Handling of Spent Nuclear Fuel and Final Storage of Vitrified High Level Reprocessing Waste

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1 INTRODUCTION

1.1 GOAL, SCOPE AND IMPLEMENTATION OF SAFETY ANALYSIS

It has been the function of a special group in the KBS project to:

- examine the methods proposed by the project for the handling and final storage of nuclear fuel waste from the viewpoint of safety and environmental considerations
- provide a basis for the design of safety-enchancing measures
- perform analyses of long-term safety and the consequences of various types of failures and accidents
- submit a summary safety evaluation

The group has continuously participated in and been kept informed of the work of other groups.

A reference group consisting of specialists from various fields has served the project in an advisory capacity.

Information has been obtained from similar foreign projects especially in the USA and Canada. In Sweden, various researchers and groups of specialists were engaged for important parts of the work.

The present report constitutes a summarizing analysis of the handling, storage and transportation of spent fuel and vitrified high-level waste and the long-term safety of the final storage of vitrified high-level waste in rock. The final storage of spent fuel will be dealt with in a forthcoming report.

More detailed accounts of the various parts of the programme are provided in technical reports, to which references are made. General safety and security matters are dealt with in chapter III:7.

After an introductory section on general safety requirements, a brief background on laws, standards and grounds for evaluation in relation to nuclear activities is provided in chapter 2.

Chapter 3 provides a general survey of the various radioactive elements and the generation of decay heat in spent fuel and vitrified high-level waste, and how this decreases with time. The same chapter deals with how the relative hazard of the various radioactive elements varies with time.

The safety-related aspects of the handling, storage and transportation of spent fuel and vitrified waste prior to final storage are dealt with in chapter 4.

Chapter 5 presents the most essential safety principles which govern the final storage of vitrified high-level waste in the Swedish bedrock.

The crucial issue of the possibility of the escape and dispersal of radioactivity from the final repository and the short- and long-term consequences of such dispersal from the viewpoint of radiation doses and medical effects is dealt with in chapter 6. The quantities of radioactive elements leached from glass, their transport via the groundwater, sorption effects, various paths of exposure and dose loads on persons living nearby and on the population in general are discussed on the basis of various conceivable dispersal mechanisms. The importance of extreme events from the safety point of view is dealt with in chapter 7.

Chapter 8 provides a summary of the safety analysis.

1.2 SAFETY REQUIREMENTS – GENERAL

There are special laws and regulations in Sweden which govern nuclear power activities and which provide a basis for the safety and protection of personnel, nearby residents and the environment.

The National Nuclear Power Inspectorate and the National Institute of Radiation Protection are responsible for the enforcement of laws and regulations in this field. These organizations are the inspection authorities designated by the Atomic Energy Act and the Radiation Protection Act. The Nuclear Power Inspectorate is mainly responsible for examining the safety of facilities and the design of various safety systems. The National Institute of Radiation Protection deals with matters pertaining to radiation protection in the working environment and in the external environment.

The general criteria for safety and radiation protection which comprise the basis for the inspection work and licensing activities of the inspection authorities require that facilities, processes, safety systems and activities be designed in such a manner that:

- the dose load to personnel, nearby residents and the regional and global population is low
- the risk level resulting from accidents is low
- every effort is made to minimize the dose load within the bounds of reasonable cost.

Chapter 2 illustrates how these criteria are applied. In the USA especially, general technical criteria have been embodied in more specific standards concerning the design of nuclear facilities and systems.

The explication of the Swedish Conditions Act (which specifies the criteria which must be fulfilled before new nuclear power plants can be commissioned) states that the final repository must be able to meet radiation protection requirements which aim at preventing radiation injuries. The final repository shall be designed in such a manner that the waste or the spent fuel can be isolated for as long a period of time as is required for its radioactivity to have decreased to a harmless level. The evaluation of the safety of the final repository shall take into consideration the risk that the waste or the spent fuel might escape to the biosphere by means of natural processes, accidents or acts of war.

2 LAWS, STANDARDS AND GROUNDS FOR EVALUATION

2.1 GENERAL

In general, extremely stringent safety rules are applied to nuclear power activities, resulting in very low risks to human health and the environment in connection with nuclear power. This high level of safety is a direct outcome of the extensive work on criteria and standards aimed at creating safe designs and the rigorous radiation protection recommendations for plant personnel and the general population issued by the International Commission of Radiological Protection (ICRP), whose principles have been accepted in all countries.

The work of other organizations as well, such as the World Health Organization (WHO) and the International Atomic Energy Agency (IAEA), as well as national authorities, has also contributed towards the safe and environmentally hygienic peaceful utilization of nuclear power.

Loss liability in the area of atomic energy is regulated in most western European countries by the Paris Convention. The Paris Convention and its supplementary convention are complemented by the Brussels Convention, which regulates loss liability in connection with the transport of nuclear fuel by sea. In Sweden, such liability is regulated by the Atomic Liability Act of 1968 (SFS 1968:45), which places the liability for an atomic accident with the plant owner. However, the plant owner's liability is limited to SKr 50 million per accident. Sums in access of this are paid by the state up to SKr 350 million, and after that up to approximately SKr 600 million by the states which have ratified the Brussels Convention.

Nuclear activities in Scandinavia must also comply with the Nordic Environmental Protection Cenvention. This convention was ratified by Sweden in 1976.

The dumping of wastes, including radioactive waste, in the sea is regulated by the London Convention. The Swedish Parliament, however, has prohibited dumping at sea altogether.

2.2 LAWS AND REGULATIONS

In Sweden and other countries, nuclear activities are regulated by a number of mutually complementary laws aimed at ensuring safety and protection for plant personnel, the general population and the environment. The central law within the field of nuclear power in Sweden is the Atomic Energy Act (SFS 1956:305), which requires permission from the government or an authority appointed by the government for the erection and operation of nuclear power plants or plants for the processing of nuclear fuel.

The inspection authority designated by the Atomic Energy Act is the National Nuclear Power Inspectorate, which is responsible for such matters as examining the safety of nuclear facilities and the design of various safety systems. Fissionable material safeguards and permits for the transportation of fissionable material also come under the authority of the Inspectorate.

The Radiation Protection Act contains provisions governing work with ionizing radiation, among other things. Permission for such activities is required from the inspection authority, the National Institute of Radiation Protection. Conditions and directives governing such activities are also issued by this authority. Matters pertaining to radiation protection in both the working environment and the external environment are also regulated by the Institute, which issues directives concerning:

- Maximum permissible radiation doses for personnel in radiological occupations and the measurement and reporting of such doses.
- Maximum permissible releases of radioactive substances and how such radioactivity is to be measured and reported.
- Environmental monitoring with sampling and analyses of different types of samples as well as direct measurements.

Other Swedish laws which govern nuclear activities include:

- Environmental Protection Act
- Workers' Protection Act
- Building Code
- Emergency Planning Act

2.3 INTERNATIONAL RECOMMENDATIONS

International organizations such as the International Commission on Radiological Protection (ICRP), the United Nations' International Atomic Energy Agency (IAEA), the World Health Organization (WHO) and the OECD's Nuclear Energy Agency (NEA) agree on the following fundamental principles:

- No activity which entails the irradiation of personnel or population shall be accepted before it can be demonstrated that its advantages outweigh its disadvantages from the viewpoint of society.
- The activity must be acceptable with respect to radiation risks.
- All radiation doses shall be kept within limits which are regarded as reasonable on the basis of economic and social considerations.
- No individual shall receive radiation doses which exceed the dose limits recommended by the ICRP, either now or in the future.

The most recently updated edition of the ICRP recommendations /2-2/ was published in September of 1977. The intentions of this and previous relevant publications from the ICRP served as a basis for the work of this project.

The recommendations concerning maximum permissible radiation doses have not been changed. Thus, the following limits apply:

- Radiation dose to personnel in radiological work, 5 rems per year.
- Radiation dose to individuals in the population, 0.5 rems per year.

The term "weighted whole-body dose" has been introduced. The purpose of this is to weigh together all doses to different organs to arrive at a representative whole-body dose.

The term "dose commitment" has been introduced. By dose commitment is meant the sum of the annual radiation doses which are the results of releases over one year. This means that the annual radiation dose in a future postulated state of equilibrium is equal to the dose commitment from the releases over a single year. Dose commitment can also designate the total dose load from failure releases.

The term "collective dose" refers to the sum of all of the individual doses within a given population. The purpose of setting a limit for the collective dose is to limit the future mean dose - and thereby the number of injuries - from a fully expanded nuclear power industry.

2.4 SWEDISH RADIATION PROTECTION STANDARDS AND CRITERIA

New regulations governing the release of radioactive substances from nuclear power plants were adopted by the government in 1977 on the basis of proposals from the National Institute of Radiation Protection /2-3/. These regulations will enter into effect in 1981. Transitional regulations will apply until then.

The new regulations stipulate limits for whole-body doses to nearby residents and collective doses to the entire population. The values which are specified are:

- The sum of the weighted whole-body dose to nearby residents should be less than 10 mrems per year.
- The global weighted collective dose commitment should be less than 0.5 manrems per year and MW installed electrical output (MWe).

These limits are considerably more stringent than former limits. They have been determined on the basis of an evaluation of what is currently the lowest dose load which can reasonably be achieved.

If these requirements are fulfilled, acute injuries to any individual will be entirely eliminated. The safety margin to direct health effects is several powers of 10.

The limits stipulated by the new standards aim at minimizing the risk of delayed effects, both somatic and genetic. The reference value of 10 mrems per year limits the radiation addition to the normal radiation environment to less than 10%.

The regulation governing releases of radioactivity from nuclear power plants also include provisions concerning:

- Counter measures in the event of elevated release levels
- Routines for inspection and reporting
- Scope of environmental studies

The transport regulations of the International Atomic Energy Agency, Regulation for the Safe Transport of Radioactive Materials (IAEA Safety Series No. 6), govern the transport of spent nuclear fuel and other radioactive material. There are also Swedish and international rules governing transports via various means. The supervisory authorities are the National Nuclear Power Inspectorate and the National Institute of Radiation Protection.

2.5 DESIGN STANDARDS

Special rules for nuclear facilities other than nuclear power plants - such as central storage depots for spent fuel and final repositories, intermediate stores for high-level waste and encapsulated spent fuel - have not yet been issued in Sweden. But the general principles established for nuclear power plants should also be applicable in respect of safeguards and the evaluation of safety and environmental aspects. Some modifications may be necessary according to the nature of the facilities and processes. General protection principles have already been established for the transportation of spent fuel and for the storage and treatment of such material.

Some design-guiding standards concerning the storage of spent nuclear fuel and radioactive waste exist in the United States as well as in West Germany. These standards apply to temporary storage. Standardization work is currently underway in these countries concerning the latter portion of the nuclear fuel cycle, including the final storage of high-level waste.

In the USA, the Nuclear Regulatory Commission (NRC) is pursuing a broadly-based programme aimed at establishing standards and licensing requirements with regard to the location, design and management of facilities for radioactive waste. This programme includes criteria aimed at ensuring the security of the environment and the safety of the personnel in the handling, transport, storage and final disposal of vitrified high-level waste. The initial results of this work are expected to be published during 1978. Work is also currently under way on design criteria for storage rooms for high-level waste. These criteria are expected to deal with the following points:

- Quality control requirements during design and construction
- Requirements on the ability of the various barriers to contain the waste
- Requirements on nuclear safety
- Compatibility between waste forms and containment media

- Protection against mechanical damage
- Security requirements

These pending licensing requirements in the area of nuclear waste have been given the working name 10 CFR 60 "Licensing of Radioactive Waste Management Facilities". More detailed design directives in this connection are expected to be presented by the NRC in Regulatory Guides.

A more thorough description of the situation concerning standardization work in the USA is provided in a KBS report /2-1/.

2.6 GROUNDS FOR EVALUATION WITH REGARD TO FINAL STORAGE

Safety criteria for final storage have not yet been established, but work is being pursued in this area in many different countries and in international cooperation. One of the requirements is that the long-term environmental load must be acceptable.

As regards the final repository, special attention must be paid to the ICRP's rule that no individual, either now or in the future, shall receive radiation doses which exceed the dose limits recommended by the ICRP. The current limit for individuals is 500 mrems/year from all activities which can give rise to radiation exposure with the exception of medical uses of ionizing radiation. On the basis of considerations of what is technically feasible and economically reasonable on the one hand and improved protection on the other, national regulations for nuclear power plants have been issued which stipulate 10-50 mrems/year for nearby residents.

Thus, the release of radioactive substances from a final repository shall not give rise to more than a fraction of 500 mrems/year and person to nearby residents for all future time. Beyond this, the usual rule that all measures which are socially and economically justified shall be adopted to reduce the dose load shall apply.

In order to protect large population groups against genetic effects in the long run, a rule concerning the limitation of collective doses similar to the one which is currently applicable to nuclear power plants should also be applied to waste and fuel management facilities.

In this connection, Sweden and the other Nordic countries are campaigning for the adoption of a rule which would specify a dose limit per unit of electrical output for nuclear power as a whole, namely 1 manrem/MWe per year /2-4/. Since 0.5 manrem/MWe per year has been allocated for the operation of nuclear power stations, 0.5 manrem/MWe remains for the other parts of the fuel cycle, including the final repository. All dose loads during the entire nuclear power era shall thereby be taken into consideration, whereby long-lived elements shall be totalled over 500 years. In the case of extremely longlived radioactive elements, the annual radiation doses shall remain low in relation to the natural radiation level. The choice of a level of 1 manrem/MWe is based on the goal of a maximum of 10 mrems/year and person and the assumption of an average global electrical power production from nuclear energy of 10 kW per person. This greatly exceeds the present total power consumption per inhabitant in the industrial countries and entails a large safety margin. As a comparison, it can be mentioned that electrical power consumption per person is highest in Norway, with an average of 2 kW per year (1975). The corresponding figure for Sweden is 1.1 kW.

2.7 MANKIND'S CURRENT RADIATION ENVIRONMENT

Radioactive elements exist in nature and ionizing radiation from these elements is responsible for a portion of the natural background radiation level. Additional irradiation of man stems from a number of different sources, for example from building materials in residences and from the medical use of radiation.

The background radiation occurring in nature comes from cosmic radiation, radiation from radioactive elements in the bedrock and radiation from radioactive elements absorbed in the body. The natural background radiation level in Sweden is between 70 and 140 mrems/year /2-5/. The body's natural absorption of potassium-40, uranium, thorium and radium along with their daughter products gives an average dose of 20 mrems/year in Sweden.

The radiation dose from building materials in buildings varies widely. Radiation doses between 20 and 200 mrads*/year in connection with uninterrupted presence indoors for the whole year are common /2-6/. Values up to 700 mrads/year have been measured as external doses /2-9/. The weighted internal whole-body dose caused by radon in our homes is between 10 and 1 000 mrems/year /2-10/. Medical irradiation provides an extra annual dose of approximately 40 mrems per person.

Most products in our environment, both natural and manufactured, are weakly radioactive. Drinking water in Sweden contains, for example, radium-226 at levels which vary between 0.1 and 40 pCi/l /2-7/. On the basis of the same calculating principles used for the final repository, this gives doses between 1 and 400 mrems/year. Water in nature also contains uranium at levels which are normally between 0.1 and 5 pCi/l, but extreme values of up to 1 500 pCi/l have been measured /2-8/.

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* The absorbed dose is given in rads, while the biologically weighted dose is given in rems. Except in the case of alpha radiation, the rad and rem values are numerically equivalent.

3 RADIOACTIVE ELEMENTS, RADIATION AND DECAY HEAT IN SPENT FUEL AND HIGH-LEVEL WASTE

3.1 WASTE QUANTITIES

The core of a light-water reactor contains uranium fuel in the form of uranium dioxide pellets encased in cladding tubes. The amount of uranium in the core varies with the size and type of the reactor. The amount of uranium in the core of the types of reactors used in Sweden is between 70 and 126 metric tons. The boiling water reactor (BWR) has a lower power density and a larger quantity of fuel at a given size than the pressurized water reactor (PWR). The fuel in the core is gradually replaced in connection with annual shutdowns for revision and fuel replacement. Each fuel element is irradiated for between 3 and 5 years before it reaches full burnup. This is 25 000 - 28 000 megawatt-days per ton of uranium (MWd(t)/tU) for BWRs and 31 000 - 35 000 MWd(t)/tU for PWRs.

The burnup of uranium fuel is associated with the production of radioactive waste by means of the following physical processes:

- fission of uranium and secondary plutonium.
- neutron capture. Neutron capture in the uranium leads to the formation of isotopes of elements which are heavier than uranium (transuranium elements). Continued neutron capture leads to the creation of increasingly heavy nuclides. The most important transuranium elements in spent fuel are neptunium (Np), plutonium (Pu), americium (Am) and curium (Cm). In reprocessing, most of the uranium (99.9%) and plutonium (99.5%) in the waste is separated while other transuranium elements remain in the waste.
- neutron capture in the cladding material, core components, etc., forming medium-level waste, which contains activation products.

The high-level waste obtained from reprocessing consists of approximately 35 kg of active and inactive fission products and heavy nuclides per ton of uranium. When this waste is vitrified, approximately 150 litres of glass are obtained per ton of uranium - i.e. about 0.5 mm^3 of glass per kWh electrical output or about 4 m^3 glass per year for 1 000 MW electrical output.

Tables 3-1 and 3-2 present examples of the composition of spent fuel and high-level waste. Table 3-3 shows the distribution of fission reactions among different nuclides. As is evident from the table, more than one-third of the energy is obtained from the fission of plutonium formed in the fuel.

Table 3-1. Composition of spent fuel upon discharge from reactor*.

	% by weight
Fission products	3.4
Uranium (of which 1% fissionable)	95.4
Plutonium (of which 3/4 fissionable)	1.1
Other heavy nuclides	0.1

Table 3-2. Composition of high-level waste immediately after reprocessing.

	% by weight
Fission products	93.4
Uranium	3.1
Plutonium	0.14
Other heavy nuclides	3.4

Table 3-3. Distribution of fission reactions among different nuclides. Average during irradiation period.

U 235	55 %	
U 238	8 %	(fast neutron fission)
Pu 239	32 %	
Pu 241	5 %	

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[*] In all tables except those for decay heat (3-7 and 3-10, the values apply for PWR fuel with a burnup of 33 000 MWd(t)/tU, power density 34.4 MW(t)/tU and enrichment 3.1% U 235. Reprocessing is assumed to have taken place after a 10-year cooling period. Separation efficiencies as per Table 3-8. Discharge is assumed to have taken place immediately after reactor shutdown.

3.2 RADIOACTIVE ELEMENTS IN SPENT FUEL

The radioactive elements can be divided into two main groups:

- fission products. A decay chain is formed for each mass number (see Fig. 3-2)
- the remaining uranium and the transuranium elements formed by neutron capture (see Fig. 3-1) and their daughter products. The decay chains for the heavy nuclides are shown in Fig. 3-3.

A more detailed account of radionuclide inventories is provided in /3-1/.

From the start, spent fuel contains several hundred different radioactive fission product nuclides, a hundred or so heavy nuclides and activation products. The latter are, however, of marginal importance. All radioactive nuclides decay with different half-

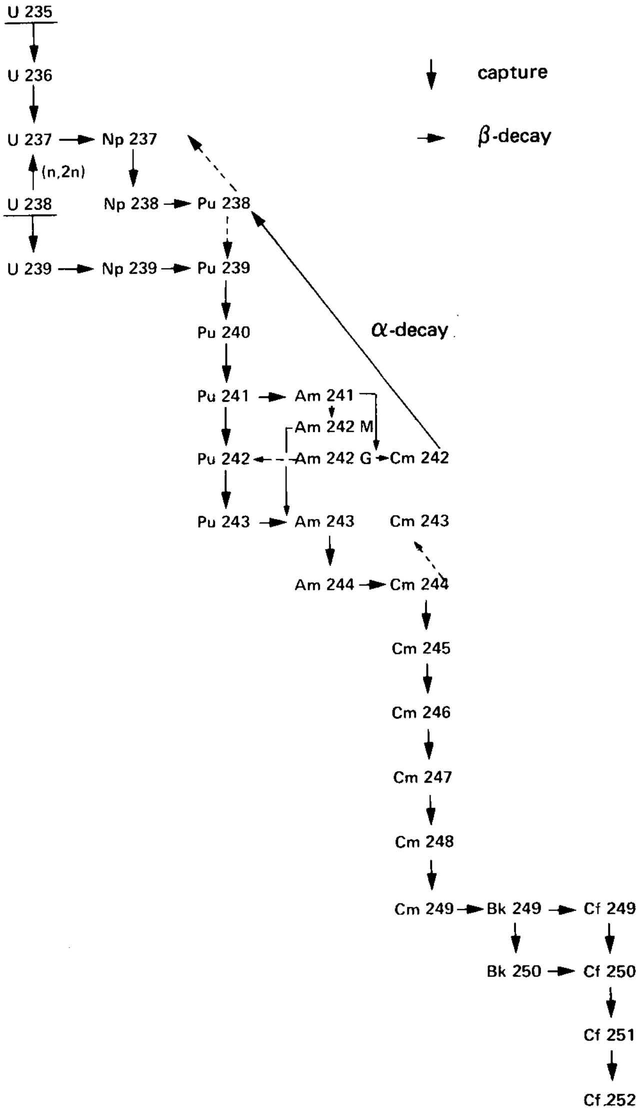

Figure 3-1. Formation of heavy nuclides. The figure illustrates how the heavy nuclides are formed during the irradiation period. The decay products created by slow alpha decay have not been included. They are given in figure 3-3.

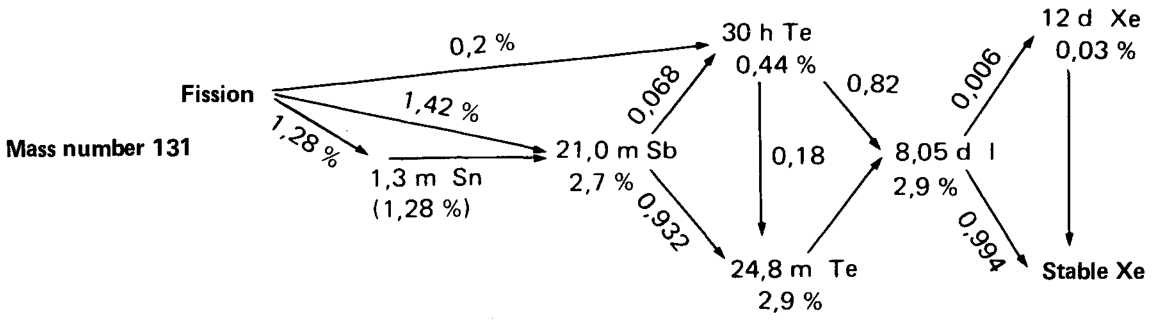

Figure 3-2. Fission product chain with mass number 131. Some radioactive isotopes are formed both by radioactive decay and as a direct result of nuclear fission. Decay proceeds to the stable end product xenon-131. The end product in a decay chain is always a stable isotope of some element.

lives, which leads to a substantial decrease in radioactivity with time. All short-lived nuclides decay during storage.

