (which is formed by neutron absorption and by the decay of ^{241}Pu, with a half-life of 14.6 years) is affected most. If reprocessing takes place after three years, the level of (and the dose from) ^{237}Np decreases by about 20%, whereas if reprocessing takes place after many more years, the level of ^{237}Np increases by a maximum of 75%.

6.4.3. Effect of the time for canister degradation

As a reference case for safety evaluation the titanium/lead canning of the glass cylinders was assumed to withstand 1000 years in the repository without loss of integrity. The case with one initially damaged canister was also analysed. The results showed that the doses due to this canister would be about 2×10^{-4} of the doses obtained for the reference case.

The consequences of a few initially damaged canisters will at worst be proportional to their number. In general, however, the local consequences will not depend upon the number of damaged canisters owing to the large size of the repository and the randomness of the damage.

Analyses of canister degradation in 100 and 500 years was also considered and found not to influence the dose results except for the slight obvious changes in the timing of the initial releases.

The extremely unlikely case with a combination of a disruptive displacement due to faulting and enhanced water flow is treated in Section 6.5.

6.4.4. Effect of waste glass leaching rate

In the reference scenario the leach rate for vitrified waste was chosen to be 2×10^{-7} $g \cdot cm^{-2} \cdot d^{-1}$, as discussed in Section 4 with short interval leachant replacements. This leach rate corresponds to a dissolution time of about 30 000 years for a glass surface area enlargement by a factor of ten compared with the nominal surface area of the glass cylinder. For the analysis of initial canister failure due consideration was taken of the temperature effect on the leach rate.

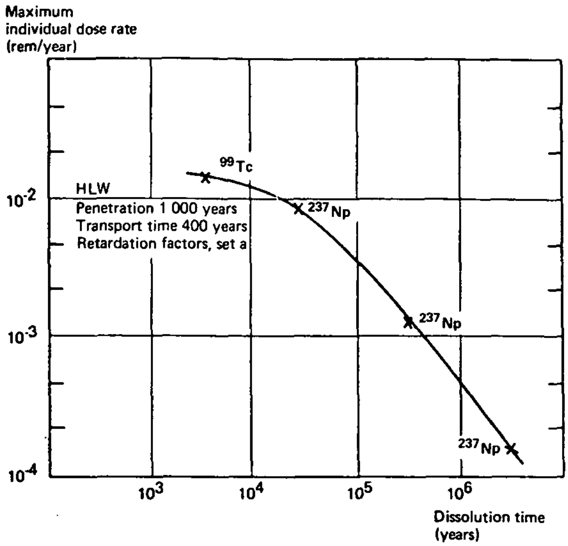

FIG.24. Maximum dose rates for different dissolution times for the glass cylinders. The crosses show total calculated dose rate and the nuclides are those which are predominant.

The water flow limitation due to the low permeability of the host rock and the bentonite clay barrier is expected to decrease the actual leach rates to values orders of magnitude lower than was assumed for the conservative reference scenario.

The leach rate will influence the doses, but dispersion and chain decay in combination with different retardation factors for parent and daughter nuclides will restrict a direct proportionality. Figure 24 shows the maximum individual doses for different leach durations for the well case.

As can be seen by extrapolation, shorter dissolution times than the reference 30 000 years do not increase dose levels in proportion owing to the dispersion effect. There is also a change of dominant nuclide.

6.4.5. *Effect of groundwater travel time and retardation factors*

Among the parameters entering the migration calculations, the groundwater travel time and the retardation factors are of key importance. As these two parameters govern the nuclide transport time, they also govern the extent to which the nuclides decay before appearing in the recipient.

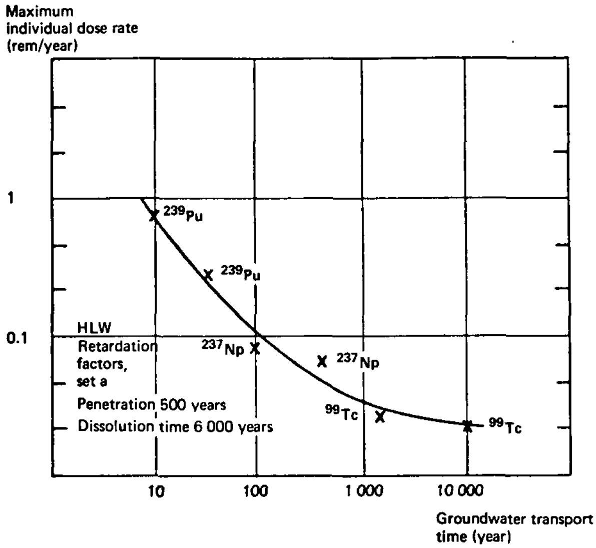

FIG.25. Maximum individual dose rates for different water transport times in the HLW scenario. The crosses show total dose rate and the nuclides are those which are predominant.

As discussed previously, the water travel time from the sealed repository to a primary recipient is dependent on rock properties and the hydraulic gradient. Four hundred years was first selected as a conservative value for the reference travel time for the whole repository. The effect of different water travel times on individual doses has been evaluated for 10, 40, 100, 400, 2000 and 10 000 years. The dose rates for the well alternative are given in Fig. 25 [19].

As expected, dose rates increase with shorter water travel times. There are also changes of dominant nuclides. It should be noted that only 6000 years was assumed for the leach duration data in Fig. 25. If the reference 30 000 years were used, the doses would be lower.

As discussed previously, the general water travel time from the main body of canisters disposed in high-quality rock will be in the range of thousands of years. Water from single canisters may, however, experience shorter travel times, but this means a considerable decrease in source strength and thus in doses. A combination of even 100 canisters with the extreme 10-year travel time yields a dose rate of 0.007 rem/a, which is lower than that of the reference scenario.

6.4.6. *Collective doses*

Annual as well as accumulated collective doses were calculated for the groundwater-borne radioactive material reaching the biosphere. The collective doses are the same for the two inland alternatives (well and lake as recipient) since the critical groups are small and the regional areas are the same. When the recipient is the Baltic, the collective doses are somewhat smaller owing to smaller doses to the region caused by the fact that the water of the Baltic is brackish and cannot be used for consumption or irrigation.

Summed over the worst 500 years, the collective dose via inland recipients has been calculated to 0.007 man·rem per MW(e)·a energy production. The corresponding number when the Baltic is the recipient is 0.006 man·rem per MW(e)·a.

The maximum global collective dose rate for the repository studied by KBS (waste from the production of 300 000 MW(e)·a of energy) will be around 4 man·rem/a and will not be reached until some hundred thousand years after the disposal.

The buildup of the accumulated global collective dose was later calculated by the Swedish Radiation Protection Institute and given as man-rem per unit produced energy [20]:

After	10^3 years	–	0.00	man·rem per MW(e)·a
	10^4 "	–	0.23	"
	10^5 "	–	0.47	"
	10^6 "	–	27	"
to infinity			300	"

6.5. Influence of extreme conditions

The probabilities of many phenomena (e.g. volcanic activity, displacements due to faulting and glaciation, and future drilling into the repository) are reduced to negligible levels when appropriate sites and repository designs are selected. Nevertheless, faulting was analysed in much detail in the KBS studies, as well as other unlikely and extreme events, to determine their potential for causing significantly increased releases of radionuclides and resulting doses.

The analyses of the geological stability of the Fennoscandian rock formation and the frequencies of earthquakes and displacements along fault planes are documented in several KBS technical reports and summarized in Ref. [1]. The most promising way to analyse the probability of displacements is by studying the frequency of displacements in bare rock walls. It has been estimated under certain assumptions for one formation that one canister in every 28 million years would be hit by a fracture movement in excess of 3 cm. For the time being it cannot be excluded that such movement will impair canister integrity and also increase the permeability locally. The leach duration of the waste glass after such an event is expected to remain at the 30 000-year level. The increased local permeability might lower the groundwater travel time but, based on the data shown in Fig. 25, the effect on both the individual and collective dose rates is expected to be small.

Future changes of landscape such as the drying up of a nearby lake or parts of the Baltic Sea may give rise to special exposure pathways, owing to the fact that the sediments may be used in agriculture. The consequences were analysed qualitatively. No increase of dose rates will occur for a lake drying up because the uptake via agricultural products grown on the sediment does not result in such high doses as fish consumption from the lake. Drying up of the Baltic may increase the individual dose for this inflow alternative by a factor of ten, owing to exposure from ^{135}Cs in agricultural products. The total individual dose will still be much lower than for the well and lake alternatives. The total collective dose rates in this case may rise by a factor of two.

An extended use of marine organisms other than fish, such as krill and macroalgae, will not increase the total global collective dose rates significantly. A replacement of 10 kg fish meat by 10 kg krill or algae will raise the collective dose rates 10% owing to the contribution from ^{242}Pu.

7. SUMMARY OF RESULTS

The factors influencing future individual dose rates arising from a repository in crystalline rock containing HLW corresponding to 3×10^5 MW(e)·a were analysed in some detail. Dose rates of 10 mrem/a or much lower were estimated for the most exposed critical group very far in the future. Even extreme values of parameter data do not yield alarming dose rates. For the severest identified case – a deep drinking water well in the vicinity of the final repository – it was calculated that the individual dose in the future could increase by a maximum of 0.4 rem over a 30-year period, which would be reached after around 200 000 years. To put these individual dose rates in perspective, Fig. 26 is helpful.

Figure 26 also presents the range of variation for the radiation doses which can be obtained from ^{226}Ra in drinking water in Sweden. These radiation doses have been calculated using the same dose factor which is used elsewhere in the study.

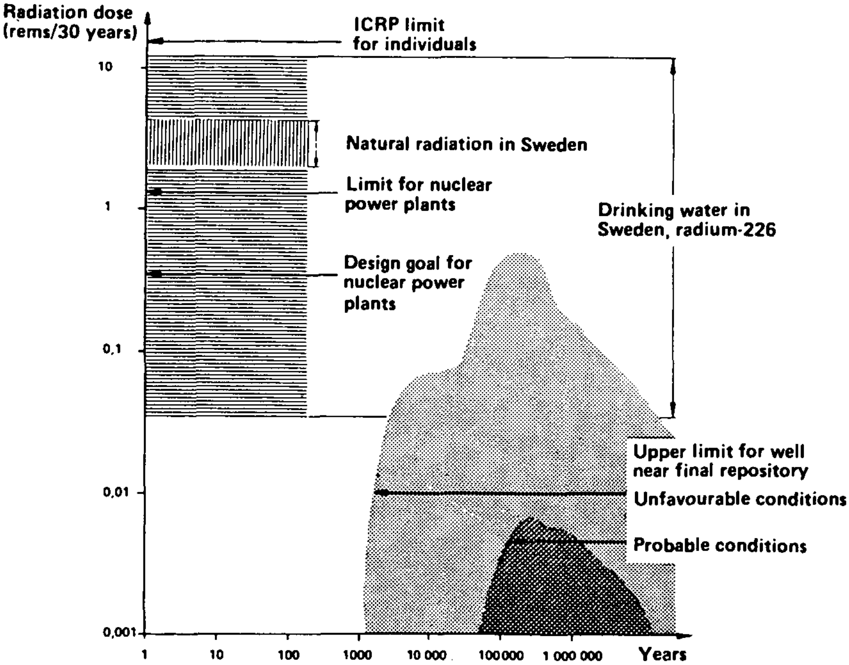

FIG.26. Calculated upper limit for radiation doses to people who live near the final repository (critical group). The calculations pertain to the slow decomposition of the canister with a well as the primary recipient. For purposes of comparison, the dose load from several natural radiation sources, as well as a number of established dose limits, have also been plotted in the diagram.

The calculated maximum dose rates are much lower than the maximum permissible radiation dose recommended by the Swedish authorities for persons living near nuclear energy installations. The increment in individual doses is less than fluctuations in the natural radiation level. In the most unfavourable case, the dose rate would be approximately equal to the target value recommended by the Swedish Institute of Radiation Protection as the goal which should be aimed at in the design of nuclear power plants.

The regional and global dose rates to large population groups were calculated for the most unfavourable 500-year period in the future. In the very long run, a maximum 500-year collective dose of less than 0.01 man·rem per MW(e) and year of operation is expected.

TABLE XXI. LEVELS OF RADIOACTIVE ELEMENTS IN WATER

Radioactive element	Levels in natural water in Sweden (pCi/L)		Maximum calculated increase in level in primary recipients near the final repository[a] (pCi/L)	
	Drinking water	Sea-water[b]	Well	Lake
^{226}Ra	0.1–40	0.3	0.1	0.002
Uranium	0.1–1500[c]	3	30	0.6
^{237}Np	–	–	90	2
^{40}K[d]	ca 20	330	–	–
^{135}Cs[d]	–	–	25	0.5

[a] Expected maximum values are less by a factor of about 100.
[b] With 3.5% salinity.
[c] Applies to natural water (not necessarily drinking water).
[d] ^{40}K and ^{135}Cs are biologically comparable, but have slightly different dose factors (24 000 as compared to 7300 rem/Ci, respectively).

The calculated increase in the level of radionuclides in the recipients to which the waste materials could possibly be dispersed are comparable to the natural levels of such elements. Neptunium-237 can be compared with uranium and caesium with potassium. Table XXI presents the ranges of variation for the level of certain elements in natural water and the levels which have been calculated for the various primary recipients in the least favourable case.

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Appendix E

CANADIAN SHIELD CRYSTALLINE ROCK REPOSITORY

R.B. Lyon

1. INTRODUCTION

The Canadian concept for nuclear fuel waste disposal is to immobilize the fuel waste, that is to render it stable chemically and mechanically, and to emplace it deep underground in a stable geological formation [1–4]. Pending a decision in Canada on fuel recycling, immobilization technologies are being developed for two options: disposal of irradiated fuel and disposal of the separated wastes that would result from reprocessing Candu fuel. The assessment discussed in this paper considers the disposal of intact fuel bundles in crystalline rock formations, known as plutons, located throughout the Canadian Shield.

The current programme for nuclear fuel waste disposal is in a "Concept Assessment" phase in which the research, development and assessment of the concept of disposal are being performed without consideration of specific sites. During this time the assessments are generic but with data derived from real locations where possible. The assessments are divided into two major parts (Fig. 27): the pre-closure and the post-closure assessments. The pre-closure assessment is considered somewhat conventional and will be commented upon only briefly before dealing in some detail with the post-closure assessment.

For the pre-closure phase the total impacts of transportation, immobilization, emplacement, backfilling and sealing and finally decommissioning and removal of surface facilities are considered. In safety studies the probability and consequences of accidents, both to the workers and to the general public, are analysed. The environmental impacts of radiological and non-radiological emissions are estimated. Consideration is given to resource utilization such as the possible use of lead as an investment material around fuel bundles and the use of land which might otherwise have farming or recreational potential.

Social and economic studies consider effects such as the impacts of an increased work force, extra traffic and the effects of extra load on local facilities like schools. Benefits are identified in this area, including long-term employment for a wide range of skills from scientific manpower to labourers.

Post-closure assessment includes the evaluation of processes within the vault (i.e. the waste repository), the geological formation and the biosphere. Methodology being developed and applied includes: computer programs, such as finite-element codes, for detailed analysis of components of the system;

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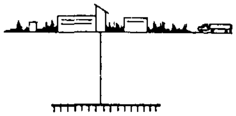
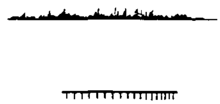
PRE-CLOSURE 	POST-CLOSURE 
SAFETY	SYSTEMS VARIABILITY ANALYSIS
ENVIRONMENTAL	HYDROGEOLOGICAL MODELING
SOCIAL AND ECONOMIC	CHEMICAL MODELING

FIG.27. Program components.

systems analysis codes for integrated analysis of many interrelated components; and 'systems variability analysis' of the whole system.

Detailed computer programs include hydrogeological and chemical modelling codes. Hydrogeological codes include finite difference codes for three-dimensional porous flow, heat-and-mass transport, finite element codes for regional flow, and codes for flow in interconnected rock fractures, together with routines for statistically analysing field measurements to prepare 'whole formation' input for fracture flow analysis. Chemical modelling codes analyse the complex equilibria between solutions and solids at one extreme and, at the other, empirical, often non-linear, relationships for representing such equilibria are being derived.