After the short-lived fission products with half-lives up to a few years have disintegrated, the following rough categorization of remaining radionuclides can be made on the basis of origin and half-life:

1. Medium-lived fission products;
   Mainly strontium-90 and cesium-137, with half-lives of around 30 years, and their daughters.
2. Long-lived fission products;
   The most interesting of these nuclides from the risk viewpoint is technetium-99 with a half-life of 210 000 years, and iodine-129, with a half-life of 17 million years, as well as cesium-135, with a half-life of 3 million years.
3. Heavy nuclides (transuranium elements, actinides);
   These nuclides exhibit a wide range of half-lives, but the most interesting are the long-lived nuclides which are formed in considerable quantity. These include, for example, neptunium-237, plutonium-239 and -240, americium-241 and -243 and the decay products radium-226 and thorium-229.
4. Other nuclides;
   These include tritium, with a half-life of 12 years, and carbon-14, with a half-life of 5 700 years. Carbon-14 originates primarily from impurities in the fuel and the cladding material /3-3/.

The decay of fission products with half-lives of several decades or more is shown in Tables 3-4 and 3-5. Table 3-6 shows the decay of heavy nuclides. All times are counted from discharge from the reactor.

The decay of the nuclides in groups 1-3 is depicted graphically in Figure 3-4.

Strontium-90 and cesium-137 dominate heavily during the first several hundred years, after which total activity is determined by

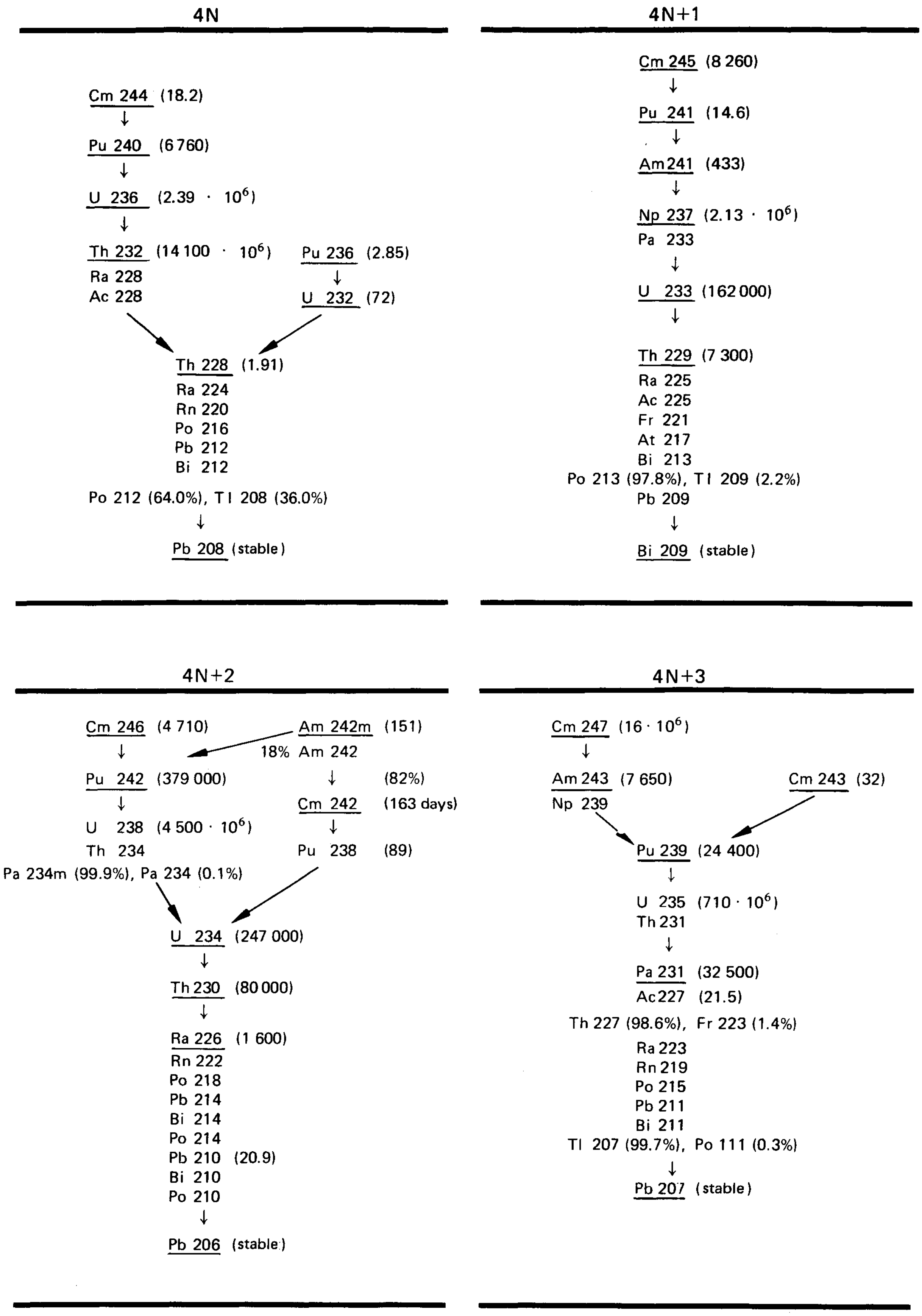

Figure 3-3. The decay chains of the heavy nuclides. Half-lives given within parentheses, in years. Half-lives shorter than 20 years have generally not been included. N is an integer. Nuclides in the chain 4N have atomic weights which are integral multiples of 4. All heavy nuclides are included in these decay chains. There is no connection between them.

Table 3-4. Important fission products in spent fuel and high-level waste during the period 10 - 1 000 years.

Nuclide	Half-life (years)	Radioactivity (Ci per ton uranium)				
		10 years	30 years	100 years	300 years	1 000 years
H 3*	12.3	420	140	2.6		
Se 79	65 000	0.39	0.39	0.39	0.39	0.39
Kr 85*	10.8	5 900	1 600	18		
Sr 90	28.1	60 000	36 000	6 500	46	
Zr 93** /Nb 93 m	$1.5 \cdot 10^6$	1.9	1.9	1.9	1.9	1.9
Tc 99	210 000	14	14	14	14	14
Pd 107	$7 \cdot 10^6$	0.12	0.12	0.12	0.12	0.12
Sn 126** /Sb 126 m /Sb 126	100 000	0.57	0.57	0.57	0.57	0.56
I 129*	$17 \cdot 10^6$	0.038	0.038	0.038	0.038	0.038
Cs 135	$3 \cdot 10^6$	0.25	0.25	0.25	0.25	0.25
Cs 137** /Ba 137 m	30.0	86 000	54 000	11 000	106	
Pm 147	2.62	7 700	39			
Sm 151	87	1 200	1 000	570	120	0.44
Eu 154	16	4 800	2 000	96	0.017	
Total spent fuel		320 000	180 000	34 000	440	21
Total high-level waste		310 000	180 000	34 000	440	21

* H, Kr and I are removed during reprocessing.

** Radioactive equilibrium. The radioactivity specification applies to the parent nuclide alone.

Table 3-5. Long-lived fission products in spent fuel and high-level waste.

Nuclide	Half-life (years)	Radioactivity (Ci per ton uranium) after						
		10 years	100 years	1 000 years	10 000 years	100 000 years	1 mill. years	10 mill. years
C 14*	5 735	1.5 (0.10)	1.5 (0.10)	1.3 (0.09)	0.4 (0.03)			
Se 79	65 000	0.39	0.39	0.39	0.36	0.14		
Zr 93 /Nb 93 m	$1.5 \cdot 10^6$	1.9	1.9	1.9	1.9	1.8	1.2	0.018
Tc 99	210 000	14	14	14	14	10	0.54	
Pd 107	$7 \cdot 10^6$	0.12	0.12	0.12	0.12	0.12	0.11	0.044
Sn 126 /Sb 126 m /Sb 126	100 000	0.57	0.57	0.56	0.53	0.28	0.0005	
I 129**	$17 \cdot 10^6$	0.038	0.038	0.038	0.038	0.038	0.036	0.025
Cs 135	$3 \cdot 10^6$	0.25	0.25	0.25	0.25	0.25	0.20	0.025
Total spent fuel		320 000	34 000	22	20	15	3.2	0.13
Total high-level waste		310 000	34 000	21	20	15	3.2	0.10

* Radioactivity from C14 in the cladding material has been included for C14. Values for high-level waste within parentheses.

** Not present in high-level waste.

Table 3-6. Heavy nuclides in spent fuel.

Nuclide	Half-life (years)	Radioactivity (Ci per ton uranium) after						
		10 years	100 years	1 000 years	10 000 years	100 000 years	1 mill. years	10 mill. years
Cm 246	4 710	0.092	0.091	0.079	0.021			
Cm 245	8260	0.43	0.43	0.40	0.19	0.0001		
Cm 244	18.2	2 000	65					
Am 243 /Np 239	7 650	21	21	19	8.5	0.0024		
Am 241	433	1 500	3 300	790	0.19	0.0001		
Pu 242	379 000	1.5	1.5	1.5	1.5	1.3	0.25	
Pu 241	14.6	68 000	960	0.40	0.19	0.0001		
Pu 240	6 760	490	490	450	180	0.018		
Pu 239	24 400	320	320	320	250	20		
Pu 238	89	2 800	1 400	1.4				
Np 237 /Pa 233	2.13 · 10^6	0.34	0.43	0.97	1.1	1.1	0.80	0.043
U 238 /Th 234 /Pa 234 m	4 510 · 10^6	0.31	0.31	0.31	0.31	0.31	0.31	0.31
U 236	23.9 · 10^6	0.27	0.27	0.28	0.36	0.41	0.40	0.30
U 235 /Th 231	710 · 10^6	0.015	0.015	0.016	0.018	0.026	0.026	0.026
U 234	247 000	0.75	1.2	1.8	1.7	1.4	0.40	0.31
U 233	162 000	0.0001	0.0002	0.0032	0.045	0.38	0.85	0.043
Pa 231 /Ac 227 /Ra 223 etc	32 500		0.0001	0.0003	0.0031	0.021	0.026	0.026
Th 230	80 000	0.0001	0.0008	0.014	0.14	0.90	0.44	0.31
Th 229 /Ra 225 /Ac 225 etc	7 300			0.0001	0.016	0.38	0.85	0.043
Ra 226 /Pb 210 /Po 210 via ett antal nuklider	1 600			0.0025	0.11	0.90	0.44	0.31
Total		76 000	6 600	1 600	450	38	16	5.5

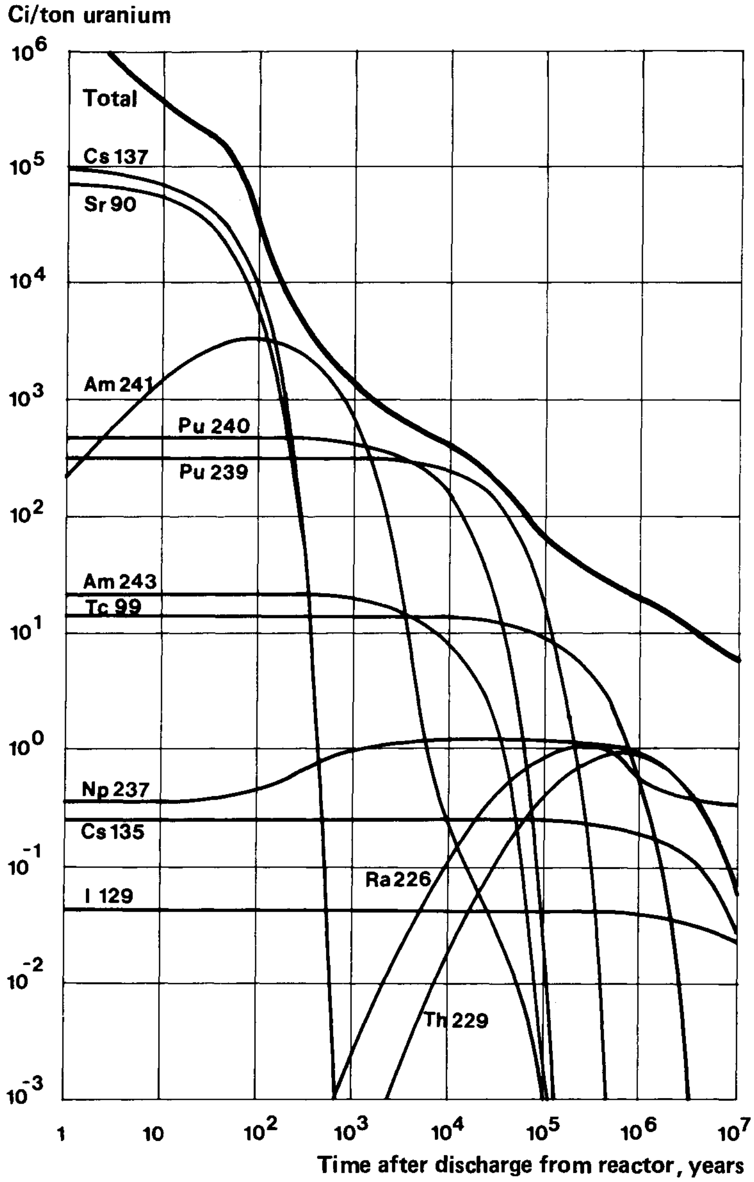

Figure 3-4. Radioactive elements in spent fuel. The graph shows the radioactive elements in PWR fuel with a burnup of 33 000 MWd (t)/tU, power density 34.4 MW (t)/tU and enrichment 3.1 % uranium-235.

the heavy nuclides. After 300 years, approximately 10 half-lives, the activity of strontium-90 and cesium-137 has dropped to one-thousandth of its original value, and after an additional 300 years, activity is one-millionth of its original value - i.e. approximately 0.1 Ci/tU. After about 150 years, however, americium-241 begins to dominate the total activity, and is later surpassed by plutonium. After 100 000 years, the picture becomes more complex, but radioactivity decreases constantly and reaches 19 Ci/tU after 3 million years, which is 1/100 000th of the activity level after three years.

3.3 DECAY HEAT IN SPENT FUEL

The power density of the fuel during reactor operation is between 20 and 40 MW(t)/tU. When the reactor is shut down and the spent fuel is discharged, it still gives off some heat - known as decay heat - which stems from the decay of secondary radionuclides. But decay heat is low compared to thermal power during operation and decreases with time. One minute after shutdown, decay heat is only 5% of operating power and continues to decline rapidly. Table 3-7 and Figs. 3-5 and 3-6 give the decay heat in PWR and BWR fuel after various cooling times longer than 1 year /3-2/.

Table 3-7. Decay heat, spent fuel*
(FP = fission products, HN = heavy nuclides)

	Thermal power (W per ton uranium)					
	P W R			B W R		
Time (years) after discharge)	FP	HN	Total	FP	HN	Total
1	11 000	630	12 000	7 200	400	7 600
2	5 900	340	6 200	3 900	190	4 100
5	1 900	260	2 200	1 400	150	1 600
10	1 000	280	1 300	840	160	1 000
20	730	290	1 000	600	170	770
50	350	280	630	290	170	460
100	100	240	340	88	150	240
200	10	180	190	8.4	120	130
500	0.073	110	110	0.059	73	73
1 000	0.022	62	62	0.018	43	43
2 000	0.022	31	31	0.018	24	24
5 000	0.022	19	19	0.018	15	15
10 000	0.022	14	14	0.018	11	11

* The following apply in Tables 3-7 and 3-10:
PWR: 33 000 MWd(t)/tU, 38.5 MW(t)/tU, 3.25 % U 235
BWR: 27 600 MWd(t)/tU, 22.0 MW(t)/tU, 2.75 % U 235

Strontium-90 and cesium-137 dominate in the beginning with regard to decay heat as well. The heavy nuclides take over after about 100 years. Decay heat decreases by roughly 2 orders of magnitude between 1 and 1 000 years after discharge from the reactor in unreprocessed fuel.

The generation of decay heat leads to increases in temperature in the encapsulation material and in the surrounding rock after final disposal of the waste.

3.4 FRACTIONATION IN CONNECTION WITH REPROCESSING

The radioactive elements in the spent fuel are separated into different fractions in connection with reprocessing /3-4/.

The most important fractions are

- most of the fission products, which constitute high-level waste
- the uranium fraction
- the plutonium fraction

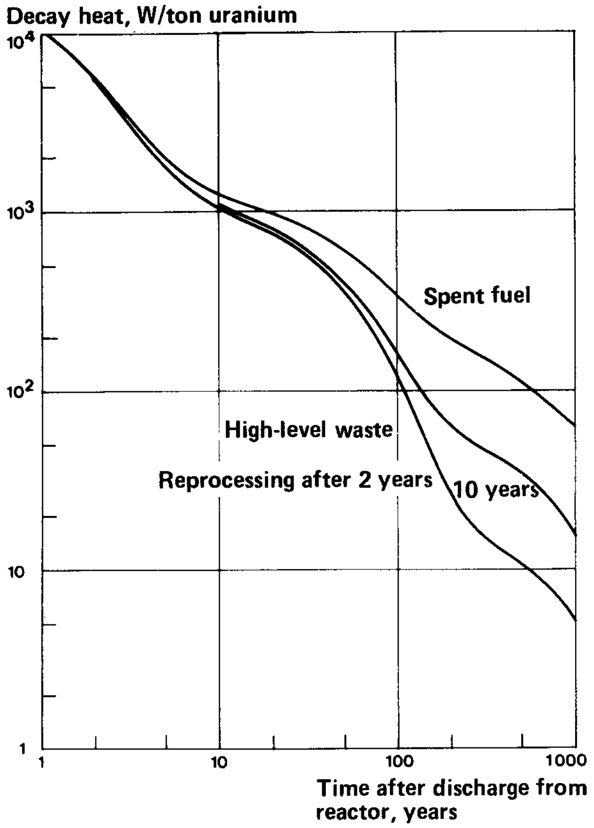

Figure 3-5. Decay heat in spent fuel and high-level waste from PWR. In PWR fuel, burnup is 33 000 MWd (t)/tU, power density is 38.5 MW (t)/tU and enrichment is 3.25 % uranium-235. Separation efficiencies are as per table 3-8, with the exception of uranium, of which it has been assumed that 99.5 % has been removed.

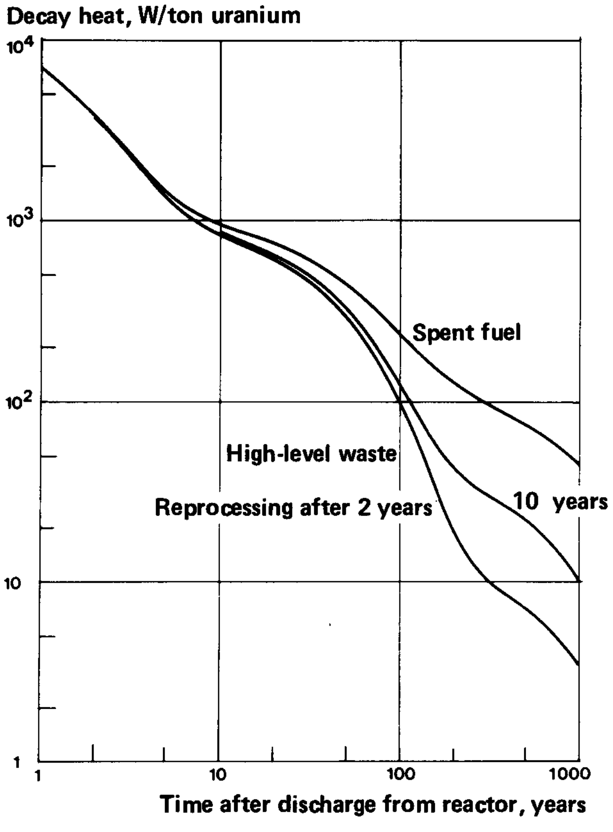

Figure 3-6. Decay heat in spent fuel and high-level waste from BWR. In BWR fuel, burnup is 27 600 MWd (t)/tU, power density is 22.0 MW (t)/tU and enrichment is 2.75 % uranium 235. Separation efficiency as per table 3-8 with the exception of uranium, of which it has been assumed that 99.5 % was removed.

The main stages in reprocessing are as follows:

- Chopping of the spent fuel rods and dissolution of the fuel pellets in nitric acid.
- Separation of uranium and plutonium from the nitric acid solution by extraction with organic complexers in kerosene.
- Separation of plutonium from uranium.

After reprocessing, the waste solution is calcined and the calcinate is melted together with vitrifying agents to form glass. Reprocessing and vitrification are dealt with in section III:4.

In connection with chopping and dissolution of the spent fuel rods, the remaining noble gas activity, which is composed almost solely of krypton-85, is evolved. Carbon-14 /3-3/ and most of the iodine-129 /3-5/ are also released. The approximately 0.1% iodine which remains in the waste solution should be evolved during calcination /3-5/, when a small portion of the ruthenium and cesium will also be released. The tritium formed in the spent fuel is also evolved here.

Uranium and plutonium are extracted from the waste with 30% tributyl phosphate in kerosene in a multi-stage process. Uranium is easier to extract than plutonium and losses to the waste are 0.1% uranium and 0.5% plutonium /3-4/. These values have been assumed in the calculations of the quantity of heavy nuclides in the waste glass. Information recently received from France puts losses to the high-level waste at less than 0.2% uranium and 0.15% plutonium. These values are said to be the best estimate currently available. The higher value for uranium entails an increase of the amounts of uranium-234 and radium-226 in the waste by a few percent. Most of these nuclides come from the decay of plutonium-238. The lower value for plutonium entails considerable reductions in the levels of many heavy nuclides in the waste.

The assumed losses of 0.5% plutonium and 0.1% uranium to the waste give higher calculated levels of all of the heavy nuclides in the waste glass which are of importance from the viewpoint of safety than what can be expected on the basis of available experiences from reprocessing.

The calculation of decay heat /3-2/ is based on the assumption of 0.5% uranium and 0.5% plutonium in the waste.

In connection with the extraction of uranium and plutonium, certain amounts of neptunium can be extracted along with these elements, but other heavy elements remain in the waste solution. It is assumed here that all heavy nuclides of elements other than uranium and plutonium are present in the waste glass.

The separation efficiencies which are used in the safety analysis are presented in Table 3-8. Note, however, that the consequence analysis in Chapter 6.9 assumes that 1% iodine-129 remains in the waste glass.

In vitrification, the waste oxides are mixed with glassforming oxides and the mixture is melted to glass. In this way, the waste becomes a part of the glass, so that the radionuclides are bound in a form which is resistant to leaching.

Table 3-8. Separation efficiencies in reprocessing.

Tritium	100 %
Carbon-14	90 - 95 %*
Noble gases (Kr-85)	100 %
Halogen gases (I-129)	100 %
Uranium	99.9 %
Plutonium	99.5 %

* of which approx. 70% is present in the undissolved cladding material.