Systems analysis programs link together the hydrogeological, chemical and mass-transport processes within the vault, geosphere and biosphere, respectively.

The systems variability analysis code integrates the total system and sample data from distributions reflecting the uncertainty and variability in the data values. The resulting output is a histogram of consequence (dose to humans) versus probability, indicating the most probable consequence of the project, and other consequence estimates, together with their probability of occurrence.

The present status of the methodology development and application is described, together with results obtained to date.

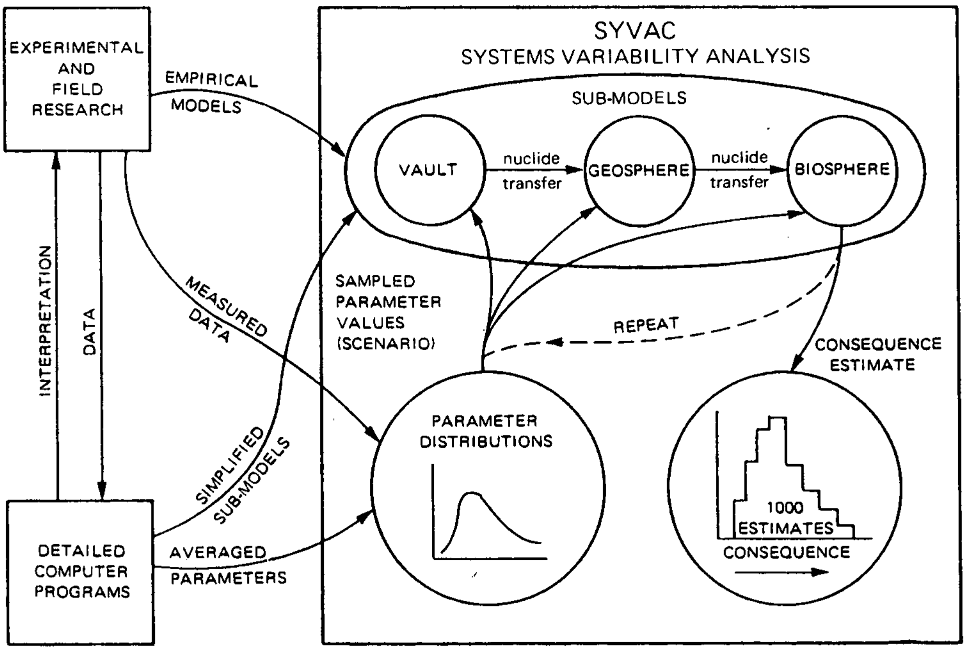


FIG.28. Post-closure assessment.

2. SCENARIO SELECTIONS

The objective of the post-closure assessment studies is to predict how radioactive material might escape from a disposal vault and migrate through the geosphere and biosphere to cause radiation dose to man.

Figure 28 illustrates the overall approach taken. Laboratory and field research provide data and empirical models, while detailed computer programs help interpret the measured data, which are used to derive averaged parameters, and often form a basis for the development of simple models. The assessment itself is carried out with a computer program called SYVAC which links together a set of sub-models to represent the disposal facility and its surroundings.

A central problem in predicting the consequence of a nuclear waste disposal operation in the distant future is the treatment of uncertainty and variability. One source of uncertainty is in the ability of mathematical models to describe the real system. This is generally resolved by validating the computer codes where possible, by comparing predictions with laboratory and field data.

Given a particular site and design, a second source of uncertainty is due to the error bands on measurement or specification of parameter values, and uncertainty about changes in the values with time. A third source is due to the requirement in the concept assessment phase that results are to be representative of a range of sites and design options.

SYVAC applies a procedure called systems variability analysis [5] to take account of the last two sources of uncertainty and variability. This is done by representing the sub-model defining parameters as distributions rather than as single values. Sampling a value of each parameter in turn from its distribution then characterizes a possible state of the system, or defines a "scenario". SYVAC then proceeds to estimate the transport of radionuclides from the vault to the biosphere and calculates a "consequence", at present taken as maximum dose to an individual in the most exposed group, irrespective of time of occurrence. Repeated sampling of scenarios and estimation of consequences result in a histogram of consequence estimates versus frequency of occurrence.

3. VAULT (REPOSITORY) ANALYSIS

As illustrated in Fig. 29, the current model for the vault (i.e. the waste repository) assumes that groundwater flow is vertical and that flow through the repository is estimated from flow in the rock, taking account of differences in hydraulic conductivity. Mass transport out of the repository is derived by assuming that the waste dissolves to its solubility limit in the water available. In the case of fuel the dissolution of the fuel matrix is considered to release the radionuclides in proportion to their concentrations in the fuel. Radionuclides considered to have migrated to the fuel/sheath gap during irradiation are assumed to be instantly released on container failure. Diffusion and advection, with delay due to chemical reaction, are analysed to estimate the transport rate across the buffer. A simple failure function has been assumed for the containers, based on engineering judgement.

4. DESCRIPTION OF HOST ROCK SITE AND REPOSITORY

The vault will be located in crystalline rock formations, known as plutons, located throughout the Canadian Shield. The characteristics of these formations are similar to those for the other hard rock formations described in Appendixes C and D. Thus, no further descriptions will be given here.

Figure 30 presents the major parameters associated with the vault, the inventory and distribution of the spent fuel in the disposal area. In this figure:

Uniform in log means that the logarithm of the variable is selected from a uniform distribution within a range from a maximum to a minimum value.

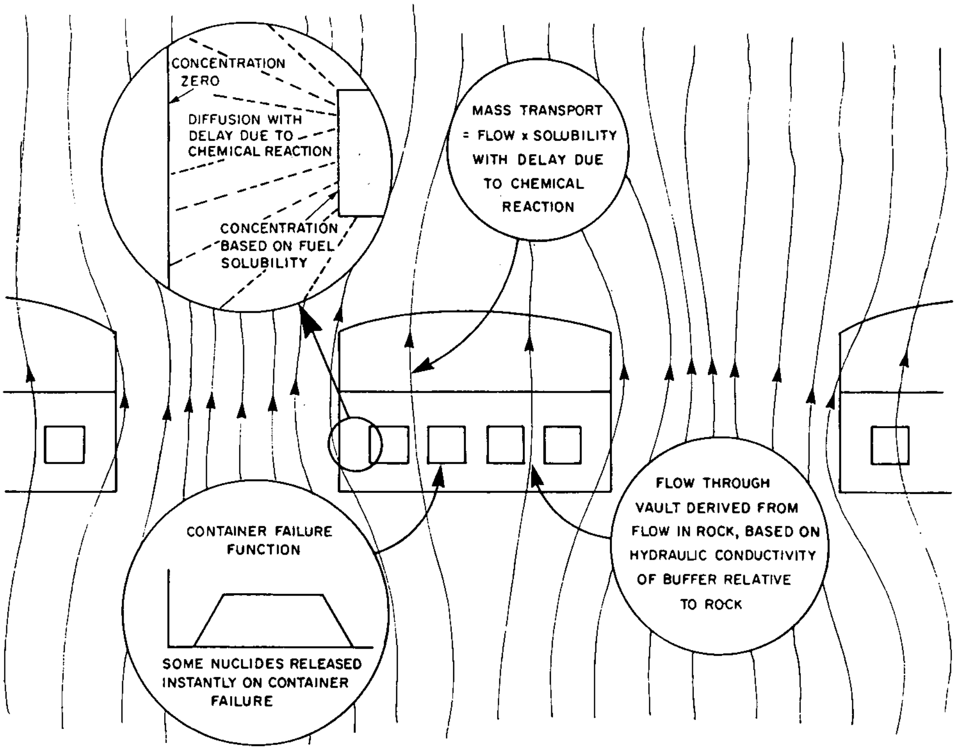
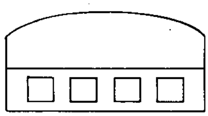


FIG.29. Vault sub-model.



	VAULT PARAMETERS
FUEL	350,000 T
CONTAINERS	246,000
TEMPERATURE	100°C
HORIZONTAL AREA	4.7 km^2
EFFECTIVE DIFFUSION AREA	9.3 m^2/CONTAINER
EFFECTIVE FLOW AREA	4.9 m^2/CONTAINER
SOLUBILITY OF UO_2	10^{-11} TO 10^{-8} MOLAL*
Cs AND I IN GAPS	0.7%
HYDRAULIC CONDUCTIVITY ROCK	10^{-12} TO 10^{-7} m/s*
HYDRAULIC GRADIENT	10^{-5} TO 10^{-3}
BUFFER THICKNESS	1 m
HYDRAULIC CONDUCTIVITY	10^{-14} TO 10^{-8}*
EFFECTIVE POROSITY	10^{-2} TO 10^{-1}*
K_d VALUES	*
TIME FOR FIRST CONTAINER FAILURE	20 TO 500 YEARS
TIME FOR LAST CONTAINER FAILURE	20 X TIME FOR FIRST

* UNIFORM IN LOG

FIG.30. Vault parameters.

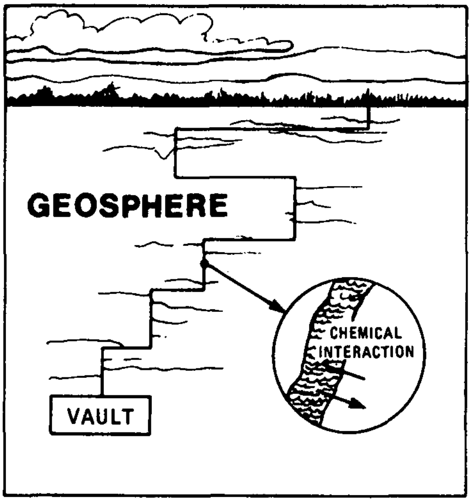


FIG.31. Geosphere sub-model.

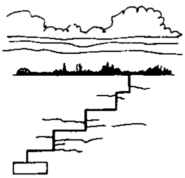
Gaps refer to the fuel/sheath gaps as referred to in the description of Fig. 29.

Buffer is a term for the backfill material immediately surrounding the waste. It could be a more expensive clay with chemicals added, compared to the rest of the backfill material.

5. GEOSPHERE ANALYSIS

The geosphere sub-model is essentially the same as that developed for the GARD [6] computer program. As illustrated in Fig. 31, a one-dimensional path is assumed for which the hydraulic parameters are derived from field measurements, results of hydrogeological codes and hydrogeologists' judgement. Chemical interactions are lumped into a simple retardation parameter which is defined as the ratio of water velocity to radionuclide velocity. This approximation is being improved as the chemical modelling studies mature.

Figure 32 lists the parameters used in the geosphere sub-model and their variability.



EFFECTIVE PATH LENGTH	1 TO 40; 40 TO 1000km
HYDRAULIC GRADIENT	10^{-5} TO 10^{-3}
PERMEABILITY	10^{-19} TO 10^{-14} m^2 *
POROSITY	10^{-6} TO 10^{-3} *

* UNIFORM IN LOG

FIG.32. Geosphere parameters.

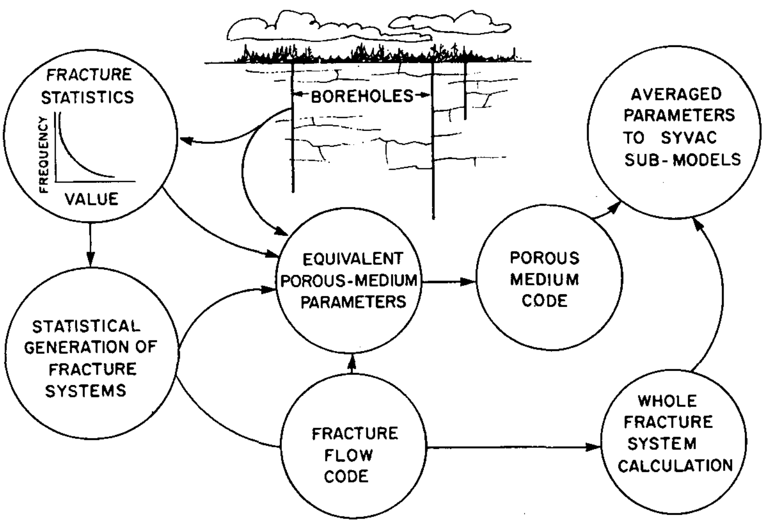


FIG.33. Hydrogeological modelling.

5.1. Geohydrological analysis

Figure 33 illustrates the application of computer programs to model the flow of groundwater in the vault and in the surrounding geological formation. It is necessary to model flow in both porous and fractured media. The latter presents some difficulty since there has been little interest in hydrogeological

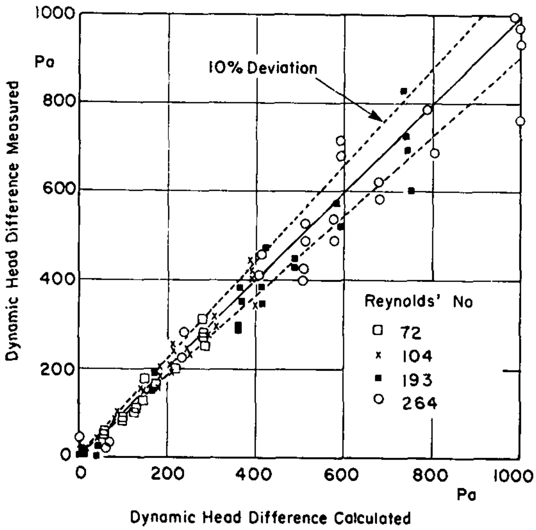


FIG.34. Fracture flow: Calculation versus experiment.

modelling of fractured media in the past and the capabilities are not well developed. Because of the advanced development of porous flow codes, it would be advantageous to represent the fractured medium as an equivalent porous medium. Suitable parameters could then be obtained either directly from field measurements, or by running the fracture flow code for a representative fracture system. The conditions under which the use of porous medium parameters would be acceptable are under investigation.

The SWIFT [7] code is mainly used for porous flow calculations and a code called FLOWNET [8] is being developed for fracture flow analysis. SWIFT employs a three-dimensional, transient or steady-state model and treats heat and solute transport as well as flow. FLOWNET analyses flow in a set of parallel-sided fractures which can intersect at arbitrary angles. To provide the input to FLOWNET, a computer program is being developed to use the fracture data derived from the field measurements and synthesize possible fracture systems.

Application of the porous flow code, or calculation of the whole fracture system with FLOWNET, provides averaged parameters for use in SYVAC. To begin the validation of FLOWNET, an experiment [8] was set up to flow water through cubical fractures formed by a lucite cube in a lucite box. Head measurements were taken over the faces of the cube and compared with values predicted from FLOWNET. As shown in Fig. 34, the comparison was good for almost a four-fold range of Reynolds' numbers.

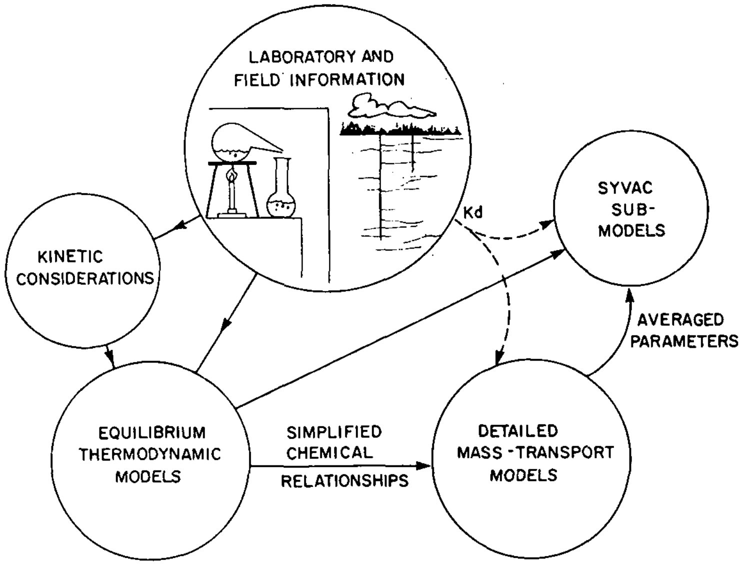


FIG.35. Chemical modelling.