3.5 RADIOACTIVE ELEMENTS IN HIGH-LEVEL WASTE

The high-level waste contains most of the fission products (with the exception of tritium, krypton and iodine), the residual uranium and the residual plutonium as well as most of the other heavy nuclides.

The most important fission products in spent fuel were listed in Tables 3-4 and 3-5, which also give the total activity of high-level waste. Table 3-9 presents the most important heavy nuclides in high-level waste for the case of reprocessing after a cooling period of 10 years. In the beginning, strontium-90 and cesium-137 dominate, just as in the case of spent fuel, but then the reduced actinide content leads to more rapid cooling of the high-level waste.

During the period from 300 to 3 000 years after discharge, americium-241 occupies a dominant position. After this, americium-243, technetium-99, neptunium-237 and thorium-229 are of primary interest. Many others contribute, but only marginally.

Since most of the plutonium and uranium in the fuel is separated during reprocessing, their daughter nuclides are of less importance in the high-level waste than in unreprocessed spent fuel. However, their levels are affected by the length of the period between discharge and reprocessing. This is especially true of the nuclides which are daughter products of plutonium-241, with a half-life of 14.6 years, i.e. americium-241, neptunium-237, uranium-233 and thorium-229. (See Figure 3-3, chain 4N+1.) This is illustrated by the following table, which gives the maximum radioactivity (at a given point in time) in the waste glass for different numbers of years between discharge and reprocessing.

Number of years	Max. radioactivity, curies per ton U or per waste cylinder		
	Pu241	Am-241	Np-237
1	530	250	0.39
3	470	570	0.45
10	340	1 470	0.64
Max		3 310	1.13
Fuel	110 000	250	0.34

Table 3-9. Heavy nuclides in high-level waste.

Nuclide	Half-life (years)	Radioactivity (Ci per ton uranium) after						
		10 years	100 years	1 000 years	10 000 years	100 000 years	1 mill. years	10 mill. years
Cm 246	4 710	0.092	0.091	0.079	0.021			
Cm 245	8 260	0.43	0.43	0.40	0.19	0.0001		
Cm 244	18.2	2 000	65					
Am 243 /Np 239	7 650	21	21	19	8.5	0.0024		
Am 241	433	1 500	1 300	300	0.19	0.0001		
Pu 242	379 000	0.0076	0.0078	0.0084	0.0089	0.0078	0.0015	
Pu 241	14.6	340	5.2	0.40	0.19	0.0001		
Pu 240	6 760	2.5	7.7	7.2	2.9	0.0003		
Pu 239	24 400	1.6	1.7	2.1	4.6	0.66		
Pu 238	89	14	9.5	0.17				
Np 237 /Pa 233	$2.13 \cdot 10^6$	0.34	0.38	0.58	0.64	0.62	0.46	0.025
U 238 /Th 234 /Pa 234 m	$4\ 510 \cdot 10^6$	0.0003	0.0003	0.0003	0.0003	0.0003	0.0003	0.0003
U 235 /Th 231	$710 \cdot 10^6$					0.0004	0.0004	0.0004
U 234	247 000	0.0007	0.0037	0.0096	0.0095	0.0075	0.0008	0.0003
U 233	162 000		0.0001	0.0021	0.026	0.22	0.49	0.025
Pa 231 /Ac 227 /Ra 223 etc	32 500					0.0003	0.0004	0.0004
Th 230	80 000	0.0001	0.0001	0.0001	0.0008	0.0049	0.0012	0.0003
Th 229 /Ra 225 /Ac 225 etc	7 300			0.0001	0.0090	0.22	0.49	0.025
Ra 226 /Pb 210 /Po 210 via a number of nuclides	1 600				0.0007	0.0049	0.0012	0.0003
Total		3 900	1 400	350	26	3.9	5.4	0.29

The fuel line gives radioactivity upon discharge from reactor. The maximum values for neptunium-237 occur after several thousand years. A period of 10 years between discharge and reprocessing has been assumed throughout the safety analysis. Figure 3-7 shows the radionuclide content of the waste for this case. The number of curies per ton of uranium refers to the uranium in the fuel and is approximately equal to the number or curies per waste cylinder.

3.6 DECAY HEAT IN HIGH-LEVEL WASTE

Table 3-10 gives the decay heats in high-level waste. The values for both BWRs and PWRs are reported, and the values have also been broken down into fission products and heavy elements.

During the period from 50 years to 1 000 years, decay heat decreases from between 340 and 430 W per ton uranium to between 10

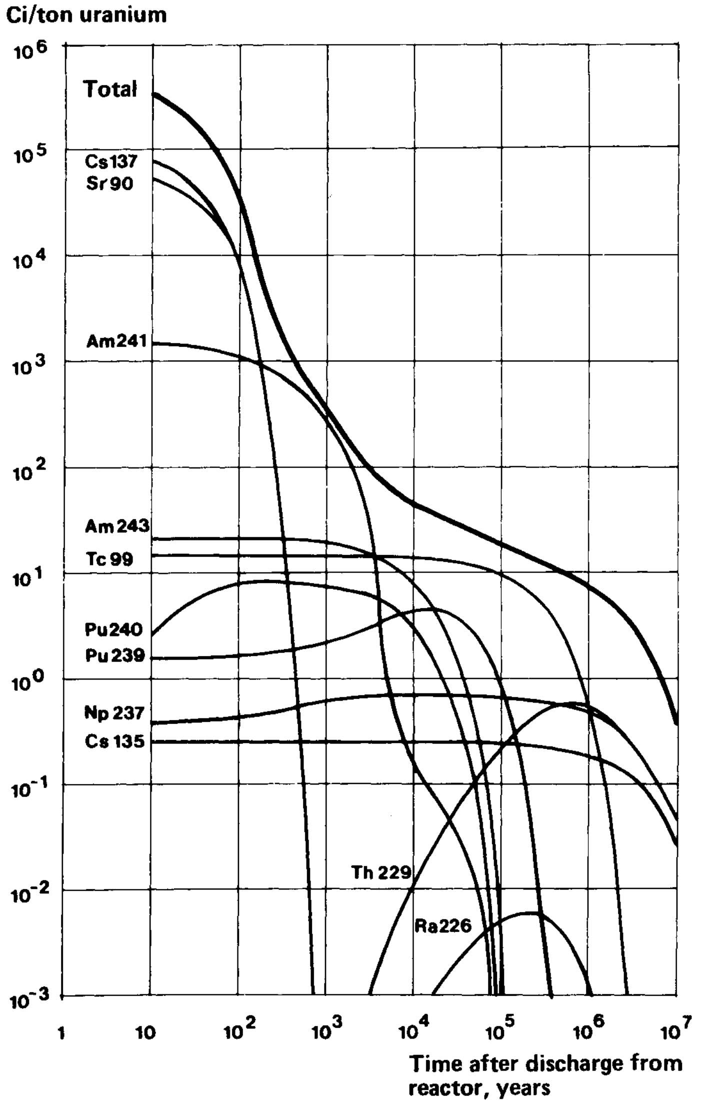

Figure 3-7. Radioactive elements in high-level waste. It is assumed that reprocessing takes place 10 years after discharge of the spent fuel from the reactor. Separation efficiency in reprocessing as per table 3-8. The fuel has a burnup of 33 000 MWd (t)/tU, a power density of 34.4 MW (t)/tU and an enrichment of 3.1 % uranium-235.

Table 3-10 Decay heat in high-level waste*
(FP = fission products, HN = heavy nuclides)

	Thermal power (W per ton uranium)					
	PWR			BWR		
Time (years) after discharge	FP	HN	Total	FP	HN	Total
10	1 200	130	1 100	830	71	900
20	730	110	840	600	62	660
50	350	83	430	290	49	340
100	100	67	170	88	42	130
200	10	55	65	8.4	35	43
500	0.073	34	34	0.059	22	22
1 000	0.022	16	16	0.018	9.9	9.9

* Reprocessing after 10 years of cooling. The separation efficiency for uranium here is 99.5%, otherwise as per Table 3-8.

and 16 W/ton, depending on the relative proportions of residual products from PWR and BWR fuel. The decay heat in high-level waste from PWRs is in the upper part of the range, owing to the fact that burnup is greater in PWRs than in BWRs.

In the beginning, decay heat generation is completely dominated by the fission products. After 100-200 years, the two fractions make equal contributions. Decay heat generation is then increasingly dominated by the heavy elements.

Figures 3-5 and 3-6 illustrate decay heat generation in spent fuel and vitrified waste. The sharp decrease with time for both spent fuel and vitrified waste is noteworthy. The differences between spent fuel and vitrified waste after 1 000 years are no more than one order of magnitude, depending on when reprocessing takes place.

3.7 RELATIVE TOXICITY OF THE RADIOACTIVE ELEMENTS

The toxicity of the radioactive elements is not solely dependent upon the level of activity expressed in e.g. curies, which is only a measure of how many disintegrations take place per unit of time.

Toxicity is also dependent upon:

- type of radiation (α-, β- or γ-radiation)
- radiant energy
- path of intake (inhalation or food) and assimilation in the body
- distribution among and within different organs
- secretion from the different organs in the body.

The relative toxicity of different radioactive elements when taken internally can be assessed on the basis of the dose commitment in rems which results from the unit intake of one Ci. Table 6-2 gives the dose commitment for various organs for the intake via food (or drinking water) of 1 Ci of several relevant nuclides.

Since different organs possess varying sensitivity to radiation, the dose to the organs can be standardized to a weighted whole-body dose using the weighting factors given in Table 6-3. The resultant relative toxicity for unit intake is given in Table 6-2 in rems (standardized whole-body dose) per Ci.

Hazard index

If the activity quantities for various nuclides per ton of waste are multiplied by the relative toxicity, a type of hazard index is obtained which takes into account both quantity of activity and relative toxicity. This index is a rough indication of the hazard potential of various elements.

Such an index does not, however, take into account the probability that a person will be exposed to these elements or the concentrations at which the elements occur. This requires data on the following dispersal factors:

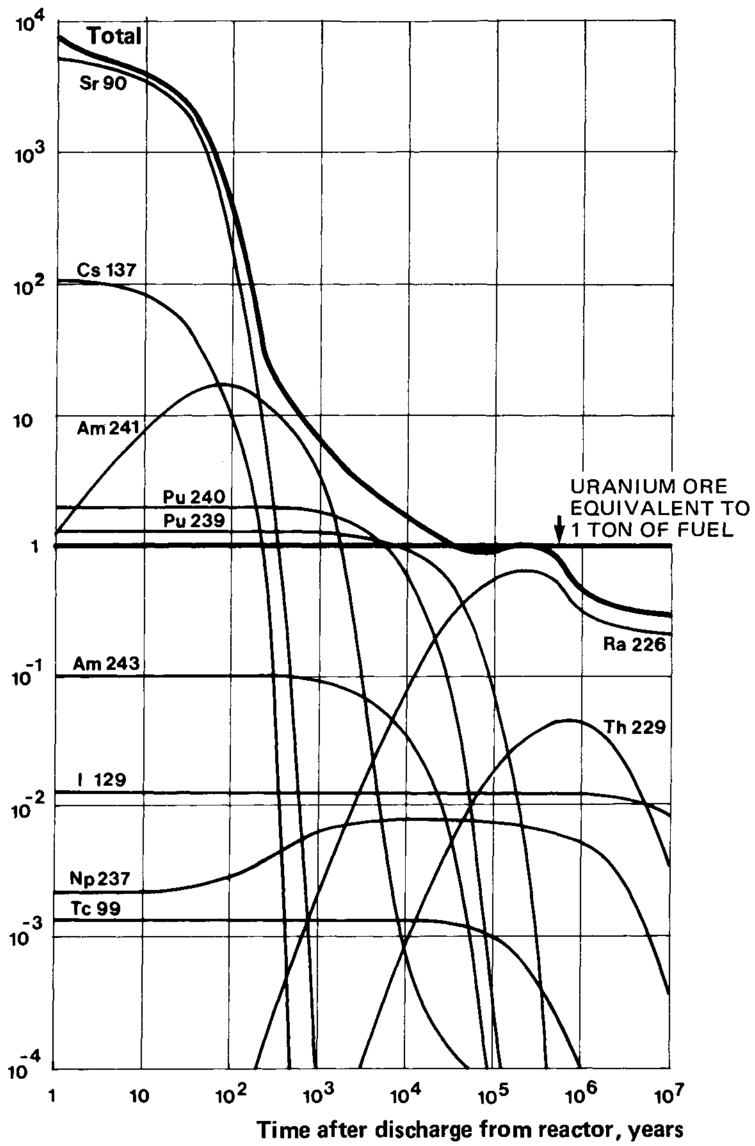

Figure 3-8. Potential hazard index for different nuclides in 1 ton of spent fuel in relation to the quantity of uranium ore which is equivalent to 1 ton of fuel. Actual hazard is also dependent upon the exposure pathways of the elements to man (see chapter 6). The fuel has a burnup of 33 000 MWd (t)/tU, a power density of 34.4 MW (t)/tU and an enrichment of 3.1 % uranium 235.

- The release of radioactive elements from the final repository.
- The transport of the elements through the filler material and cracks in the rock.
- The disintegration of the elements and the formation of daughter nuclides in the final repository and during transport.
- Concentration and dilution factors in the groundwater and in the biosphere.
- Paths of exposure to man.
- The population's consumption of water and food which may have been contaminated by radioactive elements from the final repository.

Chapter 6 presents an analysis of final storage on the basis of the above factors.

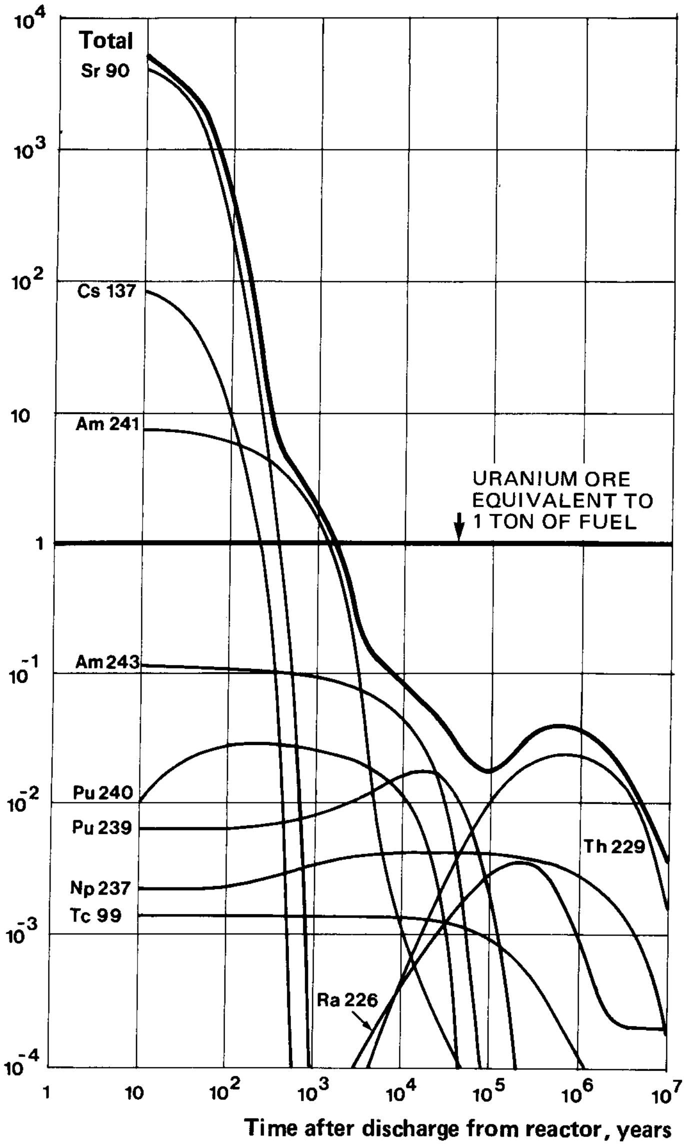

Figure 3-9. Potential hazard index for different nuclides in high-level waste from 1 ton of spent fuel in relation to the quantity of uranium ore which is equivalent to 1 ton of fuel. Actual hazard is also dependent upon the exposure pathways of the elements to man (see chapter 6). The fuel has a burnup of 33 000 MWd (t)/tU, a power density of 34.4 MW (t)/tU and an enrichment of 3.1 % uranium 235.

Hazard index for spent fuel and high-level waste

Since the hazard index has received so much publicity, owing to the fact that it is easy to calculate, potential hazard indices for spent fuel and high-level waste are given in Figures 3-8 and 3-9. The scales are related to the quantity of uranium ore which corresponds to one ton of fuel.

As in previous publications, the maximum permissible levels in water given in the RCG* have been used as a measure of relative toxicity instead of the weighted whole-body dose from unit intake, but the difference is often negligible and should not be very great in any case.

Strontium-90 dominates the hazard index for spent fuel during the first 300 years. Then comes a period of 50 000 years during which americium-241 and plutonium-239 and -240 dominate. The subsequent period is dominated by radium-226.

The hazard index of the vitrified waste is also determined by strontium-90 during the first 300 years. The hazard index then drops more rapidly than in the case of spent fuel, since most of the plutonium isotopes are separated during reprocessing. Between 300 and 50 000 years, the hazard index is dominated by americium-241 and -243 as well as plutonium-239, after which thorium-229 takes over.

The hazard index for uranium ore, which is used as a reference, is dominated by radium-226, which is a fourth level daughter nuclide of uranium-238 and is in equilibrium with uranium-238. (See Figure 3-3, chain 4N+2.)

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* Radioactivity Concentration Guide. For the most part, the values are equal to one-tenth of the ICRP limits for 168 hours and critical organ. The RCG values are in the library for the ORIGEN computer program so that the program can be used for direct calculation of relative toxicities.

4 SAFETY IN HANDLING, STORAGE AND TRANSPORTATION

4.1 INTRODUCTION

This chapter analyzes safety in the handling, intermediate storage and transportation of spent fuel and vitrified high level waste. A more detailed description is provided in a KBS technical report /4-1/.

The following handling stages are dealt with (see Figure 4-1):

- Transportation of spent fuel elements from the reactor station to a central storage facility.
- Reception and storage in the central storage facility for about 10 years.

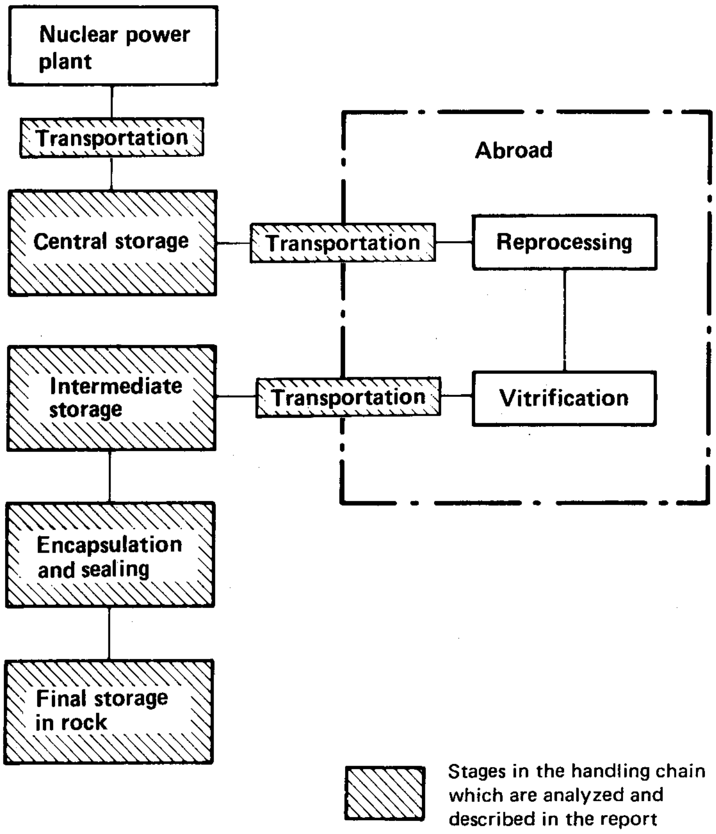


Figure 4-1. Handling chain for spent fuel and vitrified waste.

- Discharge of spent fuel from the central storage facility and transportation to a foreign plant for reprocessing, where the high-level waste is vitrified.
- Transportation of vitrified waste back to domestic intermediate storage facility.
- Storage of waste for 30 years in intermediate storage facility.
- Encapsulation of vitrified waste in canisters of titanium and lead.
- Deposition of the encapsulated waste in rock repositories approx. 500 m down in the bedrock.

Safety in connection with final storage is dealt with in other chapters.

The following is reported for each handling stage:

- A brief technical resumé of processes and handling procedures.
- An account of various safety measures.
- An evaluation of normal operating releases.
- A failure analysis of accidents which could happen despite safety measures.

Personnel and the environment may be subjected to radiation in connection with both normal operation and incidents. Shielding and other normal radiation protection measures shall be introduced and enforced to such an extent that radiation doses are limited in compliance with the recommendation and requirements of the ICRP and the National Institute of Radiation Protection. Personnel protection and safety is dealt with in Chapter III:7.

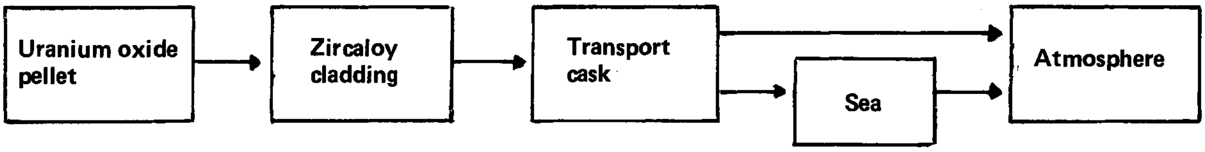
Adverse environmental impact stemming from accidents at specified plants and in connection with transportation between such plants is judged to be much less serious than in the case of nuclear power plants, in terms of both probability and consequences. This is due to the fact that the chances of a sudden and heavy release of radioactive substances are nil or negligible. Among other things, pressure and temperature are lower. Furthermore, such events develop much more slowly and allow more time for countermeasures.

Normal releases in the treatment stages are considerably lower than for nuclear power plants and reprocessing plants. This is because the temperatures are lower and the radioactive substances are continuously enclosed and are not subjected to any processing.

4.2 METHOD FOR FAILURE ANALYSIS

The method which is used for failure analysis has been called the barrier model. It is based on the fact that in order for radioactivity to be released to the environment, a series of barriers must be penetrated.

In their normal condition, certain barriers can be virtually impenetrable to the enclosed radioactive substances and their radiation, while other barriers may permit some penetration even in their normal condition. This aspect is illustrated in the following figure, which depicts the barriers against atmospheric dispersal in the sea transport of spent fuel.



The uranium dioxide pellets constitute a barrier to the dispersal of fission products and transuranium elements in the ceramic material. However, there can be a dispersal of "gap activity" from radioactive substances which have diffused out of the fuel and accumulated in the gap between the pellets and the fuel cladding.