5.2. Geochemical analysis

The object of the chemical modelling calculations is to predict the course of chemical reactions between the groundwater solutions and the contacted solids. This includes the processes of container corrosion, waste dissolution and mass transport with chemical retardation through buffer, backfill and fractured rock.

Figure 35 illustrates the general approach to chemical modelling. Water movement underground is expected to be very slow, so a great deal of attention is being given to equilibrium thermodynamic models. The major focus in this area is the SOLMNEQ [9] computer program, which has been modified and for which the data base has been extended to include uranium and plutonium species [10]. Kinetic effects are taken into account mainly by identifying those reactions which are slow even in the time scales of significance for disposal. Such reactions are then excluded from the equilibrium modelling.

Output from the chemical modelling can be used directly in SYVAC or in the form of simplified chemical relationships in detailed mass-transport codes. Figure 36 presents some of the chemical modelling results [11] which have been used directly in the SYVAC assessment. The total solubility of uranium species is plotted against pH. Eh buffering with magnetite-haematite represents redox

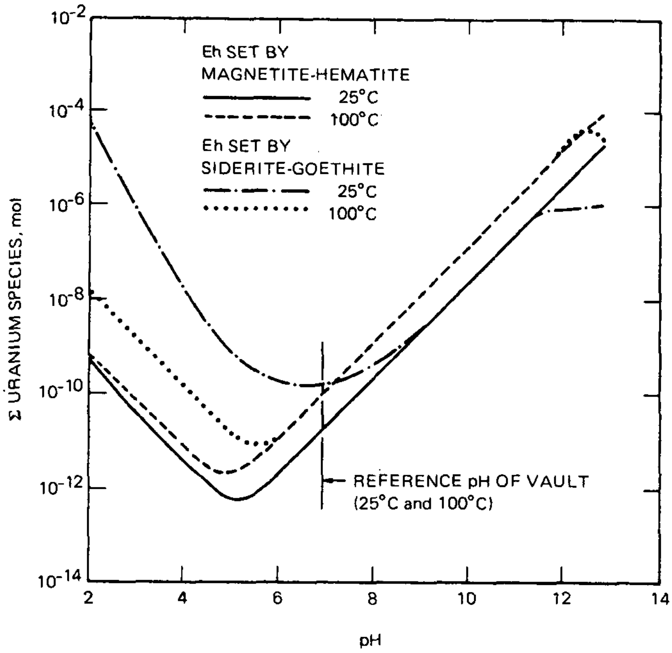


FIG.36. Solubility of uranium under disposal-vault conditions.

conditions expected in deep granite groundwater, and Eh buffering with siderite-goethite represents redox conditions expected in soils nearer the surface. From these curves it can be deduced that, over the range of conditions expected for a disposal vault in granite, the range of uranium solubility will be 10^{-11} and 10^{-8} molar. Uranium solubility is a very important parameter in the assessment studies; it is expected to control the rate of release of radionuclides from the fuel matrix.

6. BIOSPHERE AND DOSIMETRY ANALYSES

Figure 37 illustrates the biosphere model. Two options are assumed to be possible for the arrival of the radionuclides at the surface: arrival at a ground surface, such as a valley bottom, or directly into a lake.

One of these models is chosen by the SYVAC sampling procedure. The radionuclide concentrations in soil and water are then estimated by use of simple compartmental models, taking account of removal by run-off from the ground compartment or by flow from the lake. Dose/concentration ratios [12, 13] are

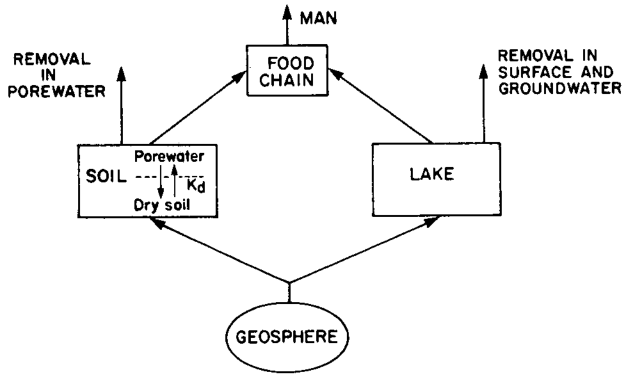
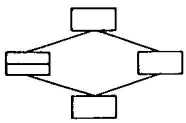


FIG.37. Biosphere sub-model.



PROBABILITY OF LAKE PATHWAY	0.9
FRACTION OF PRECIPITATION PENETRATING SOIL	0.9
VOLUME FRACTION OF WATER IN SOIL	0.2
SOIL COMPARTMENT DEPTH	1.2m
SOIL DENSITY	1500 kg/m^3
DISCHARGE SOIL AREA	20 TO 200 km^2
DISCHARGE LAKE AREA	1 TO 1000 km^2
PRECIPITATION LESS EVAPORATION	0.2 TO 0.6m/a
DISCHARGE LAKE DEPTH	15 m
CATCHMENT AREA / DISCHARGE LAKE AREA	1 TO 10
SOIL K_d VALUES	BY RADIONUCLIDE
DOSE PER UNIT SOIL CONCENTRATION	BY RADIONUCLIDE
DOSE PER UNIT WATER CONCENTRATION	BY RADIONUCLIDE

FIG.38. Biosphere parameters.

then used to calculate the dose to man. The ratios were derived by use of the FOOD II [14] and NEPTUN [15] computer programs which contain the assumption that interactions between the radionuclides in soil and plants, animals and man, occur sufficiently rapidly compared with changes in the soil and lake concentrations, that steady state is reached. Figure 38 gives the parameters used for the biosphere sub-model.

Figure 39 presents results of a single run through the sub-models with one set of parameters or "scenario". In this case the maximum dose, which is taken as the consequence estimate, is due to release of the ^{129}I in the fuel/sheath gaps.

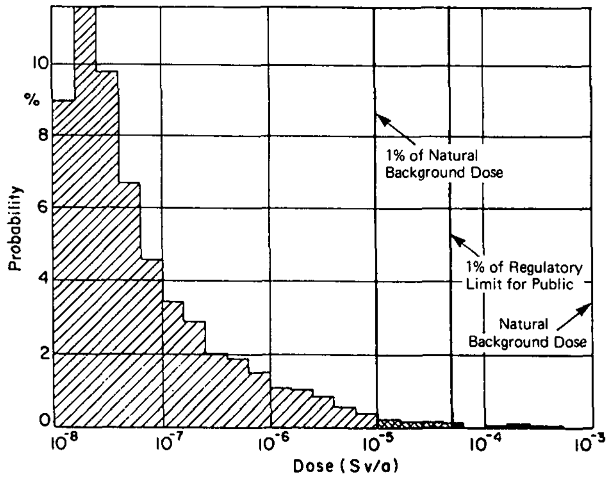


FIG.39. Maximum dose to most exposed individual.

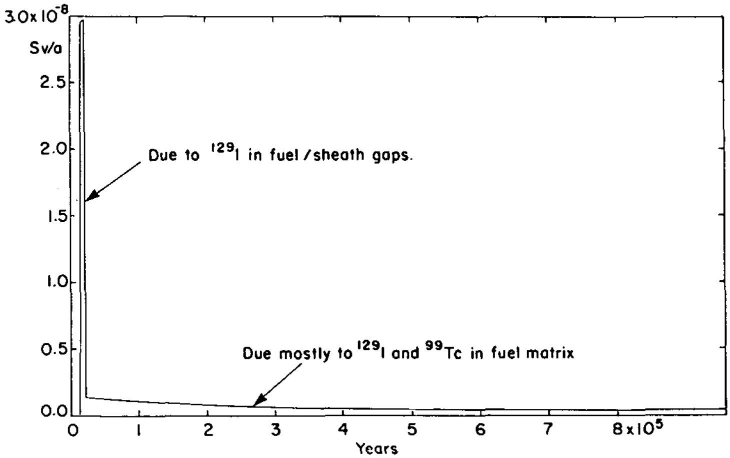


FIG.40. Single case – dose versus time.