During its service in the reactor, the zircaloy cladding which encases the fuel pellets may have sustained some damage, mainly in the form of tiny pin-holes. It has been assumed in this study that 0.2% of the fuel rods have cladding damage, a value which is relatively high compared to experiences from actual reactor operation.

Current international standards relating to type B packaging for the transportation of nuclear and radioactive material impose very strict requirements on the imperviousness of the casks /4-2/. Obviously, the integrity of the transport cask must somehow be violated in order for a release to occur through the above barriers. If the transport cask is damaged, however, the uranium dioxide pellets and the zircaloy cladding do not offer complete protection against a dispersal of radioactivity. How much radioactivity will be dispersed in connection with an accident depends on the extent of the damages to the barriers involved and on the physical process of dispersal.

A normal condition and a number of accident conditions have been defined for each barrier in order to provide as good a description as possible of the release of radioactivity in connection with accidents.

In the actual failure analysis, interest is focused on events which can damage one or more barriers. This analysis provides a systematic review of routine operations, handling equipment and plant design features. In this manner, a list is obtained of possible improper actions and equipment failures.

The basis for the calculations has been 30 years of operation of 13 reactors. This entails an annual discharge of spent fuel corresponding to 300 tons of uranium. It has been assumed that the fuel has a burnup of 33 000 MWd/ton U, which corresponds to the maximum burnup obtained for PWR fuel. For the sake of simplicity, a burnup of 33 000 MWd/ton U has been used in some cases for BWR fuel as well. These values are higher than the average burnup, leading to some over-estimation of the available quantity of radioactive substances in the calculations. Such conservative assumptions have been made throughout the safety analysis so that the specified released quantities and consequences can be regarded as upper limits and not as reasonable average values.

The failure analysis reports the quantities of radioactivity which are available for release in the event of different accidents and failures and the probability of such releases actually occurring.

The goal has been to base estimates of probability on available statistics, whereever possible. In cases where available statistical material has been unsatisfactory, a conservative value has been assigned to the failure probability.

In order to obtain a sensitive measure of the radiological consequences of failures and accidents, two quantities have been used: individual dose and collective dose commitment.

Three paths of exposure have been considered with respect to atmospheric dispersal:

- External irradiation by passing airborne activity.
- Internal irradiation by inhaled airborne activity.
- External irradiation by ground deposited activity for six hours (after this period of time, it is assumed that the most highly exposed population group will have been evacuated).

4.3 CENTRAL STORAGE FACILITY FOR SPENT FUEL

4.3.1 Technical resumé

It has been assumed in the study that the central storage facility /4-3/ will be situated in a rock cavern with a rock cover of about 30 m. It consists of a receiving section, a storage section and auxiliary systems. Spent fuel is transported up and down via ascent and descent tunnels. Water, electricity, ventilation, compressed air and other services are supplied through vertical shafts.

The transport cask is carried on a trailer from the harbour to the receiving section. After being unloaded from the trailer, the cask is placed in a holding pen, where the shock absorber is removed and the cask is raised. After sampling of its water, the cask is connected to a cooling and cleaning system, its temperature is reduced and the activity in the water is absorbed in ion exchangers and filters. The cask is then moved to the receiving pool. The fuel is lifted out of the cask and placed in cassettes. The filled cassettes are moved from the receiving section via water-filled conveyor channels to the storage section.

The storage section consists of a number of pools where the fuel is stored in its cassettes or in permanent racks of similar design. In this study, cassettes which hold either 25 BWR or 9 PWR fuel elements have been assumed.

The auxiliary systems include cooling and cleaning systems for transport casks, receiving and storage pools and systems for decontamination and disposal of spillage water and filter materials.

4.3.2 Safety measures

The facility is designed with a special emphasis on:

- minimizing the radiation doses to the personnel
- preventing the release of radioactive substances which could be dispersed to the environment

The radiological safety of the personnel is guaranteed by a number of safety measures, such as:

- Amply dimensioned radiation shields in the form of e.g. concrete walls and water barriers.
- Monitoring instruments for checking direct radiation and airborne activity.
- Remote control of radioactive components and systems.
- Dose monitoring of personnel.

The storage pools are located underground with a 30 m rock cover, which provides the stored fuel with an effective barrier against external forces.

The ventilation systems are sized to guarantee acceptabel temperature and humidity. A special ventilation system serves the unloading cell and other areas where there is a risk of airborne activity and thereby keeps them at a lower pressure than the surrounding parts of the building. Radioactive substances in the special ventilation system are separated in filters before the air is mixed with the exhaust ventilation air.

The storage pools are designed as free-standing thick-walled units lined with stainless steel. The external walls of the concrete structure are accessible for inspection and any leakage can be diverted via collection ducts behind the welded joints in the stainless steel plate and collected in a drainage system. This permits leakage to be detected at an early stage.

The pools do not have any low-level pipe connections. This prevents inadvertent emptying of the pools and guarantees water coverage of the fuel. The cooling system is designed to maintain a water temperature of $25-30^{o}C$ normally and below $60^{o}C$ in the event of isolated component failures. If external cooling should fail completely, despite redundant components and a back-up power system, the temperature of the pool water will rise (about $10^{o}C$ per day when the pool is filled). In the meantime, steps can be taken to restore cooling. If no countermeasures are adopted it will take more than a week for the pools to reach the boiling point. However, the fuel can be kept covered with water even after boiling, so the fuel will not be damaged due to exposure.

The configuration of the fuel in the tanks has been analyzed for criticality. The calculations show a good safety margin to criticality, even for unirradiated fuel ($K_{eff} < 0.95$). It is improbable that the fuel will be redistributed due to an accident to a configuration which possesses higher reactivity than the normal storage configuration. The fact that the fuel in the central storage facility is spent further reduces the possibility of a critical configuration.

In order to minimize the fire hazard, the facility is divided into fire cells and equipped with automatic fire alarm, fire ventilation and fire extinguishing systems adapted to the nature of the different areas. The fire load is low throughout.

In order to eliminate the risk of rock fall, comprehensive geological and rock mechanical surveys will be conducted prior to the start of construction and the quality of the rock will be checked continuously during the construction period. In addition to systematic bolting of the roof and walls of the rock cavern, reinforced concrete can be used for added strengthening where required.

The 70-110 ton cask for spent fuel is transported from the harbour to the receiving section on a trailer towed by a tractor vehicle. In order to reduce the risk of accidents in transit, transport speed has been limited to 10 km/h. The grade in the tunnel is a maximum of 1:10 and long straightaways have been avoided.

The transport cask is moved horizontally within the receiving section by means of an overhead crane over a reinforced floor section. Where vertical movements are required, lifting heights are minimized. The overhead cranes are equipped with extensive safety devices. For example, redundant lifting machinery and brake systems can be installed.

4.3.3 Normal operation

Many years of experience are available from the storage of spent fuel in water pools. There is normally some small leakage of radioactivity from the fuel to the water in the pool. This is taken care of by decontamination circuits in the same way as in reactor stations. Small quantities of krypton-85 and tritium are entrained in the ventilation air and released to the atmosphere. Traces of iodine and particulate activity may also be discharged to the air. But most remains in the water and is collected by the ion exchangers in the decontamination circuits.

Most of the radioactivity which is released comes from the receiving section where the fuel is handled. If it is conservatively assumed that all the gap activity in fuel with cladding damage arriving at the central storage facility is released, the following quantities of activity are released (table 4-1).

The values for krypton and iodine were calculated on the basis of the assumption that 300 tons of spent fuel with 0.2 % canning damage arrive per year and that the gap activity in fuel with cladding damage is released in the central storage facility.

A considerable portion of the tritium which diffuses into the cladding gap is probably bound to the cladding material. Tritium is released from the stored fuel into the pool water in the form of tritiated water (HTO).

The figures for other nuclides were estimated on the basis of data from /4-11/.

Table 4-1. Estimated releases during normal operation from central storage facility for spent fuel.

Nuclide	Release Ci/year	Recipient
Tritium	100	Air/water
Krypton-85	2 000	Air
Iodine-129	2×10^{-5}	Air/water
Other beta emitters:		
Cs-137	0.01	Water
Alpha emitters:		
Pu-239	10^{-7}	Water

The releases give rise to only very small radiation doses to the environment; on the order of 10^{-4} mrems/year to nearby residents.

4.3.4 Failures and accidents

The facility shall be so designed as to prevent the occurrence of serious incidents. American requirements specify that special attention shall be devoted to the following cases:

Case	Preventive measure
- Criticality	Safe configuration
- Fuel exposed by loss of water	Stable pool construction, leakage monitoring, high-level connections
- Fall of heavy object onto fuel	Dealt with under point 2 below
- Extensive crushing of fuel elements by external forces	Situation in rock
- 100% cooling failure	Redundant components, make-up feed, back-up power generator

The central storage facility has been designed in such a way that the failure probabilities have been reduced to very low levels in keeping with practice in the nuclear power industry.

The following incidents have been analyzed with respect to consequences and probability:

- A transport cask is dropped
- A fuel cassette or other object is dropped
- A fuel element is dropped

Such accidents entail a release of no more than 4 000 Ci krypton-85, which gives doses of less than 0.1 millirem. Probability is estimated to be around 4×10^{-4} per year in this case.

1. A transport cask is dropped

The facility is designed so that the cask is never lifted more than 9 m. This means that if the cask is dropped, none of the enclosed activity will escape. It is possible, however, that the floor may be damaged or broken through, which may cause damage to underlying areas. This does not lead to any dispersal of activity to the environment, but it may entail an increase of the dose load on clean-up personnel.

2. A fuel cassette is dropped

A fuel cassette may be dropped while being moved in the pool. It is assumed to fall in such a manner that not only the falling cassette itself, but also two other cassettes standing in the pool sustain canning damages to 10% of the fuel. The incident need not necessarily cause damage to any of the cassettes. The consequences of dropping other objects are considerably less serious, since there are no heavy lifts (aside from the cassettes) over the pools.

Based on experiences from heavy industry, the probability of a lifting accident is estimated to be 5 . 10^{-6} per lift /4-4/. This figure does not take into account the extra safety devices on the cranes mentioned in the preceding section. The number of lifts into the pools per year is 48 for BWRs and 20 for PWRs.

3. A fuel element is dropped

A fuel element may be dropped and damaged when it is lifted out of the transport cask. The incident is assumed to cause 10% canning damage. The number of lifts of incoming fuel per year is 1 200 for BWRs and 180 for PWRs.

Probabilities, releases and doses for incoming fuel are presented in Table 4-2. In the case of 11-year-old outgoing fuel, releases and doses are roughly half as large /4-1/ and probabilities the same as in the table.

Table 4-2. Results of failure analysis for central fuel storage facility. Calculated activity releases, radiation doses and probabilities.

Event	Fuel type	Release Ci	Dominant nuclide	Max.individual dose, rems	Collective dose, manrems	Probability per year
1. Cask is dropped	-	-	-	-	-	-
2. Fuel cassette is dropped	PWR BWR	4 000 3 800	Kr-85 Kr-85	1 x 10^{-4} 1 x 10^{-4}	2,0 1,8	1 x 10^{-4} 2,5 x 10^{-4}
3. Fuel element is dropped	PWR BWR	150 60	Kr-85 Kr-85	3 x 10^{-6} 1,5 x 10^{-6}	0,07 0,03	9 x 10^{-4} 6 x 10^{-3}

4.4 TRANSPORTATION OF SPENT FUEL AND VITRIFIED HIGH-LEVEL WASTE

4.4.1 Technical resumé

Sea transports of spent fuel and high-level wastes are performed by a small vessel which takes 6-8 casks on each run. The acquisition of a special ship for these transports is planned. But ordinary ships have been used to date.

The casks will be transported between the facility and the harbour by trailer. Road speed will be limited to 10 km/h. A stationary dockside crane or the roll-on roll-off method will be used for loading and unloading.

The NTL 11 transport cask has been selected as a reference cask for the transportation of spent fuel. It is assumed that NTL 12 type casks will be used to transport waste cylinders from a foreign reprocessing plant to a Swedish intermediate storage facility. The transportation system in its entirety is described in greater detail in III:2.

4.4.2 Safety measures

Type B fuel casks will be used for the transports. These casks comply with the rules issued by the IAEA (International Atomic Energy Agency) for design, construction and testing. The basic aim of these rules is to ensure that the transport cask for the radioactive material provides sufficient security against the escape of radioactive substances in connection with accidents. The intention is that all transports shall be possible with the use of conventional, generally available vehicles without any special radiation surveillance. The rules therefore prescribe the following categorical tests, which are designed to simulate a serious accident:

- Fall from 9 m onto a hard, flat surface
- Fall from 1 m onto a solid steel cylinder with a diameter of 15 cm
- Heating for 30 minutes to 800°C
- Immersion in water at a depth of 15 m for 8 hours

The first three tests shall be carried out in sequence on the same cask. The cask shall pass these tests without any leakage. Specified requirements are also made on radiation shielding and cooling.

The cask has a multi-layer construction. Outside of the innermost steel encasement is a radiation shield to absorb gamma rays and a neutron shield. Heat transmission from the contents to the walls of the cask and from the walls to the environment takes place by means of free convection via cooling flanges to the surrounding air. Connections and instruments are recessed in the top of the cask and protected by a shock-absorbing hood against damages in the event of fire or collision.

The cask shall be handled in accordance with detailed check lists. External transports must be registered with and approved by the authorities in the concerned countries in advance.

Safety can be further enhanced by the use of specially-built ships. The ship which is planned for the purpose has a low fire load and will be equipped with special safety-enhancing equipment:

- Strengthening for navigation in ice
- Monitoring equipment for radiation
- Carbon dioxide system for fire extinguishing
- Automatic sprinkler system
- Device for tracking after sinking
- Extra communication equipment

A detailed description of transportation is provided in the preliminary study carried out for PRAV /4-5/ and in section III:2.

4.4.3 Normal operation

The surface dose rate on the transport cask will not exceed 100-200 mrems/h, and at a distance of one metre from the surface of the cask, the dose may not exceed 10 mrems/h, according to IAEA recommendations. Personnel enter the holds only for loading and unloading. The design of the ship and the equipment and the location of the casks must be such that individual doses are kept low.

Adverse environmental impact in connection with the transportation of both spent fuel and vitrified waste is negligible.

4.4.4 Failures

The design criteria for the transport casks ensure that they will be able to retain their integrity in the event of serious accidents. Observations from actual transport accidents as well as special full-scale collision tests show that the casks can actually withstand greater stresses than those specified in the aforementioned criteria. A failure analysis will therefore be limited to events which give rise to extreme stresses. Such events are of an extreme nature and their probability is very low. Even in these types of events, it is possible that the cask will remain intact in most of the described cases. But in order to investigate the potential environmental consequences of various transport accidents and failures, it has been assumed that the casks are damaged.

There are three different external transport phases:

- Transportation of fuel elements from the reactor station to the central storage facility.
- Transportation of fuel elements from the central storage facility to a foreign reprocessing plant.
- Transportation of vitrified waste from a foreign reprocessing plant to a Swedish intermediate storage facility.

The types of accidents and failures which have been considered are:

1 The transport cask is dropped in connection with loading or unloading.

2 The ship runs aground and sinks.
3 Ship collision.
4 Long-lasting fire onboard.
5 Collision and fire onboard.
6 Trailer collision with and without fire.

1. The transport cask is dropped in connection with loading or unloading

Nowhere is the cask lifted higher than the 9 m for which it is designed to fall without springing a leak. For this reason, no escape of activity to the environment is assumed in such an event. If the cask is dropped from maximum lifting height down into the ship, the hold may be damaged so that the cask goes through the bottom and sinks, but without being damaged. The water is not so deep and the cask can be salvaged immediately.

2. The ship runs aground and sinks

The probability of a cask being damaged in such an event must be regarded as being very low since the speed of the ship is lower than the collision speed which the cask is designed to withstand. The boat and the casks can be salvaged without any damage to the casks.

3. Ship collision

This case is divided into two sub-cases:

- The ship stays afloat.
- The ship sinks.

Probability has been calculated with the aid of WASH 1238 /4-6/ and Liverpool Underwriters' Statistics /4-7/ to be 10^{-7} per km. The same statistics show that the boat floats in approximately 50% of the cases.

The probability of the transport casks being damaged in the event of a collision has been assumed to be 10^{-1}. This can be regarded as a pessimistic value /4-8/.

The casks are stowed in such a manner that no more than two casks can be damaged in the event of a collision. Each cask contains 3.2 tons of fuel.

The following maximum consequences are postulated for the case of damaged casks:

The casks are damaged to such an extent that 100% of the gaseous elements and 1% of the other nuclides which have diffused into the cladding gap escape. Cladding damage frequency has been set at 10%. It has been assumed that 30% of the krypton-85 and 1% of the cesium are present in the cladding gap. In the transport of vitrified waste, the cask is damaged in a similar manner, but there is no release of activity from the glass to the atmosphere. There will, however, be some leaching if the vessel sinks.

Probabilities, released activity and doses are given in Table 4-3.

4. Long-lasting fire onboard

This case can be divided into two sub-cases:

- The ship stays afloat after the fire stops.
- The ship sinks after the fire.

Probability has been calculated on the basis of the same references as above to be 3 . 10^{-9}/km. The same statistics show that the ship sinks in 90% of the cases.

According to /4-8/, it is relatively easy with existing technology to equip the ship with such effective fire protection systems that long-lasting fires are even less probable with this type of cargo. However, this has not been assumed in the calculations.

The following are postulated to be the maximum consequences:

The fire is assumed to lead to 10% cladding damage and damage to the casks so that 100% of the gaseous and 1% of the other nuclides escape from the cladding gap in fuel elements with damaged cladding. There is also some leaching from pellets or glass in the event the ship sinks.

Probability, released activity and doses are given in Table 4-3.

5. Collision and fire on board

The incident is assumed to lead to an increased release of activity if the two casks which have been damaged in accordance with case 3 are also heated due to a minor fire.

The total probability of the event is 3 . 10^{-10} per km /4-1/.

Probabilities, releases and doses are given in Table 4-3.

6. Trailer collision with and without fire

Trailer collides.

Casks are transported on trailers via public roads only in connection with transportation to the intermediate storage facility for vitrified waste. This will probably be a guarded transport at restricted speed. A collision under these circumstances must be considered to be improbable, and if it should occur, damage to the cask is highly unlikely. In other words, the event does not lead to any release

Trailer collides and burns

The same trailer as above is assumed to collide in such a manner that a fire of greater duration than that for which the transport is designed occurs. The cask and the cladding are partially damaged, and there is some vaporization of volatile nuclides in the glass.

The probability of such an accident is estimated to be 6 . 10^{-10} per km /4-6, 4-9/. Released activity is less than 1 Ci cesium-137.

4.4.5 Radiation doses

Table 4-3 gives the calculated maximum radiation doses for failures leading to the release of radioactivity. The incidents can occur far from land or near centres of population. Depending on weather conditions and the distribution of the population in the vicinity of the accident site, the dose loads may vary widely or may be nil. The dose values given in the table are maximum values calculated using a model which grossly overestimates the doses. The probabilities are averages over a transport distance of 10 000 km/year, which means that the severest type of failure has an annual probability of 3 . 10^{-6}. The probability of maximum consequence is, however, lower. The influence of various factors on the radiation doses associated with accidents during transport by sea are analyzed in a special KBS report /4-10/.

Table 4-3. Results of failure analysis for transportation to central fuel storage facility. Activity releases, consequent doses and probabilities.

Event	Release Ci	Nuclides	Max.individual dose rems	Collective dose manrems	Probability per km
Ship collision					
a) ship floats	2 000	Kr-85	0,0004	1	5 x 10^{-9}
	12	Cs-134	0,41	1 200	
	7	Cs-137	0,18	1 600	
total			0,59	2 800	
b) ship sinks	2 000	Kr-85	0,0004	1	5 x 10^{-9}
	1 200	Cs-134	0,02	2 300	
	700	Cs-137	0,0071	9 000	
total			0,03	11 300	
Long-lasting fire					
a) ship floats	6 000	Kr-85	0,0012	3	3 x 10^{-10}
	36	Cs-134	1,2	3 500	
	20	Cs-137	0,55	4 700	
total			1,8	8 200	
b) ship sinks	6 000	Kr-85	0,0012	3	2.7 x 10^{-9}
	3 600	Cs-134	0,06	6 800	
	2 000	Cs-137	0,02	27 000	
total			0,08	33 800	
Ship collision with fire					
a) ship floats	2 000	Kr-85	0,0004	1	2,7 x 10^{-10}
	120	Cs-134	4	12 000	
	70	Cs-137	1,8	16 000	
total			5,8	28 000	
b) ship sinks	2 000	Kr-85	0,0004	1	3 x 10^{-11}
	1 200	Cs-134	0,02	2 300	
	700	Cs-137	0,0071	9 000	
total			0,03	11 300	

4.5 INTERMEDIATE STORAGE, ENCAPSULATION AND DEPOSITION OF VITRIFIED WASTE

4.5.1 Technical resumé

The intermediate storage facility is assumed to be situated in a rock cavern with a rock cover of 30 m, directly above the final repository.

The transport cask with the waste cylinders is carried down to the receiving section on a trailer via a tunnel with a grade of 1:10. The receiving section contains a series of concrete radiation-shielded cells in which the cylinders containing high-level waste can be handled. The transport cask is then lifted from its trailer, raised and transferred by overhead crane to one of the "hot cells".

The cask is emptied by remote control. The waste cylinders are lifted up to the storage section's charging level where they are placed in a radiation-shielded transfer cask. The transfer cask can then be moved by overhead crane to one of the storage positions.

The storage section consists of four concrete trenches in two groups with 150 vertical steel pits per trench. Each steel pit holds 10 waste cylinders. The pits are filled and emptied from an overlying level. The floor on the overlying level is made of thick concrete with holes for each pit position. These holes are sealed by radiation-shielding concrete plugs.

When the steel pits are to be filled, the concrete plug is replaced with a mobile sliding valve. The transfer cask with the waste cylinder is positioned above the valve, the valve is opened and the waste cylinder is lowered into the pit.

Air-cooling of the waste cylinders is maintained by redundant fans for each group of trenches in the storage section. The air passes from below up through the gaps between the steel pits and the waste cylinders. It is extracted below the concrete floor and passes through a filter system before reaching the exhaust stack.

After approximately 30 years of storage, the waste cylinders are lifted out of their storage positions and transferred with the aid of the transfer cask to a cell for encapsulation. The encapsulation section consists of a number of work stations in a radiation-shielded cell. A lead jacket with a titanium casing is placed over the cylinder and lead is pured into the space between the lead jacket and the steel casing on the glass. The canister is sealed with a titanium lid by means of tight welding. The entire encapsulation process is remote controlled. The waste canister is lifted into a transfer cask which is carried on a wagon to the hoist shaft leading down to the rock repository.

The intermediate storage section and encapsulation station are described in greater detail in III:5.

The elevator is of the same design as a conventional mine elevator with guides and a winding sheave and with a number of independent braking systems. The elevator cage is suspended from a number of cables which are strong enough so that a few cables alone can support the load with a good margin of safety (10-fold safety). As an additional safety precaution, there is a water pool at the bottom of the hoist shaft which dampens the impact of a falling elevator.