The subsequent, slowly reducing dose is mostly due to release of ^{129}I and ^{99}Tc as the fuel matrix dissolves.

Figure 40 gives the results of over 3000 estimates of maximum dose to an individual in the most exposed group. On this histogram are drawn lines at natural background dose, at 1% of natural background and at 1% of the regulatory limit for members of the public. About 1% of the dose estimates exceeded 1% of the natural background and about 0.5% exceeded 1% of the regulatory limit.

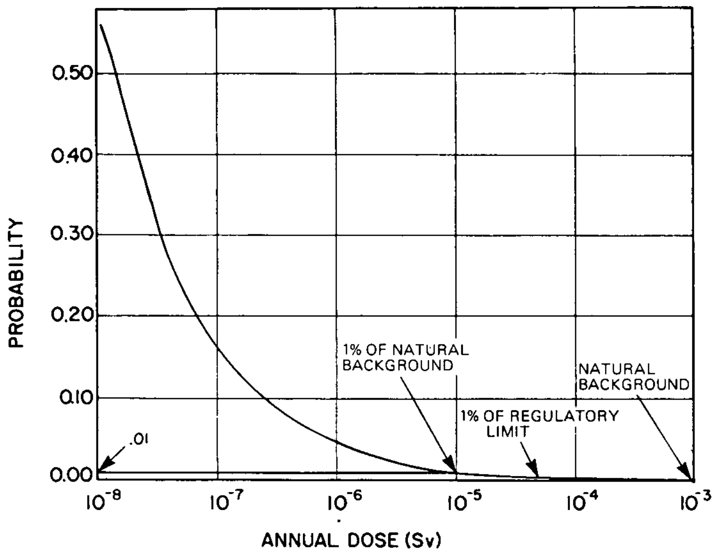


FIG.41. Downward cumulative probability of maximum annual dose.

None of these estimates exceeded natural background. Approximately 2000 other cases, in which no dose results before one million years (owing to an estimated transit time to the surface exceeding a million years), were excluded from the histogram. This time cut-off was arbitrarily chosen but probably could be justified on the basis that doses beyond this time are due mostly to the daughters of ^{238}U. One thousand cases were carried out to 10 million years and showed no significant effect on the histogram other than a higher proportion of uranium doses due to ^{234}Th, as would be expected.

An alternative way of plotting these results is shown in Fig. 41 where the downward cumulative probability is plotted against the annual dose estimate. Again, possible measures of acceptability are indicated and the probability of exceeding those measures can be read from the curve. For example, the probability of a consequence estimate exceeding 1% of natural background can be read as 0.01, or 1%.

This approach is believed to provide a framework for defining a criterion of acceptability for a nuclear waste disposal project. If this is acceptable to the regulatory authorities, it would then be appropriate for them to define suitable consequence estimators, an appropriate level of acceptability and an acceptable probability of exceeding that level. One consequence estimator is maximum dose to an individual in the most exposed group, irrespective of time of occurrence. Other estimators could be used, such as population dose integrated over time.

7. CONCLUSIONS

The Canadian studies on the environmental and safety assessment for nuclear fuel waste disposal to date have considered the pre-closure and post-closure phases for the disposal of intact fuel bundles in crystalline rock of the Canadian Shield. Most of the research and development are focused on the post-closure phase. The current assessments of the post-closure phase pay particular attention to the problem of uncertainty in predictions into the distant future. Results to date, while based on preliminary information from the research programmes, indicate that future generations should not experience radiation doses, due to the disposal operation, exceeding a small fraction of natural background.

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Appendix F
BELGIAN CLAY REPOSITORY

A. Bonne

1. INTRODUCTION

In 1974 a special working committee was established by the Belgian Minister of Economic Affairs to assess the various aspects of nuclear energy and to give the authorities independent advice on the future energy policy. Among the numerous conclusions and recommendations of this committee, published in March 1975, it was stated that, taking into account the Belgian potentials, argillaceous formations are believed to be the most suitable available ones for the disposal of high-level and alpha-bearing wastes. The report also recommended that research on that disposal concept should be supported.

Since that time, CEN/SCK (Nuclear Energy Research Centre) of Mol considerably expanded the work in its research and development programme (already started in 1974) concerning the possibilities of radioactive waste disposal. This programme aims to obtain knowledge about the disposal system behaviour and the appropriate engineering technologies so as to assess the safety and feasibility of the disposal concept.

The research and development programme is site-specific and is focused on the Boom clay formation underlying the nuclear site of Mol. Thus, the safety assessment exercises now under way are based on a repository concept specific for the Boom clay at the site, and the geoscientific data to be used in the safety analysis are specific to the geological context of the area. The current programme on safety assessment considers only the post-closure phase; assessments for the operational phase are to be performed later.

The policy at present adopted for the safety analysis studies is to limit the development of new models and codes and to rely generally on, and to use, already existing modelling. This allows validation of existing models by using them with real data for the specific site. The CEN/SCK programme is carried out under a contract between the Commission of the European Communities (CEC) and CEN/SCK, and thus preference is given to models and codes developed within the framework of the actions undertaken by the CEC. Thus, part of the safety analysis studies for the Belgian clay repository is performed in close collaboration with the Joint Research Centre of the Commission of the European Communities at Ispra (Italy), which developed its own methodology for safety analysis.

At present no comprehensive safety assessment report for the Belgian clay repository is available, the studies still being in progress. The following sections of this Appendix thus outline the approach and methodology applied.

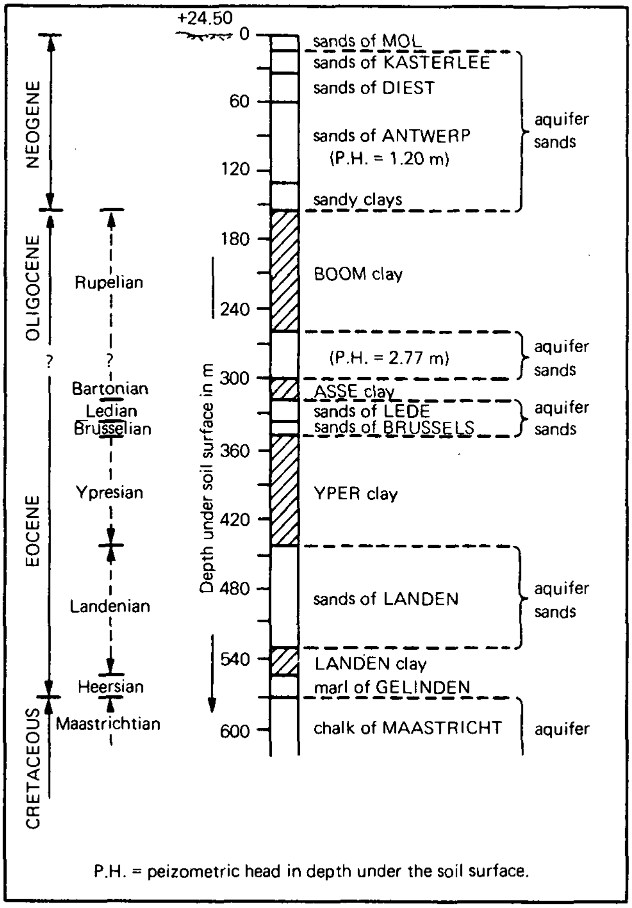

FIG.42. Simplified geological column section at the Mol site.

2. DESCRIPTION OF HOST ROCK SITE AND REPOSITORY

2.1. Host rock and site characteristics

The repository site is located in the northeastern part of Belgium, where the national nuclear research centre of Mol and other nuclear facilities are situated. In that location the Boom clay (see simplified geological column section in Fig. 42) occurs from approximately 160 to 270 m below land surface, which is about 25 m above the present mean North Sea level. This clay formation belongs to a more extended Oligocene clay sedimentation province (± 3 - 3.5 X 10^7 years old) found in northern and central Europe. In the Mol area the Boom clay dips (± 1%) to the northeast and is covered by more recent, mainly Miocene glauconiferous sands.

The characteristics of the Boom clay were determined on samples cored during several drilling campaigns at the Mol site (see Table XXII). System analysis studies are under way to evaluate changes of these characteristics due to waste host rock interactions, especially near-field phenomena. Also in-situ determination of some of these characteristics is planned for future work (e.g. influence of temperature on in-situ geomechanical properties, in-situ thermal conductivity, etc.).