The rock repository consists of a system of tunnels located approximately 500 m below the surface. There are some 40-odd parallel tunnels, 1 km long and approx. 3.5 x 3.5 m in cross-section. In the bottoms of the tunnels are holes approx. 5 m in depth and 1 m in diameter and spaced at a centre-centre distance of 4 m. Communications with ground level and the intermediate storage section take place through four vertical shafts. In addition to elevators, the shafts also house evacuation routes, drain pipes, ventilation ducts and electrical cables.

Horizontal transports are railbound. A wagon with a radiation-shielding transfer cask and a hoist is used to transport the waste canisters. The wagon is drawn by an electric tractor. The bottom of the deposition hole is filled with buffer material, after which the canister is lowered. The hole is then filled up with more buffer material. The entire handling chain is radiation-shielded.

4.5.2 Safety measures

The intermediate storage facility is designed on the basis of experiences from the Marcoule plant in France. The same design criteria as for the central fuel storage facility have been followed, which means that safety requirements have governed design and construction.

The location of the intermediate storage facility in rock provides good protection against external forces, acts of war etc.

The steel casing on the glass body is thoroughly surface-decontaminated, but must be enclosed in a radiation shield in order to be handled. The content of active waste in the glass has been limited to 9%. The waste in the glass has decayed for a cooling period of about 10 years, so that heat flux has fallen to about 1.2 kW/cylinder when the waste arrives at the intermediate facility. The waste cylinders arrive contained in transport casks. Subsequent operations take place in "hot cells" - closed rooms with thick, radiation-shielding walls, constant negative pressure and instrument monitoring. Work in the cells is performed by means of externally operated gripping arms (telemanipulators). Inside the cells, the transport cask is emptied and the cylinders are checked for external contamination. Experiences to date from Marcoule have been positive. There has been no escape of radioactivity and the cylinders have remained tight and clean and have not been damaged during handling.

Transfer casks with adequate radiation shielding are used to move the waste cylinders. The cylinders are carried by overhead cranes close above the floor. The hoist which transports the encapsulated waste down to the final repository is equipped with redundant brake systems and the hoist cables can carry 10 times their design load. If the elevator cage should fall, the fall will be dampened by the water pool at the bottom of the hoist shaft. Even if the canister were damaged, the glass would not shatter into such tiny particles that the particles would become airborne. The water pool also serves as a radiation shield for the waste canister.

The most important safety precaution is cooling of the storage section. Heat generation is a maximum of 3 000 kW in each group (of 2 storage trenches). Normally, cooling is provided by two fans connected in parallel, each with a capacity of 75 000 Nm^3/h. A third fan can be put into service as required. There is also a fourth fan on the surface. An auxiliary diesel generator constitutes a back-up power supply for the fans.

The intake air is coarse-filtered. The filters can be by-passed in the event of an excessive pressure drop or fan failure. Normally, the temperature of the exhaust air is about $80^{\circ}C$. In the event of the failure of one fan, the back-up fan is automatically brought into service. If only one fan is operative, the air flow (and the partial vacuum) will be slightly reduced and the temperature will rise to about $110^{\circ}C$ (after 40 hours). In the unlikely event that all fans should fail, a by-pass line with an automatic damper allows air to circulate with natural convection without passing through the filter system. In this case, the temperature will rise to max. $340^{\circ}C$ (after 40 h). This does not lead to any release of radioactivity.

4.5.3 Normal operation

Owing to the fact that the steel casing on the glass body is thoroughly cleaned during fabrication and that all handling of waste cylinders is performed under dry conditions, the risk of surface contamination is little. Since the waste is tightly bound in the glass, which is in turn encapsulated in welded cylinders, there will be no release of activity during the storage period. During 10 years of storage of high-level glass at the Marcoule plant in France, no activity has been detected in the ventilation filters.

4.5.4 Failures

The probability of failures in the intermediate storage facility has been minimized by design precautions. Overheating to such high temperatures that volatile nuclides are vaporized will not take place even in the event of a complete cooling failure as described above. The probability of fire is reduced by a low fire load and advanced fire protection.

Mechanical damages leading to the release of airborne radioactivity are deemed to be impossible.

5 SAFETY PRINCIPLES FOR FINAL STORAGE OF VITRIFIED HIGH-LEVEL WASTE

5.1 BARRIERS

In order to ensure an absolutely safe final storage of the high-level waste, the radioactive substances are isolated and surrounded by a number of barriers:

- chemical bonding to low-soluble borosilicate glass
- encapsulation of the glass in metal casings
- storage of the encapsulated waste cylinders in rock at a depth of 500 m

Each of these barriers provides protection against the escape and dispersal of the radioactive substances. But each possesses different protective properties and functions which both reinforce and complement each other.

In the final repository, the waste cylinders are emplaced in vertical boreholes extending from the bottom of a tunnel. A buffer material between the canister and the rock fixes the waste cylinders in place. It also serves as an additional barrier, since it possesses low permeability and some ion exchange capacity. The same sort of buffer material is also used to fill tunnels and shafts to prevent the dispersal of the radioactive substances this way. Dispersal of the radioactive substances is also retarded by sorption effects in rock fissures, should they penetrate the inner barriers.

Requirements on the isolation of the radioactive substances diminish as their radioactivity declines. Only a combination of flowing water and penetrated barriers can lead to a dispersal of radioactive substances from the final repository.

Figure 5-1 shows how the encapsulated waste is emplaced in the final repository. The design of the final repository is described in Chapter III:6. See special drawings 1 and 7 in this chapter.

High-level waste can be vitrified by means of several different processes which have been developed and demonstrated. The French process has come farthest as regards industrial application in the plant at Marcoule (AVM = Atelier de Vitrification Marcoule). Comprehensive studies of glass produced by this process have shown that such waste glass is highly durable. (See Chapter III:4.2-4 and IV:6.3.)

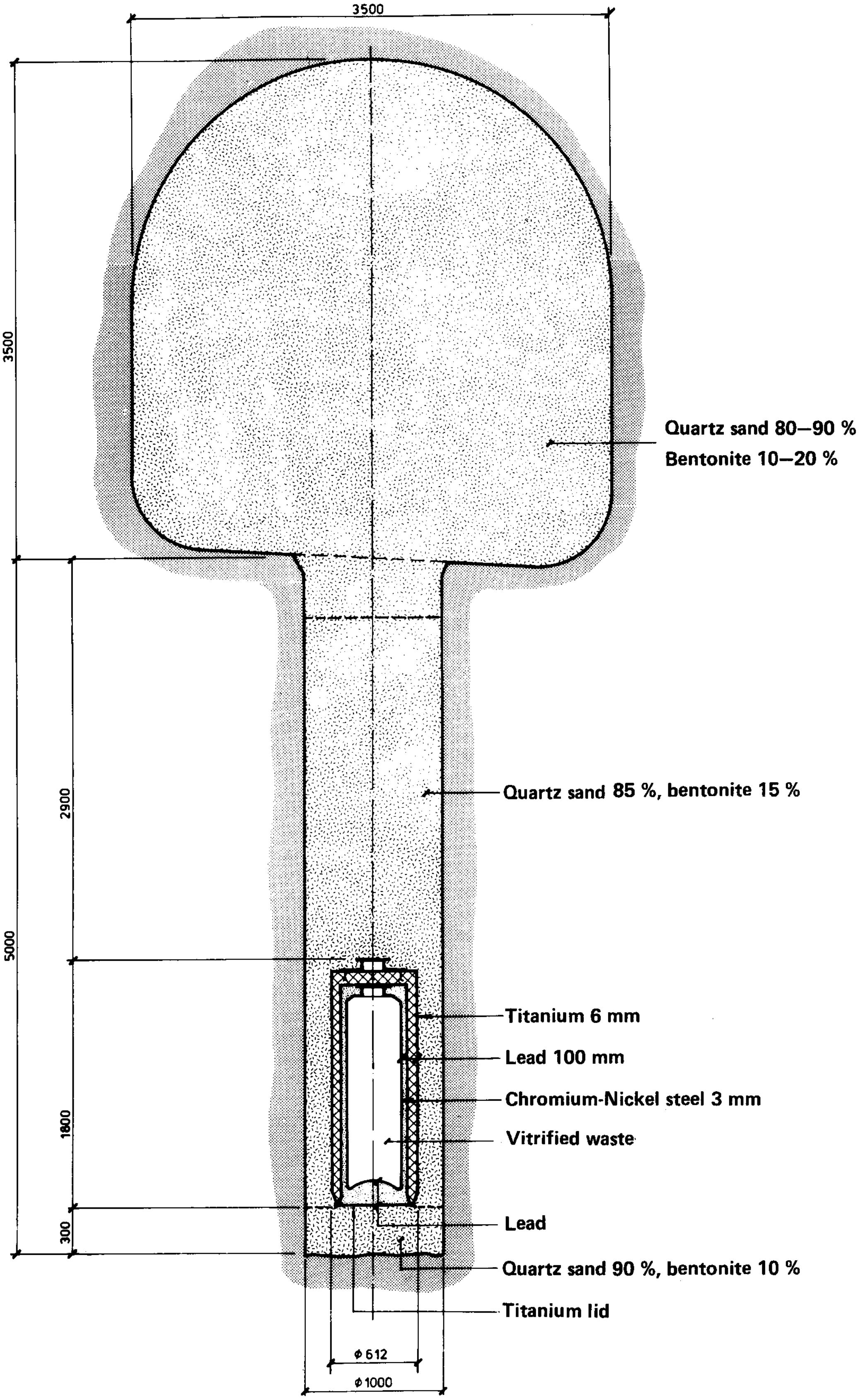


Figure 5-1. Encapsulated waste cylinder with vitrified high-level waste in a sealed final repository.

The metal canister consists of three layers: 3 mm stainless steel, 100 mm lead and 6 mm titanium. The canister is designed for a very long service life. (See Chapter III:5.3 and IV:6.2.)

The buffer material consists of a mixture of quartz sand and bentonite. This mixture has been chosen for an optimum combination of bearing capacity, plasticity, low permeability, good thermal conductivity, ion-exchanging capacity and documented long-term stability. In choosing the composition of the material, priority has been given to long-term stability and good mechanical properties over ion-exchanging capacity (see Chapter III:6.3).

Final storage of the waste shall take place at a great enough depth in the rock so that erosion phenomena (e.g. an ice age) on the surface of the earth will not have any effect. It is also important to choose good-quality rock with low permeability (low water flow) situated in an area of low seismic activity. (See Chapter III:1.3 and volume II.)

The combination of rock and buffer material with low water flow (around 0.2 litres per m^2 and year) and glass with high resistance to dissolution in water minimizes and retards the dispersal of radioactive substances, even if the metal canister should develop a leak.

Location of the repository at a depth of 500 m provides not only erosion protection as described above, but also good protection against external events such as acts of war, meteorite impacts and the like. (See Chapter IV:7.) It is absolutely safe with respect to direct radiation from the waste bodies.

5.2 TEMPERATURE CONDITIONS

The radioactive elements in the vitrified waste cause the temperature of the glass, the canister and the rock to rise /5-1/. Figure 5-2 gives the temperature on the surface and in the centre of the hottest glass body. The temperature increase in the rock at different times after deposition is given in Figure 5-3.

The temperature in the metal casings is almost the same as the temperature of the surface of the glass. At the time of final disposal (40 years following discharge from the reactor), the temperature of the titanium surface is around 60°C, which is well below temperatures which could cause gas evolution or corrosion attack. After 500 - 1 000 years, the temperature of the entire canister is below 40°C, which ensures high resistance to leaching if the canister should be violated.

5.3 RADIATION LEVELS

The radiation emanating from the vitrified waste is effectively shielded by the lead jacket. Radiation dose rate (mrems/h) as a function of canister thickness is presented graphically in figure 5-4 /5-2/.

The total dose rate from both gamma radiation and neutrons is 1 000 mrems/h after 30 years with a 10 cm thick lead jacket.

Extra radiation shielding is therefore required for moving canisters from the encapsulation station to the final repository site. The shielding is thick enough to minimize radiolysis in the final repository and prevent gas evolution resulting from such radiolysis. Calculations /5-3/ of oxygen and hydrogen levels at considerably higher dose rates (0.5 rad/s) give equilibrium values which are considerably lower than the solubilities of these gases in water.

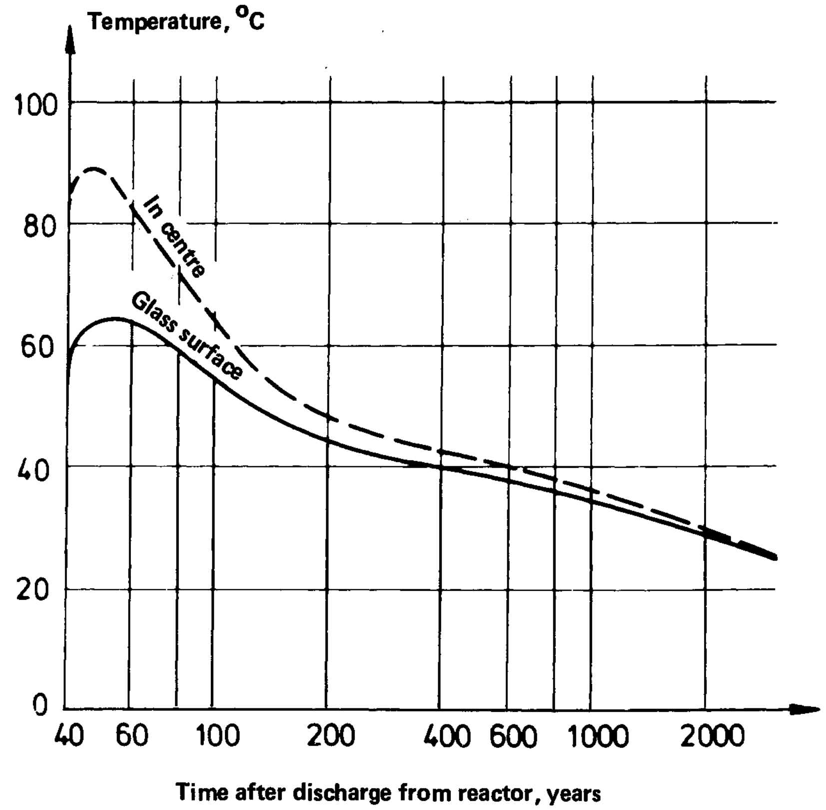
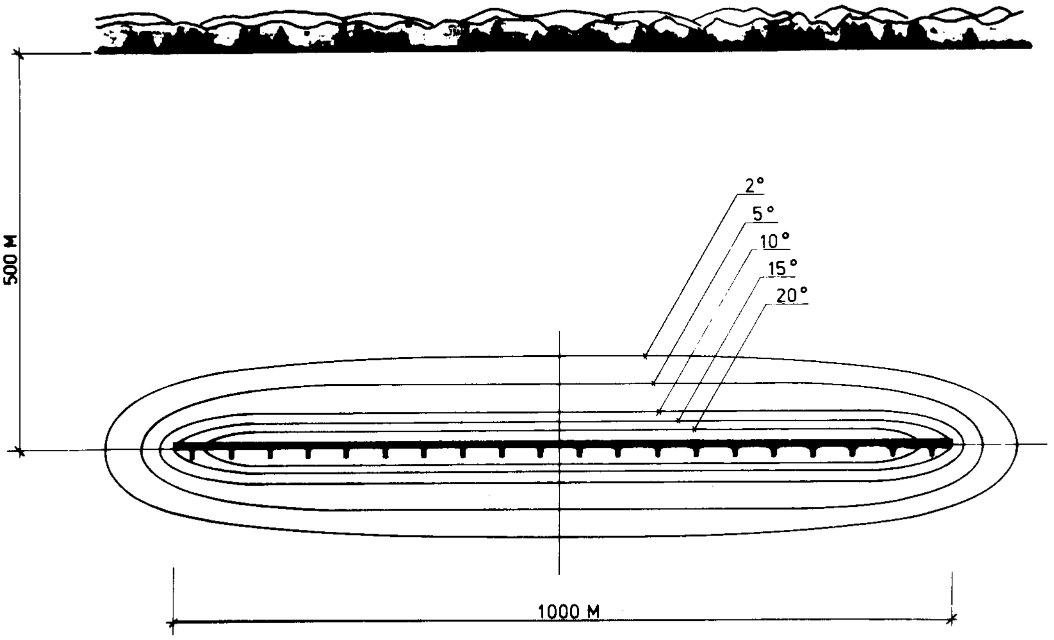
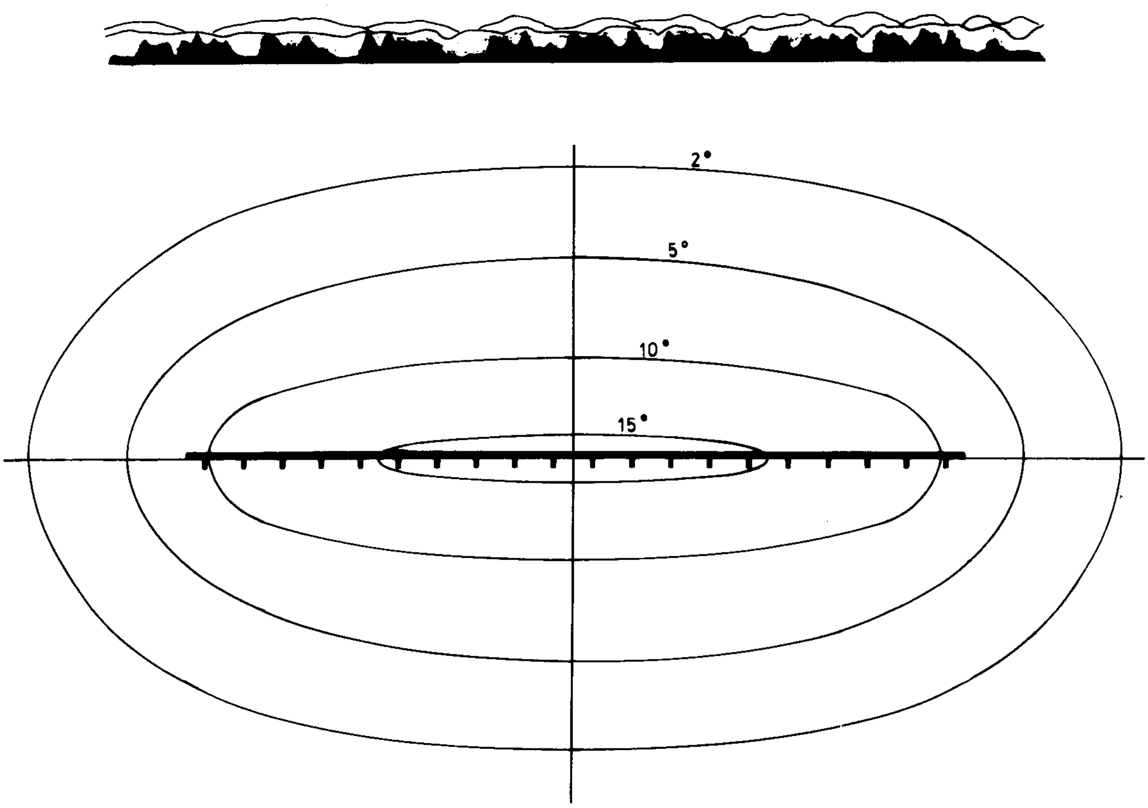


Figure 5-2. Temperature of the hottest glass body in the final repository as a function of time.

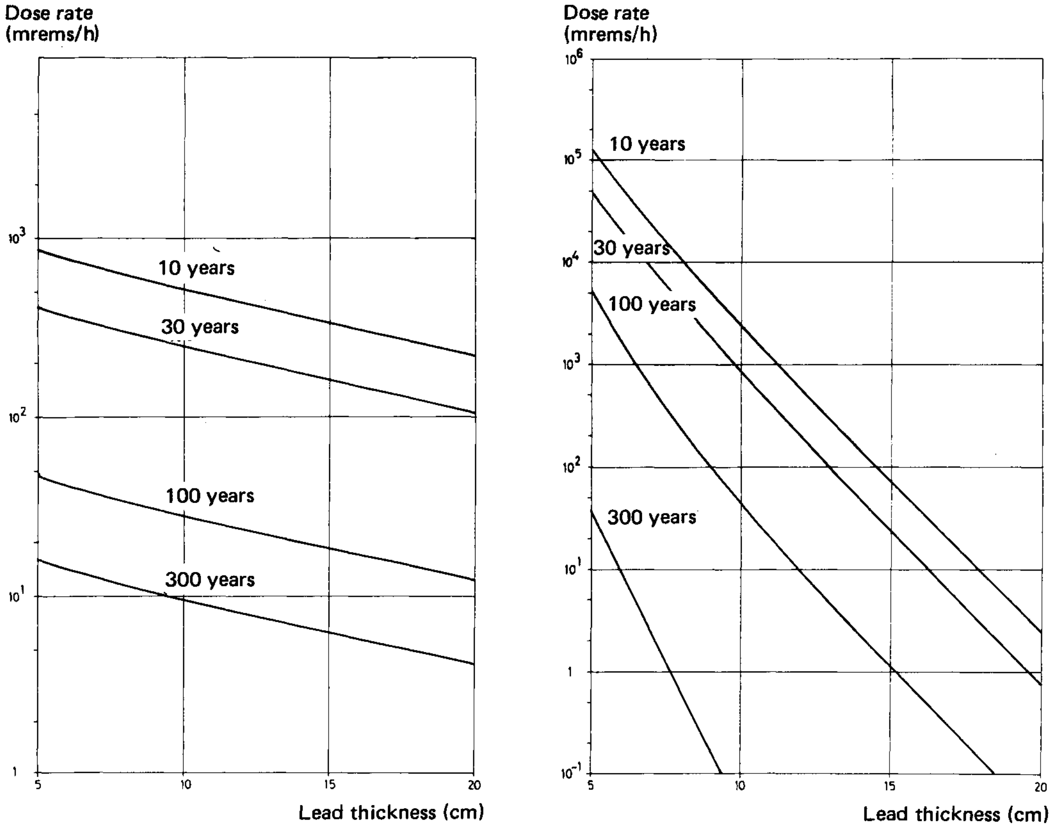


50 years after deposition



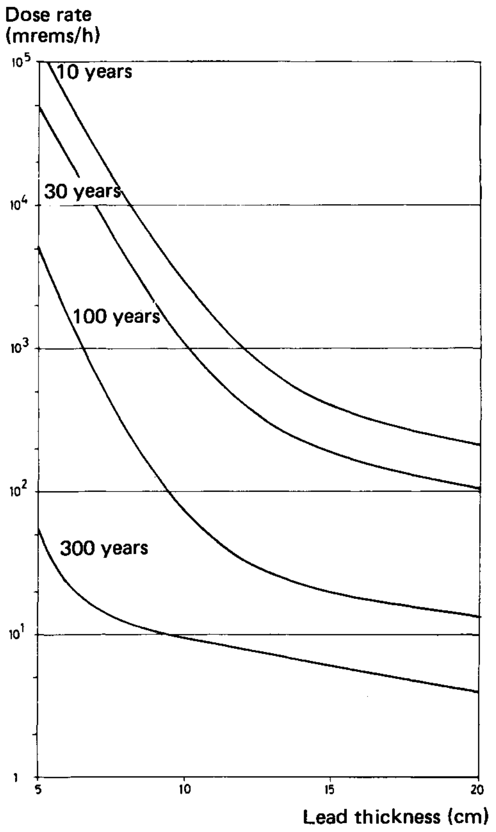
600 years after deposition

Figure 5-3. Temperature increase in the rock formation around the final repository 50 and 600 years afer deposition.