In the context of site investigations, characteristics of the overlying and underlying formations were also studied in field experiments (e.g. pumping tests) or laboratory tests. Hydraulic parameters and data in particular were determined. The underlying aquifer (Berg sands) is composed of very fine sands. Its average hydraulic conductivity (k), determined by pumping tests at the site, is approximately 4.2×10^{-5} $cm \cdot s^{-1}$. The overlying neogenic sandy deposits may be subdivided into four subunits:

Mol sands (k = approx. 1.8×10^{-2} $cm \cdot s^{-1}$); Kasterlee sands (k = probably similar to Mol sands); Diest sands (k = approx. 2.8×10^{-3} $cm \cdot s^{-1}$) and Antwerp sands (k = approx. 1.2×10^{-3} $cm \cdot s^{-1}$).

A regional hydrological survey has shown that at several places different aquiferous layers are present in the overburden sands of the Boom clay. From well observations it was also learned that at the Mol site a downward gradient of 0.01–0.02 exists. Further details about the site may be found in Ref. [1].

2.2. Description of the repository and waste inventory

For the Belgian clay case the repository size is based on the amount of vitrified high-level waste and alpha-bearing wastes arising from the 30 years operation of a 10 GW(e) nuclear power programme or 300 GW(e) · a equivalent. The disposal area needed is roughly 1.5 km^2. The disposal horizon could be in the mid-plane of the Boom clay, at a depth of ⩾ 200 m below land surface.

The high-level waste is vitrified as borosilicate glass and enclosed in corrosion resistant cylinders (30 cm in diameter and 150 cm long). Similar canisters are considered to be used for cladding wastes. Because of heat limitations the high-level waste will be stored for at leat 50 years after reactor discharge before emplacement in the repository. Alpha waste and medium-level waste are packaged in 200-litre carbon steel drums, with concrete or bitumen immobilization.

TABLE XXII. SOME CHARACTERISTICS OF THE BOOM CLAY (MOL SITE)

Chemical composition of dry material (%)	~ 64 SiO_2, ~ 14 Al_2O_3 ~ 5.9 Fe_2O_3, ~ 2.2 K_2O ~ 1.4 Na_2O, ~ 0.6 CaO ~ 0.5 TiO_2, ~ 0.7 MgO weight loss at 1000°C: ~ 10
Natural water content (wt %)	~ 26
Mineralogical composition of the fraction: < 2μm (parts per ten)	Illite (2–3), smectite (2), vermiculite-like (3), illite-montmorillonite interstratified (1–2), chlorite + chlorite-vermiculite-like interstratified (1)
Organic matter (%, 14 samples)	2.3–5.5
Granulometric omposition (%)	d < 2 μm: 49 2 μm < dia. < 60 μm: 47 60 μm < dia. < 200 μm: 3.5 dia. > 200 μm: 0.5
Bulk density	~ 1.93
Dry density	~ 1.53
Hydraulic conductivity ($cm \cdot s^{-1}$)	Between 1.4×10^{-8} and 4.7×10^{-10}
Porosity (%)	Between 34.6 and 44
Saturation degree (%)	Between 88.4 and 100
Plasticity limit (%) (average 50 samples) geotech. core drilling)	~ 27
Liquidity limit (%) (average 50 samples)	~ 76
Index of plasticity (%) (average 50 samples):	~ 49
Thermal conductivity ($Wm^{-1} \cdot K^{-1}$)	0.9–1.3 (from 20 to 90°C) 0.3–0.5 at 100°C and 0.6–0.8 at 500°C
Natural radioactivity ($Bq \cdot kg^{-1}$ dry sample)	$^{40}K \sim 7.4 \times 10^2$ $^{226}Ra \sim 7.4 \times 10^1$ $^{232}Th \sim 4.4 \times 10^1$
Cation exchange capacity (meq. per 100 g dry clay)	20–40, depending on sample and technique used

The physical inventory of waste packages, representing 300 GW(e) · a energy production, is summarized below:

Unshielded drums	6 000
Shielded drums	150 000
High-level waste canisters	9 000
Cladding waste canisters	9 000

The radionuclide inventory in the high-level waste is calculated by the ORIGEN code.

As far as alpha waste is concerned, only one stream, including radionuclides from reprocessing (1% Pu loss) and MOX fabrication (2% Pu loss), has been considered.

Iodine waste is considered as a separate waste type, 99% of the iodine being removed during the fuel dissolution step in the reprocessing plant and 1% remaining in the high-level waste; however, all the ^{129}I could be considered to be emplaced in the repository.

A feasibility study for the underground repository led to the design of several repository geometries, resulting from the different possible emplacement techniques for the high-level waste. For the initial safety analysis studies, only one design, described below, was taken into account.

The underground part of such a disposal facility could be composed of seven parallel disposal galleries (secondary galleries), interconnected by a main gallery, allowing the transport of the waste from the access shaft towards the disposal galleries. The distance between the disposal galleries for high-level waste could be 225 m and for medium-level, alpha-bearing waste and cladding hulls 35 m. The length of these galleries could be approximately 2.5 km, with the exception of the gallery for cladding hulls, around 1.8 km long. All galleries are cylindrical. The emplacement of canisters of high-level waste and cladding hulls could be performed in inclined lined holes at the bases of the galleries. The distances between the inclined stacks of high-level waste and cladding hulls are, respectively, 20 and 4.5 m.

After emplacement of the waste packages, the tunnels, holes and shafts will be backfilled with clay or clay mixtures.

Further details about the concept design of the underground facility may be found in Ref. [2]. Alternative designs may also be developed and used in forthcoming analyses.

3. SCENARIO SELECTIONS

The scenario analysis aims at evaluating how disposed radionuclides could leave the repository and/or the host formation and how they could return back to the biosphere and ultimately to man.

TABLE XXIII. MEAN PROBABILITY OF DISRUPTIVE EVENT OCCURRENCE FOR THE BOOM CLAY AT THE MOL SITE

Receptor / Time span (years)	Groundwater	Land surface	Atmosphere
2×10^3	9.65×10^{-5}	3.35×10^{-6}	1.1×10^{-8}
2.5×10^4	8.54×10^{-4}	2.44×10^{-5}	1.34×10^{-7}
1×10^5	3.8×10^{-3}	7.9×10^{-4}	5.6×10^{-7}
2.5×10^5	5.2×10^{-3}	3.4×10^{-3}	1.34×10^{-6}

Several elements of the repository system act as a barrier, hampering the release of the radionuclides. To define the scenarios by which the barriers could be breached by disruptive events or processes originating from outside the repository, a screening was made of the slow processes known to have been active in the site area during the Quaternary and Tertiary geohistory. These processes are associated with eustatic movements, glaciation, epeirogenesis, etc. Based on the same rates or intensities for these processes in the future as reckoned from the past, it is estimated that such slow processes will not reach or perturb the top of the host formation in a time span of 2×10^5 years [3].

A more comprehensive assessment of geological containment failure was performed by applying the fault-tree analysis technique [4]. This is the first step in applying the JRC (Ispra) methodology for the Belgian clay case. The fault-tree analysis technique allows for identification of the possible failure modes of the geological containment and assessment of the probability of occurrence of such events. In the analysis sudden and slow natural events and human activities, inspired by the needs of natural resources, were considered. In this approach three possible release receptors were identified – groundwater, land surface and atmosphere. For each receptor failure, probability ranges were estimated for the following time spans: 2×10^3 years, 2.5×10^4 years, 10^5 years, 2.5×10^5 years. In Table XXIII the mean probability values are given for the four time spans and the three receptors. This analysis also revealed that, for shorter time spans (2×10^3 years), the human activities have a higher weight than the natural events. For longer time spans (more than 2.5×10^4 years) natural events present a higher weight in the probability values.

From an examination of the fault-tree for the release to land surface it is seen that aquifer contamination followed by radionuclide migration through the subsoil may be the most likely scenario able to cause an environmental contamination, mainly during the longest time spans. For what concerns the aquifer contamination, faulting phenomena were found to be among the principal mechanisms having the potential to cause radionuclide release to groundwater; in particular, faults characterized by a displacement greater than 5 to 10 m can play a major role in governing the failure probability.