A. Dose rate from neutrons.

B. Dose rate from gamma radiation.



C. Total dose rate from gamma radiation and neutrons.

Figure 5-4. Dose rate on the outside of encapsulated vitrified waste from PWRs at different lead thicknesses.

6 DISPERSAL PROCESSES

6.1 THE DISPERSAL PROCESS – GENERAL

6.1.1 Dispersal possibilities

As is explained in Chapter 5, the waste is isolated from the biosphere by a number of barriers. Various possiblities for the dispersal of the radioactive elements from the final repository are discussed below.

A distinction can be made between:

- Slow processes
- Extreme events which lead to sudden dispersal of radioactive elements

In order for the slow dispersal of radioactive elements to be possible at all, the metallic canisters around the glass bodies must be damaged in such a way that water comes into contact with the waste glass. Chapter 6.2 describes the two studied cases of initial capsule damage and slow decomposition of the encapsulation material.

The leaching of radioactive elements from vitrified high-level waste takes place at a very slow rate. The leaching rate is influenced by a number of factors. These factors are discussed in Chapter 6.3, which also specifies the values for annual leached quantity of radioactive elements used in the analysis.

The dispersal of radioactive elements in and through the rock is affected by groundwater movements in the vicinity of the final repository and by sorption in buffer material and rock fissures. These factors are dealt with in Chapters 6.4 and 6.5.

The probability of extreme events which can penetrate the rock barrier and cause the rapid dispersal of radioactive elements is extremely low. The most important cases and their consequences are discussed in Chapter 7.

6.1.2 Consequences of dispersal

If the radioactive elements penetrate the various barriers, this can eventually lead to their coming into contact with the biosphere. Since the only dispersal mechanisms of any importance which have been identified entail dispersal via the groundwater, such contact is mainly achieved in a receiving body of water.

Three main cases of primary recipients have been identified:

- deep-drilled well close to the final repository
- lake close to the final repository
- Baltic Sea

Transport in the biosphere, different exposure situations and the determination of radiation doses and medical effects are discussed in Chapters 6.6 - 6.8.

The consequences of dispersal are analyzed in Chapter 6.9. The results are reported in the form of calculated inflows of radioactive elements to the primary recipient and calculated maximum individual doses (expressed as rems per 30 years to an individual) and collective doses (expressed as manrems per MW electricity and year). A comparison is also drawn between these calculated consequences and the permissible limits which are used in evaluating radiation protection (see Chapter 2) and natural radiation levels in Chapter 8.2.

6.2 CANISTER DAMAGE

6.2.1 Initial canister damage

The metallic canisters will be fabricated under very rigorous quality control. The canisters will also be inspected after fabrication for both leakage and weld flaws (see Chapter III:5.5). Nevertheless, the possibility that an isolated canister might have small defects which could lead to its penetration by water, cannot be completely disregarded. The probability of such an event can, however, be reduced to a very low level.

A special study was conducted of matters pertaining to initial canister damage and failure as a result of loading /6-1/.

Calculations /6-6/ show that the lead-titanium canister can withstand the hydrostatic groundwater pressure without the support of the waste cylinder.

Lead is a relatively ductile material, so there are no residual stresses left after casting. The risk of pin-holes being created during casting is low, due to the thickness of the lead jacket.

Titanium is considerably less ductile than lead and residual stresses may occur in the welds. Failure of the titanium casing is possible due to cracking in e.g. welds. Nor can penetrating weld defects be discounted completely. Non-destructive testing greatly limits the probability of such defects.

The probability of the canister being damaged in connection with handling accidents is very low. Furthermore, such events occasion special inspection and, if required, repairs. As is established in Chapter 4, the probability that the load will be dropped in connection with lifts by overhead crane is estimated to be around once per 200 000 lifts.

In order to get an idea of the probability of initial canister damage, we can make certain comparisons with other fields, such as various disciplines within nuclear power technology, pressure vessel fabrication, fuel rod fabrication etc. For example, experience shows that the defect frequency for ASEA-ATOM's fuel rods is $3 \cdot 10^{-5}$ per rod /6-2/. American statistics for ordinary pressure vessels show that the risk of failure for ordinary pressure vessels at a confidence level of 99% is around or below 10^{-5} per year /6-5/. The consecutive metal casings of steel, lead and titanium should give an even lower frequency of leaking canisters in the final repository.

The probability of initial canister damage is estimated on the basis of the above considerations to be considerably lower than 10^{-4} per canister in the final repository. The total number of canisters in the final repository will be about 10 000. One reference case which is studied in the consequence analysis is that of one damaged canister in the final repository. This single waste body is dealt with in the analysis as if it were unencapsulated and the entire glass surface were accessible for leaching. Cases involving more than one or two initially damaged canisters are deemed to be so improbable that they do not have to be considered.

6.2.2 Slow decomposition of canisters

The encapsulation materials have been selected with the intention of ensuring very good durability and service life under the conditions prevailing in the final repository. The properties of the encapsulation materials are dealt with in Chapter III:5.3. The question of the service life of the canister is also discussed in /6-3/, which was prepared by the Swedish Corrosion Research Institute (KI) and its reference group. Some of the members of this reference group estimated the period of time during which the canister remains completely intact to be at least 1 000 years, while others estimated its life to be at least 500 years.

It is clear from the KI report /6-3/ that these estimates assume that local corrosion could be the limiting factor which determines the time during which a canister is completely intact. However, local penetrations of the canister after 1 000 (or 500) years do not necessarily mean that the glass body in its entirety will be exposed to the groundwater and its leaching action. In order to get an idea of how long a time it can be expected to take before the entire glass surface is exposed to leaching, the following factors are analyzed:

- General corrosion of lead and titanium
- The rate of water flow around the canisters
- The solubility of corrosion products from the encapsulation materials

The rate of general corrosion due to oxidation of titanium has been determined by various studies to be between 0.1 and 0.5 µm/year in seawater of relatively high temperature /6-3/. Under these relatively unfavourable conditions, a period of 12 000 - 60 000 years is required to penetrate a 6 mm thick titanium plate. Both the salt content (Cl^-) and the temperature in the final repository are expected to be considerably lower than in the test which gave the above corrosion rates. (See Chapter III:6.) This should have a favourable effect on the service life of the canister.

The corrosion of lead in the conditions prevailing in the final repository will be limited by the availability of oxygen. According to ref. 6-3, appendix 4, the maximum possible corrosion rate will be 1.24 kg lead per 1 000 years and metre of canister length. A canister contains about 3 900 kg lead and is about 1.7 m long, so it would require about $1.8 \cdot 10^6$ years to oxidize all the lead in the canister.

As long as metallic lead is in contact with the chromium-nickel steel cylinder, the lead acts as a cathodic protection for the cylinder. This reduces the probability that the chromium-nickel steel will corrode away.

The rate of water flow around the canisters is very slow. As is noted in Chapter II:5, this rate is estimated to be about 0.1 - 0.2 litres per year and m^2 in normal cases. The cross-sectional area of a canister is about 1.5 m^2. General corrosion is therefore also limited by how rapidly the corrosion product can be carried away by the water.

In the corrosion of lead, low-soluble films and precipitations of various lead carbonates are formed. Grenthe and Wallin /6-7/ calculated lead levels in ambient water in cases where the lead casing on the cylinder is in contact with the groundwater and discussed the importance of the composition of the water. With the conditions prevailing in the final repository, levels of around 0.02 µm, i.e. $5 \cdot 10^{-3}$ mg/l, are obtained. Gelin /6-4/ has calculated the lead levels and the time required for complete dissolution under certain circumstances on the basis of the levels of sulphate and carbonate in the groundwater. It is estimated that the lead level could be $5 \cdot 10^{-3}$ mg/l. With a water flow of 1 litre per canister and year, it would require $7 \cdot 10^{11}$ years before a quantity of lead equal to that incorporated in the casing has been carried away by the groundwater.

It was not possible to perform a systematic analysis of how long a time it can be expected to take before the entire glass body is exposed after the canister has been breached. The above considerations would indicate that it probably takes at least tens of thousands or hundreds of thousands of years. However, in lieu of more precise data, it is necessary to carry out a consequence analysis on the basis of hypothetical cases which can be regarded as being more unfavourable than actual cases.

The following reference cases have been assumed for the purposes of the consequence analysis:

1 The integrity of the canister is violated due to local corrosion no earlier than 1 000 years after the fuel has been discharged from the reactor. (As is evident from the consequence analysis, the difference is insignificant if 500 years is assumed instead.)

2 The canisters are gradually decomposed over a period of 1 000 - 6 000 years following discharge of the fuel from the reactor. For the calculations, this means that the surface accessible for glass leaching increases linearly from 0 after 1 000 years to 100% after 6 000 years, when the entire surface area referred to in Chapter 6.3 is accessible.

6.3 LEACHING OF RADIOACTIVE ELEMENTS FROM VITRIFIED WASTE

6.3.1 Leaching of glass - general

Different types of glass possess widely varying properties with regard to the leaching of components in contact with water. Leaching of radioactive elements from the types of borosilicate glass which are of interest here has been studied by means of direct experimental tests of glass bodies which are representative for the different production processes. Such tests have been conducted in France, England, the USA and Canada. The results have been compiled in a special report /6-8/. Swedish verification measurements are currently in progress /6-9/. Industrially produced glass for the vitrification of high-level waste is judged to possess leaching resistance properties which are equivalent to those of the test bodies /6-10/.

Leaching rate is specified as weight loss per unit of surface area and time, normally gram per cm^2 and day ($g/cm^2 \cdot d$). In order to be able to calculate the leaching rate from a given body, it is necessary to know the surface area which has been exposed to water. In the case of waste glass, it is therefore necessary to estimate the surface enlargement which may take place in connection with handling. The leaching rate is influenced primarily by temperature and chemical environment. The quality of the glass may be affected by imperfections in fabrication or by ionizing radiation from the disintegration of the radioactive elements.

The low rate of water flow in the final repository must be considered in evaluating the leaching of the waste glass.

A detailed review of the properties of the waste glass is submitted in Chapter III:4.2-4. Following is a brief resumé of the factors which are of interest in connection with the safety evaluation.

6.3.2 The mechanism of leaching

It has been found experimentally that the leaching rate (ℓ) from the glass surface for a given element at constant temperature varies with time (t) according to the following formula:

$$\ell = \frac{A}{\sqrt{t}} + B \qquad g/cm^2d$$

where A and B are empirical constants.

This can be explained theoretically by the hypothesis that leaching is a combined process of diffusion-controlled ion exchange and pure dissolution of the glass. The first term, $A/\sqrt{t}$, represents the diffusion-controlled ion exchange, while the second term, B, represents dissolution.

The ratio of A to B for a given element is determined by its ionic charge and ionic radius and the manner in which it is bound in the glass. In the ion exchange process, elements in the glass surface are replaced by H_3O^+ and a depleted film is formed which induces ions on the inside of the film to migrate out towards the surface in order to reduce the concentration gradients in the glass. This leads to the growth of the depleted film. Increasing film thickness leads to increasing migration resistance and the growth rate declines and eventually becomes equal to the rate of dissolution, i.e. $A/\sqrt{t} \approx 0$, and ℓ approaches an equilibrium value of B.

6.3.3 Leaching of different elements

When experimental data are reported, the leachin of an element is usually converted to an equivalent dissolution of the glass as such based on the leaching of the element in question:

$$\ell = \frac{a}{A_o} \cdot \frac{P}{S} \quad g/cm^2d$$

where a is the leached activity in Ci/day, A_o is the original activity in the specimen in Ci, P is the original weight of the specimen in grams and S is the contact surface area between the glass and the leachant in cm^2.

In this way, different glasses containing different levels of radioactive material can be directly compared.

The main factor which distinguishes different nuclides is the duration of the diffusion phase. After 100 days or so, the leaching rate for most nuclides has reached a constant value equal to the rate of dissolution of the glass itself. There are minor differences between the different nuclides, however.

6.3.4 Experimental results

The exact composition of the waste glass which will be obtained from France after contracted reprocessing has not yet been established. But it will contain 9% fission products.

Leaching rate differs widely between different types of glass. Values between 10^{-5} and $5 \cdot 10^{-11}$ g/cm^2d are given in the literature - the latter value for a Canadian nepheline syenite glass which was "leached" in soil below the water table for about 15 years /6-11/.

The most relevant values for the present safety analysis are those for French glass intended for high-level waste from LWR fuel. A leaching rate of $2 \cdot 10^{-7}$ g/cm^2d at 25°C has been chosen for the safety analysis /6-12/. See also Chapter III:4.

6.3.5 Temperature dependence

Higher temperature leads to more rapid diffusion and dissolution of the glass. The magnitude of this effect also depends on the composition of the glass. In the case of French glass, the leaching rate increases by a factor of 10 for an increase of temperature from 25°C to 70°C and a factor of 35 for an increase from 25°C to 100°C /6-8, 6-13/.

In the case of other types of glass, such as English glass, greater temperature dependence is claimed, but the English sampling method does not appear to be relevant in this case. Measurements of leaching at elevated temperature carried out within the KBS project indicate a temperature dependence of the same order of magnitude as that specified for French glass /6-14/.

During final storage, some temperature increase is obtained in the glass as a result of decay heat generation. Figure 5-2 shows the temperature of the glass body as a function of time (reprocessing after 10 years). It can therefore be assumed that the temperature which determines the leaching rate is:

Time period	Temperature	Temperature factor
0 - 100 years	approx. 70°C	10 times
100 - 2 000 years	approx. 40°C	3 times
2 000 - years	approx. 25°C	1 time

The temperature factor times the leaching rate at 25°C gives the leaching rate at the temperature in question.

During storage for extremely long periods of time, as in the final storage of high-level waste, a very slow crystallization may take place. For obvious reasons, it is difficult to study such slow processes in realistic experiments, but the temperature is so low that it cannot be expected to make any essential contribution to surface enlargement /6-15/. (See also III:4.4.4.)

6.3.6 Surface enlargement

Following production of the waste cylinders, the surfaces of the steel cylinders are decontaminated by spraying with high-pressure water. Some crack formation is thereby obtained in the glass, which normally leads to a surface enlargement by a factor of 2-3 /6-13/.

The behaviour of borosilicate glass under rapid cooling has been studied at Battelle-Northwest /6-16, 6-17/. A low, controlled rate of cooling must be maintained in the glass casting process. If the glass is allowed to cool freely in air, it cracks into blocks, leading to surface enlargement by a factor of 10-15. Faults in the process could possibly lead to an equivalent surface enlargement, but they should be detected and the glass can be recast.

Mechanical damage and surface enlargement could occur in connection with transportation and during lifts.

The magnitude of the surface enlargement caused by mechanical damage is highly dependent on the forces involved. At Battelle-Northwest, tests have been conducted with respect to this factor /6-18/. Borosilicate glass bodies of varying size encapsulated in stainless steel were dropped from varying heights onto a flat surface. An impact speed of 130 km/h gave up to 40 times surface enlargement, while half this speed gave up to 10 times surface enlargement.

The probability of surface enlargement due to damage in transit is low. If such damage nevertheless occurs, the glass can be returned for recasting or other measures may be adopted to minimize the consequences.

It has been assumed in the safety analysis that the deposited waste cylinders have a surface enlargement of, on the average, about 5 times the geometric surface area, i.e. it is assumed that each waste cylinder is leached over an area of 10 m^2 of glass surface. (The geometric surface area is about 2.0 m^2.)

6.3.7 pH effects

The influence of pH on the leaching resistance of the waste glass is elucidated in Chapter III:4.3.3. The following factors are of primary importance:

- water flow rate near the glass
- influence of leaching products
- influence of buffer material

As has been mentioned in other contexts (see Chapter II:5), the rate of water flow in the repository has been calculated to be 0.1 - 0.2 litres per m^2 and year. This means that the water flow over the surface of the glass when the canisters have been violated may be virtually stagnant. Products of reactions between the glass and the water tend to increase the pH. Static leaching experiments show that the pH stabilizes at 8, due to the dissolution of boron oxides /6-9/.

Water chemistry in the buffer material is discussed in Chapter III:6.3. The conclusion reached there is that the pH can be expected to be between 8 and 9. Locally, the pH value may exceed these limits for a limited period of time. But this can only occur in or near corrosion pits which penetrate the lead-titanium canister.

As is evident from Chapter III:4.3.3.3, a considerable change in pH is required in order to obtain a 10-fold increase of the leaching rate for the waste glass. The pH variations which can be expected over large glass surfaces exposed to groundwater in a final repository are considerably less than those required for such large changes in the leaching rate. The expected pH interval is 8-9, while a pH greater than 11 or less than 4 is required for a 10-fold increase. (See III:4.3.3, III:6.3 and ref. /6-19/.) In the consequence analysis, the influence of pH on the average leaching rate for the entire leached glass surface is therefore neglected. Significant local variations could cause no more than a local doubling of the leaching rate. This in turn has little effect on the average value, in comparison with e.g. a minor variation in the assumed surface enlargement value (5 times).

6.3.8 Easily soluble glass phases

In the fabrication of certain types of waste glass, it has been found that there is a certain risk for phase separation. Among other things, an easily soluble molybdate phase is formed in certain cases (see Chapter III:4.4.5 and /6-20, 6-21/). Different methods have been demonstrated for preventing the development of such a molybdate phase. When it develops, it mainly contains inactive components and approx. 1% of the strontium content of the glass. Since strontium-90 decays during the period of time for which the canisters are expected to remain intact, it is irrelevant to the consequence analysis, even if it is assumed that a molybdate phase forms in some cylinders.

The probability of the formation of a molybdate phase can be reduced by careful fabrication control. If a molybdate phase nevertheless forms, its influence on the final result of the consequence analysis is insignificant. The influence of a molybdate phase can therefore be neglected.

6.3.9 Radiation damage

The influence of radiation on the properties of the waste glass is discussed in Chapter III:4.4.2. The conclusion drawn from completed experiments is that beta, gamma and neutron radiation have an insignificant effect on the resistance of the glass to leaching. As regards alpha radiation, including He formation, experimental results obtained to date indicate a nearly unchanged degree of leaching. A number of follow-up studies are currently being conducted, including in France, involving the long-term irradiation of glass containing high levels of curium-244. These experiments are aimed at simulating alpha radiation corresponding to a normal dose over 1 000 years. Theoretical calculations show that large effects are improbable /6-16, 6-21, 6-22, 6-23/. Since the type of glass in question contains only 9% fission products and a correspondingly low quantity of alpha emitters, an increase in the leaching rate due to radiation damage is not expected.

6.3.10 Leaching with limited water flows

At the low water flows which are obtained in the final repository, dissolution of the glass is also limited by its solubility in the avaiable quantity of water. References /6-24, 6-25, 6-26/ establish the solubility of amorphous silicic acid (SiO_2), which is formed on the surface of the glass, to be about 150 ppm at 25^oC and about 400 ppm at 100^oC.

The glass mass in a waste body weighs about 420 kg, of which about half is SiO_2. At a water flow of 0.2 litres per m^2 and year, approximately 30 mg/year or $1.3 \cdot 10^{-7}$ of the silicic acid in a glass body can be dissolved per year at 25^oC.

But glass corrosion cannot be regarded solely as the dissolution of silicic acid in water. Leaching of radioactive substances is also dependent upon their diffusion through the amorphous silicic acid film which is formed on the glass surface. However, preliminary calculations indicate that such diffusion is extremely slow /6-27/.

A leaching rate of $2 \cdot 10^{-7}$ g/cm^{2}d (see 6.3.4) entails a removal of 10^{-4} cm glass in about 1 000 days (three years). It also entails the removal of more than 4 g SiO_2 per glass body and year. This is more than 130 times greater than is permitted by dissolution in the water flow specified above. Available experimental material on which to base a careful analysis of the leaching of radioactive elements at a limited water flow is, however, limited. Conservative assumptions must therefore be made in the consequence analysis.

6.3.11 Summary of assumptions concerning glass leaching

On the basis of the above analysis, the following cases have been considered in the consequence analysis as regards the glass leaching rate.

Case 1 The leaching rate at 25°C is $2 \cdot 10^{-7}$ g/cm^{2}d. At 40°C, it is three times as great and at 60-70°C it is ten times as great. The average leached glass surface is 10 m^2 per glass body, which means an average surface enlargement of 5 times. The leached fraction per year is:

Time period	Temperature	Fraction of original weight per year
0 - 100 years	70°C	$1.7 \cdot 10^{-4}$ year^{-1}
100 - 2 000 years	40°C	$5.1 \cdot 10^{-5}$ year^{-1}
2 000 - years	25°C	$1.7 \cdot 10^{-5}$ year^{-1}

In order to simplify the calculations and make some allowance for unforeseen variations due to currently unknown phenomena, this process has been further approximated as a linear process of dissolution and a complete dissolution of the entire glass body over a period of

a)	initial canister damage	3 000 years
b)	slow canister degradation	30 000 years

Case 2 Leaching of the glass is limited by the supply of water. Leaching is assumed to be limited to what is permitted by the solubility of silicic acid in water. This means a leached fraction of $3 \cdot 10^{-7}$ per year of the original weight of the glass mass or a complete dissolution in approximately 3 million years.

Of the above cases, case 2 must be regarded as the most realistic. Case 1 has been included owing to the fact that the supply of water has been more or less unlimited in all experimental studies of the leaching of radioactive glass. The only experiment which comes close to simulating the conditions existing in a final repository are the Canadian tests of nepheline syenite glass buried in soil. In these tests, a reduction in the leaching rate for strontium-90 from $4 \cdot 10^{-8}$ g/cm^{2}d in 1960 to an average of $5 \cdot 10^{-11}$ g/cm^{2}d during the period 1968-1974 was observed. As far as is understood, however, this reduction is not due to a limited supply of water. Thus, adequate experimental verification is lacking for the low leaching rates in case 2.

6.4 WATER MOVEMENTS IN THE BEDROCK

6.4.1 General calculation premises

The groundwater flow Q perpendicular to a given surface area A is calculated from the equation:

$$Q = k \cdot I \cdot A \qquad m^3/s \text{ (Darcy's law)}$$

where k is a permeability constant (m/s) and I is the hydraulic gradient, i.e. the slope of the water table. The hydraulic gradient coincides for the most part with the average slope of the surface of the ground. In a flat landscape, groundwater movements are therefore slow.

The permeability constant k is dependent upon the frequency and geometry of fissures in the rock. Fissure frequency and therefore groundwater flow declines with depth. The k values obtained in the geological field survey often amount to 10^{-6} m/s in the top 100 m stratum. At a depth of 500 m, the k values were below the measurability limit, which means that they were less than 10^{-9} m/s. A permeability of $5 \cdot 10^{-11}$ m/s was measured in the Stripa mine.

The values of I and k measured in the field studies at Kråkemåla and Finnsjö Lake give groundwater flows in good rock at a depth of 500 m of roughly 0.1 - 0.2 litres per m^2 and year.

Groundwater velocity and thereby the transit time for substances dissolved in the water depends not only on permeability, but also on the porosity of the rock, i.e. the fraction of the total volume which consists of open, water-bearing fissures. The water velocity v is given by the equation:

$$v = \frac{k}{\varepsilon} \cdot I$$

Thus, water velocity is determined not only by the hydraulic gradient I, but also by the ratio k/ε.

Like permeability, porosity normally declines with depth. Both permeability and porosity may be different in different directions, a phenomenon known as anisotropy. In a rock formation with predominantly horizontal fissures, permeability and porosity are greater in the horizontal than in the vertical direction.

Most of the dissolved radioactive waste elements in the groundwater are substantially retarded in comparison with the flow of water as a result of various chemical processes. Such retardation is dealt with in Chapter 6.5.

6.4.2 Theoretical calculations

Comprehensive theoretical studies have been carried out within the KBS project in an effort to shed light upon the pattern of flow of the groundwater in rock at various depths. Data on fracture frequency, permeability and groundwater composition and age were obtained through the geological survey programme described in Volume II.

One study conducted at the Department of Land Improvement and Drainage at the Royal Institute of Technology in Stockholm /6-28/ has shown that some of the locally infiltrated water can penetrate down to a depth of thousands of metres, after which it eventually finds its way to an outflow area at the interface between the geosphere and the biosphere. The flow rate is determined by the gradient, which is in turn dependent on topographical conditions and the extent of interconnection between fracture systems and crush zones.

A study carried out by Hagconsult et al. /6-29/ provided a somewhat different picture of the flow pattern of the groundwater. It was found that the groundwater flows affected by local conditions extend to a depth of only a few hundred metres. Below this level, the groundwater flow is said to be determined primarily by the regional gradient, which would mean that the deeper groundwater would flow horizontally for greater distances, finally discharging into the sea.

The increase in temperature in the rock around the final repository which takes place during a period of approximately 1 000 years as a result of the heat generated by the waste affects groundwater movements in different ways. The heated groundwater is of lower density and therefore strives to flow upwards. The heating leads to an expansion of the rock, reducing the width of fissures and thereby the rate of water flow. The viscosity of the water also decreases, however, leading to an increased rate of flow. The overall effects of the increase in temperature of the rock have been studied in /6-29/. The conclusion is that the pattern of groundwater flow is not appreciably altered by the increase in temperature. Similar conclusions can be drawn on the basis of Häggblom's calculations /6-30/.

Other calculations /6-29/ show that the increase in temperature in the rock, which does not exceed approx. 40°C, does not give rise to any new fractures and thereby new flow paths for the groundwater. See also the report on rock mechanics conditions in Chapter II:7.

The time it takes for the groundwater to fill the cavities in the final repository after the repository has been closed and sealed has been estimated in /6-29/ to be approximately 100 years. During this period of time, groundwater flows are directed towards the repository, so there can be no transport of substances from the repository.

A special study /6-28/ of the possible effects of solar and lunar gravitation on the movements of the groundwater ("tidal effects") have shown that such effects are negligible with respect to the transport of radioactive elements.

6.4.3 Groundwater age

Age determinations using the C^{14} method which have been carried out on water samples from different depths in boreholes in Kråkemåla (see Section II:6.3.4) have given the following results:

Borehole	Depth	Age determined by C^{14} analysis
K1	407 m	11 000 years
	493 m	8 200 years
K2	291 m	4 400 years
	510 m	4 300 years

A corresponding analysis of water from a depth of 136 m in a rock well in Finland (see Section II:5.1.7) gave an age of 4 000 years, while an age of 9 800 years was obtained from a rock tunnel at a depth of 300 m at Storjuktan.

Since the analysis requires large quantities of water (about 70 litres), age determinations have only been possible on water from relatively permeable zones. Water from less permeable formations would be expected to exhibit a higher age. The analyzed groundwater from Kråkemåla may contain some flush water from the drilling work. Such flush water is younger surface water, so the actual ages cannot be less than those indicated above.

The age determinations indicate that the rate of water flow in the rock around the final repository will be very slow.

6.4.4 Groundwater transit time

Calculating the transit time of the groundwater, i.e. the time it takes for the water to migrate from a final 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 in question. These data are not yet fully known. But calculations have been carried out for a number of reference cases employing various input data. The results obtained exhibit considerable variations depending on the choice of premises /6-28/. If the repository is situated below an inflow area for groundwater in impervious rock, the transit times may be roughly equivalent to the ages determined by the C^{14} methods given above.

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

6.5 NUCLIDE MIGRATION AND RETARDATION IN THE BEDROCK

6.5.1 General

The radioactive nuclides which are leached out of the waste can be further transported to the biosphere via groundwater flow and diffusion. Since most nuclides react chemically with surrounding geologic media, these nuclides will migrate more slowly than the groundwater. As a result of this retardation effect, also known as retention, the radioactive nuclides have more time to decay.

Various chemical reactions are responsible for retention, 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 cesium (mineralization), protactinium and americium (precipitation as hydroxide compounds).

6.5.2 Calculation model

A mathematical model which treats sorption as an ion exchange process has been used in the calculations of nuclide transport. The model and associated computer program (GETOUT) were first developed by Burkholde et al. /6-31/ and have been adapted to Swedish conditions for the purpose of the project /6-32/.

Available experimental data indicate that a safety margin can be obtained with respect to retention by treating sorption as ion exchange.

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 et al. /6-33/ and Neretnieks /6-34/ have carried out determinations of the equilibrium constants for the buffer material (10% bentonite clay and 90% quartz sand), granite and various zeolites. The data in /6-33/ encompasses 14 elements. Abroad, Burkholder /6-31/ has specified retardation factors for a large number of elements in a type of soil called Western US Desert Subsoil and Ross for granite. Landström et al. /6-35/ have carried out in-situ measurements of retardation in rock fissures at Studsvik.

Retention in the buffer material immediately surrounding the waste canisters has been neglected. Neretnieks /6-34/ has shown that diffusion through 20 cm of buffer mass is relatively rapid. But 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 /6-49/ has shown that diffusion over the distances in question is extremely low.

The retardation factor in rock can be written as follows:

$$K_i = 1 + K_a \cdot a_1$$

K_i = retardation factor

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

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

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

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

K_d = mass-based equilibrium constant $\left(\frac{Ci/kg \text{ rock}}{Ci/m^3 \text{ solution}}\right)$

a_2 = specific area for laboratory specimen $\left(\frac{m^3 \text{ rock}}{kg \text{ rock}}\right)$

Allard 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 specimens used in Allard's measurement are regarded as solid spheres, a value of approx. 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 shall 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 has been assumed to be solid, i.e. the value of a_1 has been set at 30 m^2/kg. The accessible surface area for ion exchange, a_1, has been calculated as the geometric surface area of the fissures, assuming that the walls of the fissures are flat and parallel. The retardation factors have been calculated for three different fissure sizes (Table 6-1) corresponding to the permeability span of 10^{-9} - 10^{-5} m/s /6-32/. The results are close to those obtained in /6-34/.

Recently completed supplementary measurements /6-33/ indicate 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 measurements were performed under conditions which entailed high valence states for the ele-

Table 6-1. Retardation factors for plane-parallel fissure walls in solid rock.

				Impervious rock	Crush zone
Element	K_d (m^3/m^3)	K_a (m^3/m^2)	K_i ($k = 10^{-8}$)	K_i ($k = 10^{-9}$)	K_i ($k = 10^{-5}$)
Sr	20	$2.8 . 10^{-4}$	27	57	7
Zr	3 000	$4.2 . 10^{-2}$	3 900	8 400	450
Tc	0	0	1	1	1
I	0	0	1	1	1
Cs	300	$4.2 . 10^{-3}$	390	840	90
Ce	30 000	$4.2 . 10^{-1}$	39 000	84 000	4 500
Nd	10 000	$1.4 . 10^{-1}$	13 000	28 000	3 000
Eu	19 000	$2.6 . 10^{-1}$	24 000	52 000	2 800
Ra	250	$3.5 . 10^{-3}$	320	700	76
Th	1 900	$2.6 . 10^{-2}$	2 400	5 200	280
U	15	$2.1 . 10^{-4}$	20	43	3
Np	95	$1.3 . 10^{-3}$	120	260	15
Pu	380	$5.3 . 10^{-3}$	490	1 100	58
Am	30 000	$4.2 . 10^{-1}$	39 000	84 000	4 500

ments neptunium and plutonium. In the chemical environment which can be expected to prevail in the fissure systems around the final repository, it is believed that both of these elements will be present primarily in the form of quadravalent ions. This also leads to an increase of the retardation factor by a factor of 10 /6-33/. Another phenomenon involved in the calculation of the retardation factor which has not been fully explored is the possibility that some portion of certain elements adheres to colloid particles and is thereby transported with the water flow without retardation. These questions must be subjected to special study before retardation factors larger than the ones used here can be assumed with certainty.

6.5.3 Example of calculation result

Figure 6-1 is an example to illustrate the type of result which is obtained from calculations using GETOUT. The curves in the graph describe the inflow of radioactivity to the recipient area as a function of time. The example is based on leaching from one glass body, i.e. from waste corresponding to one tone of uranium in the spent fuel. For the sake of clarity, nuclides of little radiological importance have been omitted from the figure.

The example is based on the following assumptions:

- Leaching starts 1 000 years after discharge of the fuel from the reactor.

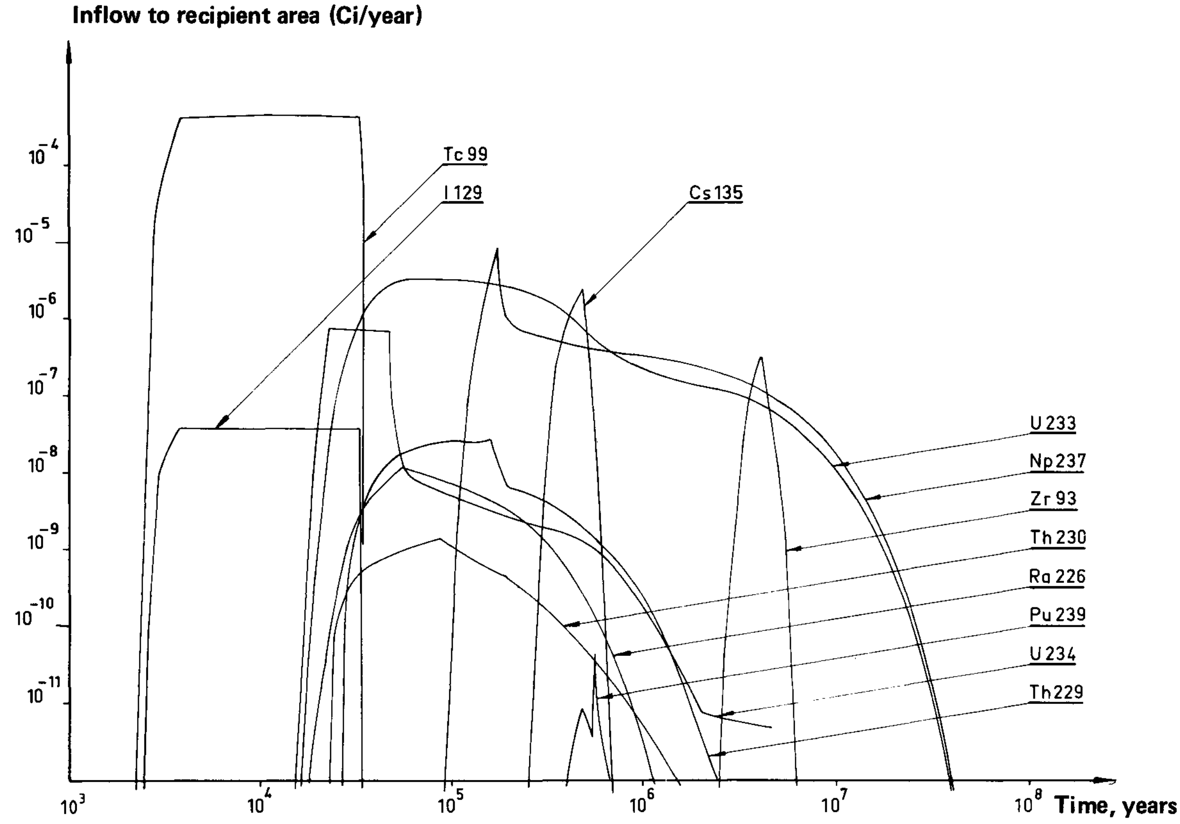


Figure 6-1. Example of calculation of inflow to recipient area at various points in time, carried out using GETOUT computer program. The dissolution time for the glass is 30 000 years.

- The leaching rate corresponds to a dissolution of the glass body in 30 000 years.
- The transit time for the water in impervious rock is approx. 400 years. The permeability of the rock is 10^{-9} m/s.
- The transit time for water in the buffer layer and a crush zone is a total of about 1 500 years. But such a long period of time has not been assumed in the consequence calculations for the safety analysis.
- Retardation factors for impervious rock obtained from Table 6-1 were used.

6.5.4 Discussion of the application of the model

GETOUT takes into account dispersion in the water flow. By dispersion is meant here that some of the constituents of the groundwater migrate faster and others slower than the average speed. A statistical distribution of retention times is obtained. The causes of dispersion are diffusion and backmixing. As a result of dispersion, a nuclide may reach the recipient faster than if groundwater flow were the only transport mechanism.

Neretnieks /6-34/ examined a form of dispersion which was not included in the GETOUT calculations and which occurs when the fissures in the rock are of different widths. In one example with a reasonable fissure width distribution, he shows that the inflow of a nuclide to the recipient may reach 5% of the maximum value after only 20% of the nuclide's transit time, due to groundwater flow alone in fissures of average width. In the consequence analysis (Chapter 6.9), the calculated inflows to the primary recipient have been subjected to standard correction in order to adjust for this type of dispersion.

Since the heavy nuclides are links in decay chains (Fig. 3-3) where the various elements possess different retardation factors, it is not possible to determine when the nuclide will first appear in the recipient on the basis of the retardation factor for one nuclide alone. Thorium-229 can serve as an example. With the parameters used in Fig. 6-1, this nuclide would appear in the recipient after about 2.6 million years if it migrated as thorium. But since thorium-229 is formed by the decay of uranium-233, which migrates much faster than thorium, thorium-229 will appear after 23 000 years.

6.6 MODELS FOR TRANSPORT IN THE BIOSPHERE

Dose loads to human beings are calculated with the aid of mathematical models (the computer program BIOPATH) for the transport of the radioactive elements in the biosphere /6-36/.

The main interface between the geosphere and the biosphere is where the groundwater comes into contact with a recipient. The hydrological conditions in the area around the final repository are therefore of decisive importance for the transport of radioactive elements. The groundwater is the means of transport for the radioactive elements which have escaped from the repository. On the basis of hydrological conditions in the areas of interest for the analysis, the models treat three main types of transport of radioactive elements to the biosphere (Fig. 6-2):

1 Outflow to a valley where a well is located
2 Outflow to a lake of specified size
3 Outflow to the Baltic Sea near the coast

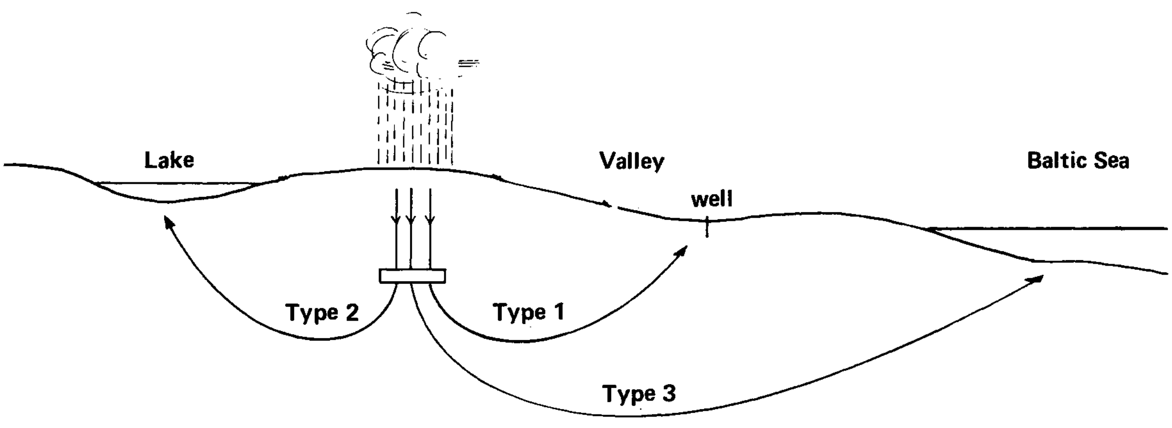


Figure 6-2. The three main types of transport of radioactive elements to the biosphere.

6.6.1 Models of ecosystems

The ecosystems are described by a model system which encompasses four interconnected areas (see Figure 6-3):

- The local area at the point of outflow from the geosphere to the biosphere.
- The regional area in the immediate vicinity of the point of outflow from the geosphere to the biosphere.
- The intermediary area consisting of the Baltic Sea with coastal areas.
- The global area.

Within and between these areas, radioactive elements can be transferred to various reservoirs, known as "compartments", such as water, sediment, earth, biota and atmosphere. Backflows also occur between the compartments in an interconnected system. The levels of radioactive elements within the compartments are assumed to be homogeneous. Figure 6-3, is somewhat schematicized and does not include the separate reservoir systems for the local and regional area considered in the model applications.

The models for the intermediary and global ecosystems apply to all main types of outflor from the geosphere to the biosphere. However, the models for the local and regional ecosystems vary depending on whether the outflow from the geosphere to the biosphere takes place in an inland area, such as in a valley or under a lake, or to the Baltic Sea.

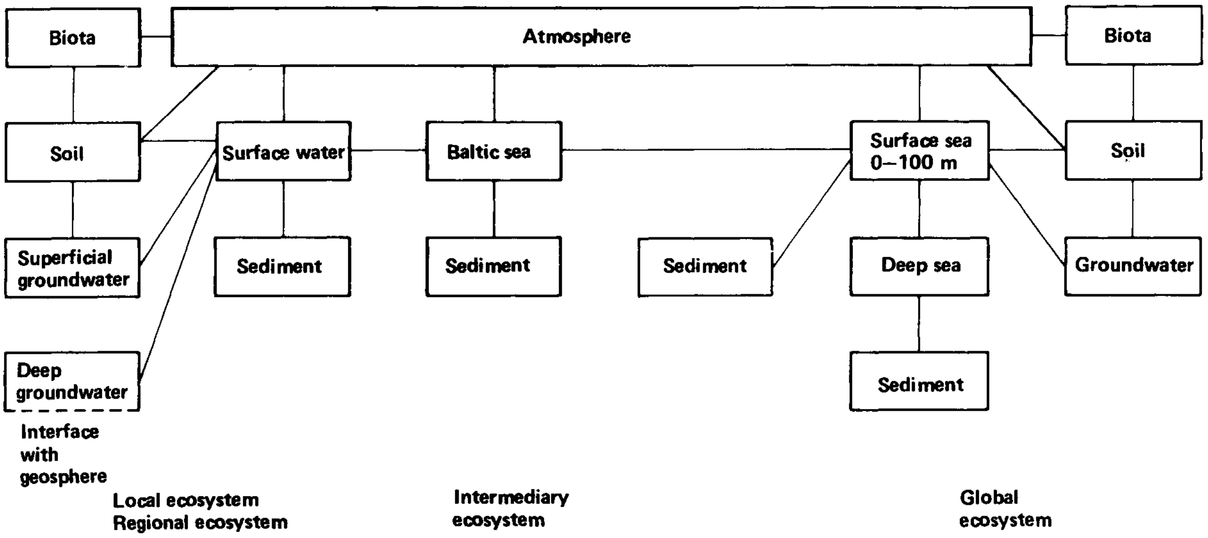


Figure 6-3. Reservoirs for the various ecosystems.

6.6.2 The local ecosystem

The various types of outflow of radioactive elements from the geosphere to the biosphere can be represented by three main cases for the local area.

The well alternative: If the groundwater issues forth in a valley, the wells and the nearest lake in the catchment area comprise the primary paths of inflow for the radioactive elements into the biosphere.

The lake alternative: If the groundwater flows out into a lake, the lake is the path of inflow.

The Baltic Sea alternative: If the groundwater containing the radioactive elements discharges into the Baltic Sea, a water area in the proximity of the coast is the primary recipient of activity.

The local ecosystem in the well and lake alternatives is assumed to consist of a 0.25 km^2 area of farmland. The area is regarded as a system of reservoirs for the radioactive elements. 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 seawater and underlying sediment within a coastal belt 2 km wide and 30 km long.

6.6.3 The regional ecosystem

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.

6.6.4 The intermediary ecosystem

The intermediary ecosystem consists of the Baltic Sea and its coastal region. The Baltic Sea reservoir comprises a water volume of $3.7 \cdot 10^5$ 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.

6.6.5 The global ecosystem

The global ecosystem encompasses a number of different reservoirs.

- The global atmosphere.

- The surface sea, which comprises the upper 100 m of the pelagic division. It mixes relatively rapidly, but has a relatively slow rate of exchange with the deep sea.
- The deep sea, which consists of the seawater below a depth of 100 m.
- The sediments at the bottom of the sea. These sediments 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. $2 \cdot 10^8$ 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 $4 \cdot 10^{18}$ kg of water which transports the radioactive elements to the surface seas and the upper soil layer.
- The biomass on the global land area - the biota. This constitutes and important reservoir in the global ecosystem for some radioactive elements with long half-lives, e.g. carbon-14, technetium-99 and iodine-129.

6.6.6 Transfer of radioactive elements

Within these regional and intermediary ecosystems, there is a turnover of radioactive elements in relation to the movements of the air and water masses which transport the activity. The elements 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 elements 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 elements 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 such mechanisms 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 intermediary areas.

The transfer of radioactive elements 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 /6-36/, mainly of the distribution of the various elements between the reservoirs, element balances, the migration of the radioactive elements from the atmospheric testing of nuclear weapons, the escape of elements 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 radioactive elements in the reservoirs when the inflow of radioactivity to the primary recipient and the coefficients of transfer between the reservoirs are known.