For a first exercise the scenario analysis of such a geological barrier failure starts from the conservative assumption that, once the formation has been breached by a fault, a fraction of the radionuclides is leached from the wastes by flowing water and transported into adjacent aquifers. Subsequent transport of the radionuclides through the lower and upper aquifers, entrance of the contaminated water plume into the zone of influence of a well, and use of this water for domestic and agricultural purposes, constitute the successive steps by which radionuclides reach man.

Up to now, only an abnormal release scenario, where a post-closure incident originates from outside the repository, has been studied. However, analysis of the repository system for a normal release scenario is also under way. This scenario is based on the assumption that the different repository components (some of them acting as barriers) will eventually be contacted by interstitial water of the clay. This interstitial solution will act as a medium for facilitating mutual interaction between the different repository components (including the near-field host rock). Natural degradation of the repository will thus occur and corrosion of canisters and waste matrix can release immobilized radionuclides, forming the source term for migration through the clay formation. In normal conditions the only way for the radionuclides to reach the aquifers is by migration through the interstices of the clay formation. For calculating the release from the host formation a three-dimensional migration model has been developed [5]. Some exercises with this model have been performed [6] but not yet specifically for the safety analysis conditions.

4. REPOSITORY ANALYSIS

The repository analysis performed up to now has to be understood in the framework of the incidental scenario of a tectonic displacement through the repository. The orientation proposed for the galleries of the repository is WSW-ENE and thus perpendicular to the most frequent direction of the detected Quaternary and Tertiary faults. Based on an assumed width of 10 metres for the fault zone, a simple calculation shows that about 0.5% of the overall waste volume could be contacted and leached by groundwater flowing through the fractured zone. To take into account the possibility of a larger fault zone, or in particular unfavourable fault plane orientations, it is assumed in the first consequence analysis exercise that a fraction as high as 5% of the disposed waste is leached. The following leach rates were adopted for the different waste types considered:

(a) *Vitrified high-level waste:* Starting from the assumption that the tectonic event occurs 2000 years after emplacement, it may be calculated that the temperature will be reduced to a very low level and the repository temperature will be nearly the normal geothermal temperature. The usual leach rates of 10^{-7} g · cm^{-2} · d^{-1} and a geometrical specific surface area of 0.5 cm^{2} · g^{-1} were assumed.
(b) *Conditioned alpha-bearing waste:* A leach rate of 10^{-7} g · cm^{-2} · d^{-1} and a geometrical surface area of 0.5 cm^{2} · g^{-1} were assumed.
(c) *Conditioned iodine waste:* A leach rate of 10^{-7} g · cm^{-2} · d^{-1} and a geometrical surface area of 0.5 cm^{2} · g^{-1} were assumed.

In the repository analysis for the abnormal scenario no value was attributed to canisters or to other engineered barriers. The leaching model used is based on the following assumptions:

(a) All isotopes are leached at the same rate;
(b) Leach rate remains constant with time;
(c) Specific surface remains constant over the leaching period;
(d) Duration of leaching is 2000 years.

Because of the early stage of work on the normal scenario, no comments will be made on the repository analysis of this case.

5. GEOSPHERE ANALYSIS

For the consequence analysis for the abnormal scenario the modelling performed for the geosphere analysis was not sophisticated.

5.1. Hydrological studies

Hydrological investigations in the site area have been carried out since 1975, and recently a more detailed network of hydrological observation wells was installed in the hydrographic basin of the site. Hydrological modelling of the multi-layered sedimentary overburden of the Boom clay and the underlying aquifer is under way.

For analysis of the abnormal scenario, water-flow velocities in the underlying and overlying aquifers were estimated, based on hydrological data gathered by field observations and testing. Water-flow velocities of 1 m · a^{-1} and

$100\ m \cdot a^{-1}$, respectively, were considered for the aquifers under and over the Boom clay.

5.2. Mass transport

Because of the appreciable amount of glauconite, the overlying sandy formations will act as a geochemical barrier by retaining or delaying the transport of released radionuclides. Retardation of radionuclides relative to the water-flow velocity can be expressed in sorption equilibrium conditions, by introducing a retardation factor (R) in the mass transfer equation. When taking also into account radioactive decay, the radionuclide migration (one-dimensional) is represented by Eq.(1):

$$D \frac{\delta C_i}{\delta t} = D_i \frac{\delta^2 C_i}{\delta x^2} - V \frac{\delta C_i}{\delta x} - R\lambda_i C_i \tag{1}$$

where C_i = concentration of radionuclide i
D_i = axial dispersion coefficient of radionuclide i
V = interstitial water flow velocity
x = distance
λ_i = decay constant of radionuclide i
R = retardation factor

The sorption parameters are based partly on laboratory investigations, partly on literature data. Some radionuclides, e.g. Tc and I, are considered not to be susceptible for sorption in these aquifers. Laboratory investigations also showed that only a small fraction of the actinide elements is freely transported [7]. In the present exercise 0.1% Pu and Am and 2.5% Np are considered as mobile fractions conveyed freely with water-flow velocity.

6. BIOSPHERE AND DOSIMETRY ANALYSIS

In the consequence analysis of the abnormal scenarios the following pathways are considered:

(a) After migration over 1 km from the release zone, the contaminated groundwater is assumed to enter the zone of influence of well-water extraction. The concentration of any radionuclide in the well may be simply calculated by Eq.(2):

$$C_i = \frac{x_i}{P} \tag{2}$$

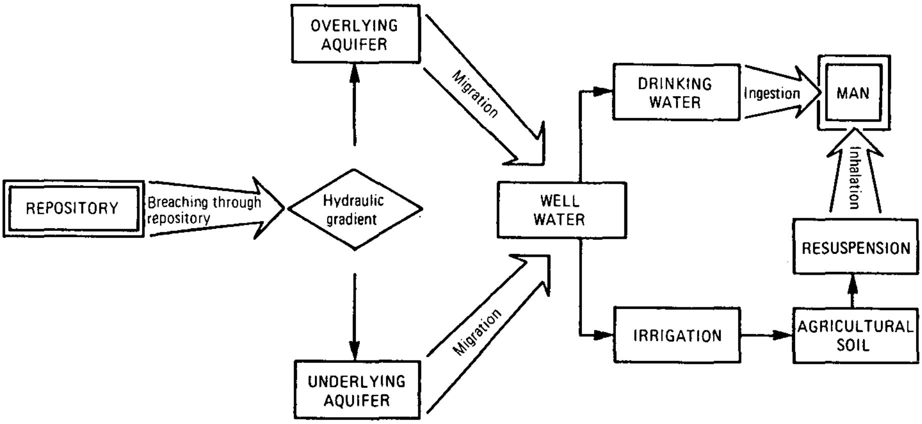

FIG.43. Release scenarios and pathways to man considered in the Belgian clay case (abnormal scenario).

where x_i = arrival rate of isotope i
P = water extraction rate.

(b) With direct consumption of this water as drinking water, the corresponding doses are calculated using the dose factors given in the literature [8].

(c) When using the well water for irrigation of agricultural soils, ingestion pathways according to different food chains can be considered. However, because the ingestion risk is expected to be governed by the direct ingestion of contaminated water, the ingestion pathways through food have not yet been studied.

(d) With an inhalation pathway (soil to air to man), calculations are based on some quantity of soil dust resuspended in the air. Inhalation doses can thus be calculated through the use of inhalation dose factors [8].

The release scenarios and pathways back to man, considered in the present exercises, are shown in Fig. 43.

7. GENERAL CONCLUSIONS AND REMARKS

The Belgian safety assessment study for disposal of radioactive waste into a pastic clay formation is site-specific and concerns only the post-closure phase.

Release scenarios for an abnormal disruptive event and for a normal future evolution of the repository are being taken into account for the safety assessments.

The first exercises for the abnormal case are being completed and results will be available soon. The analysis based on the normal scenario of natural degradation of the repository is in an early phase. In the future work much attention will be drawn on data uncertainty, parametric sensitivity of the model output and coupling of the incidental and degradational models.

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