6.7 PATHS OF TRANSPORT AND EXPOSURE SITUATIONS

When the radioactive elements have arrived at the reservoirs in the biosphere, they can reach man in basically two different ways. The elements can be ingested into the body either through food and water or through inhalation. As long as they remain in the body, they can give rise to so-called "internal irradiation". Knowledge concerning the transport and enrichment of the radioactive elements in the food chains is therefore of great importance for being able to calculate the dose load on man. Human beings can also be irradiated by radioactive elements outside of the body - "external irradiation". Figure 6-4 illustrates some of the paths through which radioactive elements can reach man.

In order to establish the dose load, the radiation doses from inhalation and from the consumption of water and food have been calculated. Radiation doses from the handling of fishing tackle and from surface ground deposits and water, for example in connection with bathing, have also been calculated.

6.7.1 Exposure in the local ecosystem

A matter of primary interest is to establish which individual may receive the highest radiation doses, especially in the local ecosystem. These persons can be identified on the basis of their occupation, diet, living conditions etc. If this information is combined with knowledge regarding where the radioactive elements from the final repository may reach the biosphere, the so-called "critical groups" can be identified.

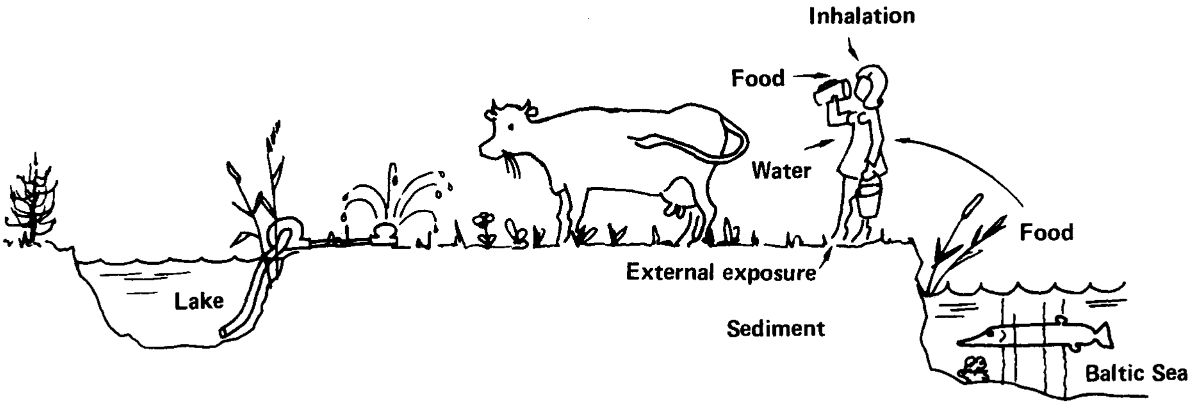


Figure 6-4. Paths of human exposure in the local ecosystem.

In the local ecosystem described in the preceding section, three alternatives were given for the primary paths of inflow to the ecosystem. They are the well alternative, the lake alternative and the Baltic Sea alternative. The local ecosystem was assumed to be 0.25 km^2 of agricultural land plus a small lake such as Finnsjö Lake near Forsmark or Götemaren Lake north of Oskarshamn. Irrigation in the area employs well water or lake water at the rate of about 200 litres per day. It is assumed that the same water supplies are used for both irrigation and drinking water.

Hydrological calculations /6-46/ show that there is a rough local balance between the inflow to the superficial groundwater from precipitation and the outflow of groundwater which issues forth and runs off with the surface water. It is therefore estimated that the radioactive elements from the final repository which reach wells in the vicinity are diluted in a groundwater volume which corresponds to the annual precipitation over the land surface (2 km^2) in the catchment area above the final repository.

The radioactive elements which enter the local ecosystem accumulate in the top 0.5 m of the ground. They are transported by the groundwater and by surface runoff. How fast the elements can enter into the ecological cycle is primarily dependent upon the sorption properties of the ground. Exposure has been calculated on the basis of the activity which reaches the local ecosystem via irrigation and the activity level which is obtained through long-term accumulation in the ground.

In the case where the outflow is into the Baltic Sea, the critical groups in the coastal zone are exposed through sea-water, sediment and fish.

6.7.2 Exposure in the regional ecosystem

The paths of exposure in the regional ecosystem are the same as those in the local ecosystem. The average exposure of the population in the region, however, is quantitatively quite different from that in the local system. The critical groups may also be different. Exposure within the regional ecosystem is used to calculate the collective dose to the population within the area.

As was explained above, the region encompasses an agricultural area 30 x 30 km within which the surface water includes a lake of the same type as e.g. Finnsjö Lake near Forsmark or Götemaren Lake near Oskarshamn. The surface water from the lake is used for irrigation within the region on roughly the same scale as the statistical average for central Sweden. The importance of the lake for irrigation in the area has also been taken into account. In the case of Finnsjö Lake, for example, 60 km^2 of the region's 900 km^2 are irrigated directly with water from Finnsjö Lake.

The coastal zone for the regional ecosystem is identical to the coastal region defined for the local ecosystem in the Baltic Sea alternative.

6.7.3 Exposure in the intermediary and global ecosystems

The intermediary ecosystem includes the Baltic Sea with its coastal regions. As regards external exposure by radioactive elements, the situation for the population in the Baltic Sea area is assumed to be qualitatively the same as in the coastal zone in the local and regional ecosystems.

The number of people who are exposed in the Baltic Sea area is defined by fish consumption. The total yield of fish from the Baltic Sea is approximately 200 000 tons per year. With an average consumption of 20 kg per year and person, the affected population is some 10 million people.

Approximately 1% of the world's population lives in coastal regions comparable to that in the Baltic Sea area. For the remaining 99% of the world's population, it is assumed that internal and external exposure can occur via the radioactive elements which have accumulated in the global model's soil reservoir.

6.7.4 Paths of exposure

The paths of exposure have been chosen so that important activity intakes and external radiation from the environment can be taken into account. For natural reasons, the various inflow alternatives (well, lake and Baltic Sea) do not have all 13 paths of exposure in common within the local and regional areas. The paths of exposure which are studied for each alternative are indicated by "W" for well, "L" for lake and "B" for the Baltic Sea in the following table:

Path of exposure	Inflow alternative[a)]	Some important nuclides
Soil - grain	W, L	Th229
Soil - green vegetables	W, L	Np237, Th229, Zr93
Soil - root vegetables	W, L	Th229
Soil - grass	W, L	I129
Grass - milk	W, L	I129, Ra226, U233, Sr90
Grass - meat	W, L	I129, Th229, U233, Sr90
Grain - eggs	W, L	
Drinking water	W, L	Np237, Ra226, U233, Tc99
Water - fish (fresh and salt water fish, respectively)	W, L, B	Cs135, I129, Ra226
Land (external exposure)	W, L, B	Th229
Beach activities (external exposure)	W, L, B	Cs135, Th229, Tc99, I129
Bathing (external exposure)	W, L, B	
Fishing (external exposure)	B	Tc99, I129, Cs135, Cs137, Th229

a) W (well), L (lake), B (Baltic Sea)

All paths of exposure are included in the global ecosystem.

Tables 6-4, 6-5 and 6-6 give the three principal paths of exposure to the critical group, depending on the primary recipient, for different nuclides.

6.8 RADIATION DOSES AND MEDICAL EFFECTS

The radiological standards dealt with in Chapter 2 specify two important types of dose specifications: The dose load to the individuals who are likely to receive the highest doses and the dose load to all people. Those persons who may receive the highest doses constitute the critical group. The collective dose load is of special importance in establishing the total effects of the doses from the radioactive elements. It is thereby necessary to take into account the length of time during which the radioactive elements remain in the biosphere. The maximum collective annual doses may appear several thousand years after a releasing event in e.g. a final repository. It is therefore important to establish the long-term cumulative dose load. Calculations of this value, known as the "dose commitment", are carried out in order to get an idea of the total radiation load from a particular radiation source.

6.8.1 Radiation doses

The radioactive elements which may leak out from a final repository can reach man via the paths of exposure described in the preceding section. The slow and protracted turnover of the various elements lead to a widely fluctuating intake of radioactive elements. If the intakes are known, the radiation dose can be calculated.

Each individual nuclide has specific properties which influence the size of the radiation dose. Such physical properties include e.g. the type of radioactive decay which is exhibited by the nuclide. Chemical and biological properties also influence the size of the dose. Examples of such properties are e.g. the element to which the nuclide belongs and the chemical form of the element. Some elements which have been absorbed by the body are excreted much more quickly than others, which can also affect the size of the radiation dose.

On the basis of the relationships between radioactive element intake and radiation doses /6-37, 6-40, 6-44, 6-45/ determined by the radiation protection authorities for each nuclide, dose and dose commitment over the various paths of exposure are calculated with the aid of the BIOPATH compartment model /6-36, 6-42, 6-43/.

Doses from external exposure are calculated on the basis of concentration in air, land and beach sediment.

Some organs are more sensitive to radiation than other organs and accumulate more of the radioactive elements which they absorb. It is therefore said that a certain organ is the "critical organ" for a certain nuclide. Table 6-2 gives the dose factors per curie for the nuclides of interest in the safety analysis.

Table 6-2 Dose factors for intake with food and water or through inhalation of 1 curie of some important nuclides.

	Whole-body* dose	Bone dose	Lung dose	Gonad dose	Thyroid dose	Weighted* whole-body dose
Dose via intake of food or water (rems/Ci)						
Sr 90	9.1×10^5	1.1×10^7		2.0×10^3		1.5×10^6
Zr 93	9.1×10^{-1}					1.7×10^2
Tc 99	4.6×10^1	1.2×10^2	1.4×10^1	4.6×10^1		5.5×10^2
I 129	9.1×10^3			3.4×10^3	1.1×10^7	3.4×10^5
Cs 135	4.6×10^3	1.8×10^4	1.8×10^3	4.6×10^3		7.3×10^3
Cs 137	4.6×10^4	1.1×10^5	1.1×10^4	4.6×10^4		5.5×10^4
Ra 226	3.0×10^7	3.0×10^7				2.8×10^7
Th 229	6.1×10^4	2.2×10^6				3.4×10^5
Th 230	6.1×10^4	2.2×10^6				3.4×10^5
U 233	4.6×10^4	5.4×10^5				1.1×10^5
U 234	4.6×10^4	5.2×10^5				1.1×10^5
Np 237	4.6×10^4	1.2×10^6				2.0×10^5
Pu 239	1.8×10^4	1.1×10^6				1.6×10^5
Pu 240	1.8×10^4	1.1×10^6				1.6×10^5
Am 241	4.6×10^4	1.1×10^6				2.2×10^5
Am 243	4.6×10^4	1.1×10^6				2.2×10^5
Dose via inhalation (rems/Ci)						
Sr 90	1.0×10^6	1.2×10^7		2.7×10^3		2.3×10^6
Zr 93	2.5×10^3	1.2×10^5				1.8×10^4
Tc 99	5.0×10^1	1.3×10^2	1.5×10^1	5.0×10^1		3.6×10^2
I 129	1.0×10^4			2.6×10^3	6.0×10^6	1.9×10^5
Cs 135	3.3×10^3	1.5×10^4	1.5×10^3	3.3×10^3		5.7×10^3
Cs 137	3.3×10^4	6.0×10^4	1.0×10^4	3.3×10^4		3.8×10^4
Ra 226	4.0×10^7	4.0×10^7				3.8×10^7
Th 229	1.0×10^8	6.0×10^9				9.0×10^8
Th 230	1.0×10^8	6.0×10^9				9.0×10^8
U 233	1.0×10^6	1.4×10^7				2.7×10^6
U 234	1.0×10^6	1.3×10^7				2.7×10^6
Np 237	1.0×10^8	3.0×10^9				5.0×10^8
Pu 239	2.0×10^8	6.0×10^9				9.5×10^8
Pu 240	2.0×10^8	6.0×10^9				9.5×10^8
Am 241	1.0×10^8	2.0×10^9				4.1×10^8
Am 243	1.0×10^8	2.0×10^9				4.1×10^8

* According to ICRP2 /6-44/

** According to ICRP26 /6-40/

The weighted whole-body dose is made up of contributions from various organs in the body. The doses are weighted with the factors in Table 6-3 so that radiation doses to different organs can be easily compared with regard delayed medical effects /6-40/.

Table 6-3. Weight factors for calculation of whole-body dose.

Organ or tissue	Weight factor
Reproductive glands	0.25
Chest	0.15
Red bone marrow	0.12
Lung tissue	0.12
Thyroid gland	0.03
Bone tissue	0.03
Remaining organs (individual organ 1/5)	0.30
	1.00

The weighting factors apply regardless of age and sex and thus correspond to an average individual.

In order to evaluate medical effects on individual persons or on parts of the national or global population, the maximum generation dose (whole-body dose) to a critical individual and the maximum collective annual dose are used. The maximum collective annual dose comprises the total contribution from the regional, Baltic Sea and global turnover of activity. The generation dose is calculated for 30 years of exposure.

Genetic effects of long-term exposure can be assessed on the basis of the maximum collective accumulated dose or the global collective dose commitment to the reproductive glands. As an alternative to dose commitment, the radiation protection authorities have discussed the concept of "cumulative 500-year dose", especially with respect to dose calculations for extremely long-lived nuclides. The collective "500-year" dose is assumed here to be 500 times the collective annual dose around the time when the maximum collective annual dose is obtained.

Tables 6-4, 6-5, 6-6 give individual doses and collective doses for some of the important paths of transport to man. An activity flow to the biosphere of 10^{-6} curie per year and nuclide has been assumed for the calculations. The nuclides specified in the tables are those which give the dominant dose contribution. The dose is specified for the point in time at which the maximum dose load occurs.

The tables show that:

- Different paths of exposure may predominate, depending on whether the nuclide is carried in the groundwater from the rock repository or whether it is generated by the decay of a long-lived parent nuclide which has already reached the biosphere, such as radium-226.
- The maximum individual dose and the maximum collective annual dose to the population for a given nuclide are often obtained at different points in time.

Table 6-4. Annual individual and collective doses at the time of maximum load after inflow of 10^{-6} Ci/year to primary recipient. Well alternative.

Nuclide	Max ind.dose (rems/year)	Time* (years)	Dominant paths of exposure (rems/year)						Coll.dose (manrems/year)	Time* (years)
			1st		2nd		3rd			
Sr 90	5.5×10^{-6}	3	meat	3.3×10^{-6}	milk	1.6×10^{-6}	water	6×10^{-6}	3.4×10^{-4}	3
Zr 93	8×10^{-11}	10	water	7.5×10^{-11}	green vegetables	2.1×10^{-12}	meat	1.6×10^{-12}	3.4×10^{-7}	2×10^{6}
Tc 99	4.9×10^{-10}	3	water	2.4×10^{-10}	milk	2.2×10^{-10}	fish	8.5×10^{-12}	3.2×10^{-7}	1×10^{5}
I 129	5.5×10^{-7}	3	meat	2.9×10^{-7}	water	1.5×10^{-7}	milk	1.1×10^{-7}	2.8×10^{-2}	2×10^{7}
Cs 135	2.3×10^{-8}	10	fish	1.5×10^{-8}	water	3.2×10^{-9}	meat	2.8×10^{-9}	1.6×10^{-4}	6×10^{4}
Cs 137	1.5×10^{-7}	3	fish	9.5×10^{-8}	water	2.3×10^{-8}	meat	2×10^{-8}	1.2×10^{-4}	6
Ra 226 1)	2.1×10^{-5}	10	water	1.3×10^{-5}	milk	7.5×10^{-6}	fish	4.3×10^{-7}	1.3×10^{-3}	6×10
Ra 226 2)	3×10^{-6}	1×10^{5}	fish	3×10^{-6}	water	1.2×10^{-9}	milk	7.5×10^{-10}	1.3×10^{-2}	1×10^{5}
Ra 226 3)	5.5×10^{-9}	5×10^{3}	water	2.8×10^{-9}	milk	1.7×10^{-9}	fish	7.5×10^{-10}	1.2×10^{-2}	3×10^{5}
Th 229	1.7×10^{-7}	10	water	1.5×10^{-7}	meat	1.7×10^{-8}	green vegetables	4.2×10^{-9}	1.7×10^{-3}	1×10^{4}
Th 229 4)	2.3×10^{-10}	6×10^{2}	green vegetables	9.5×10^{-11}	root vegetables	6.5×10^{-11}	cereal	4.9×10^{-10}	1.1×10^{-5}	1×10^{5}
Th 230	1.7×10^{-7}	10	water	1.5×10^{-7}	meat	1.7×10^{-8}	green vegetables	4.2×10^{-9}	4.3×10^{-7}	2×10
U 233	6×10^{-8}	10	water	5×10^{-8}	meat	5.5×10^{-9}	milk	1.9×10^{-10}	2.1×10^{-6}	3×10^{2}
U 234	6×10^{-8}	10	water	5×10^{-8}	meat	5.5×10^{-9}	milk	1.9×10^{-10}	2.1×10^{-6}	5×10^{2}
Np 237	1.2×10^{-7}	20	water	9×10^{-8}	meat	2×10^{-8}	green vegetables	2.6×10^{-9}	2.2×10^{-5}	3×10^{6}
Pu 239	8.5×10^{-8}	10	water	7×10^{-8}	meat	1.6×10^{-8}	green vegetables	2×10^{-9}	1.7×10^{-6}	2×10^{2}
Pu 240	9×10^{-8}	10	water	7×10^{-8}	meat	1.5×10^{-8}	green vegetables	2×10^{-9}	1.7×10^{-6}	2×10^{2}
Am 241	1.3×10^{-7}	10	water	9.5×10^{-8}	meat	2.1×10^{-8}	fish	5×10^{-9}	1.8×10^{-5}	3×10
Am 243	1.3×10^{-7}	10	water	9.5×10^{-8}	meat	2.1×10^{-8}	fish	5.5×10^{-9}	2×10^{-5}	2×10^{2}

*) Time to reach 90% of maximum annual dose after inflow into biosphere.

1) Dose not include decaying Th-230.

2) Refers to doses due to Ra 226 generated by the decay of Th 230 released at the rate of 10^{-6} Ci/year.

3) Refers to doses due to Ra 226 generated by the decay chain U 234 → Th 230 → Ra 226 due to the release of U 234 at the rate of 10^{-6} Ci/year.

4) Refers to doses due to Th 229 generated by the decay of U 233 released at the rate of 10^{-6} Ci/year.

Table 6-5. Annual individual and collective doses at the time of maximum load after inflow of 10^{-6} Ci/year to primary recipient. Lake alternative.

Nuclide	Max ind.dose (rems/year)	Time* (years)	Dominant paths of exposure (rems/year)						Coll.dose (manrems/year)	Time* (years)
			1st		2nd		3rd			
Sr 90	1.5×10^{-7}	3	meat	6.5×10^{-8}	fish	4.2×10^{-8}	milk	3×10^{-8}	3.4×10^{-4}	3
Zr 93	2.1×10^{-12}	10	water	1.5×10^{-12}	fish	5.5×10^{-13}	green vegetables	4.1×10^{-14}	3.4×10^{-7}	2×10^{6}
Tc 99	1.8×10^{-11}	3	fish	8.5×10^{-12}	water	4.9×10^{-12}	milk	4.5×10^{-12}	3.2×10^{-7}	1×10^{5}
I 129	1.6×10^{-8}	3	meat	6×10^{-9}	fish	5×10^{-9}	water	2.9×10^{-9}	2.8×10^{-2}	2×10^{7}
Cs 135	1.5×10^{-8}	10	fish	1.5×10^{-8}	beach	8.5×10^{-11}	water	6.5×10^{-11}	1.6×10^{-4}	6×10^{4}
Cs 137	1×10^{-7}	3	fish	9.5×10^{-7}	water	4.1×10^{-10}	meat	3.6×10^{-10}	1.2×10^{-4}	6
Ra 226 1)	8.5×10^{-7}	10	fish	4.3×10^{-7}	water	2.4×10^{-7}	milk	1.5×10^{-7}	1.3×10^{-3}	6×10^{1}
Ra 226 2)	6×10^{-6}	9×10^{4}	fish	3×10^{-6}	water	1.8×10^{-6}	milk	1.1×10^{-6}	1.3×10^{-2}	1×10^{5}
Ra 226 3)	1.5×10^{-9}	6×10^{3}	fish	7.5×10^{-10}	water	4.3×10^{-10}	milk	2.5×10^{-10}	1.2×10^{-2}	4×10^{5}
Th 229	9.5×10^{-9}	1×10^{4}	beach	9.5×10^{-9}	fish	1.4×10^{-10}	water	4×10^{-11}	1.7×10^{-3}	1×10^{4}
Th 229 4)	1.8×10^{-11}	4×10^{4}	beach	1.8×10^{-11}	fish	2.8×10^{-14}	water	8×10^{-15}	1.1×10^{-5}	1×10^{5}
Th 230	4.4×10^{-10}	10	fish	3.3×10^{-10}	water	9.5×10^{-11}	meat	1.1×10^{-11}	4.3×10^{-7}	2×10^{1}
U 233	2.3×10^{-9}	50	fish	1.2×10^{-9}	water	1×10^{-9}	meat	1.1×10^{-10}	2.1×10^{-6}	3×10^{2}
U 234	2.3×10^{-9}	40	fish	1.2×10^{-9}	water	1×10^{-9}	meat	1.1×10^{-10}	2.1×10^{-6}	5×10^{2}
Np 237	4.4×10^{-9}	80	fish	2.1×10^{-9}	water	1.8×10^{-9}	meat	4×10^{-10}	2.2×10^{-5}	3×10^{6}
Pu 239	2.3×10^{-9}	150	water	1.4×10^{-9}	fish	5.5×10^{-10}	meat	3.1×10^{-10}	1.7×10^{-6}	2×10^{2}
Pu 240	2.3×10^{-9}	150	water	1.4×10^{-9}	fish	5.5×10^{-10}	meat	3.1×10^{-10}	1.7×10^{-6}	2×10^{2}
Am 241	7.5×10^{-9}	30	fish	5×10^{-9}	water	1.8×10^{-9}	beach	4.7×10^{-10}	1.8×10^{-5}	3×10^{1}
Am 243	8.5×10^{-9}	60	fish	5.5×10^{-9}	water	1.9×10^{-9}	beach	5×10^{-10}	2×10^{-5}	2×10^{2}

*) Time to reach 90% of maximum annual dose after inflow into biosphere.

1) Dose not include decaying Th-230.

2) Refers to doses due to Ra 226 generated by the decay of Th 230 released at the rate of 10^{-6} Ci/year.

3) Refers to doses due to Ra 226 generated by the decay chain U 234 → Th 230 → Ra 226 due to the release of U 234 at the rate of 10^{-6} Ci/year.

4) Refers to doses due to Th 229 generated by the decay of U 233 released at the rate of 10^{-6} Ci/year.

Table 6-6. Annual individual and collective doses at the time of maximum load after inflow of 10^{-6} Ci/year to primary recipient. Baltic Sea alternative.

Nuclide	Max ind.dose (rems/year)	Time* (years)	Dominant paths of exposure (rems/year)						Coll.dose (manrems/year)	Time* (years)
			1st		2nd		3rd			
Sr 90	7.3×10^{-10}	10	fish	7.3×10^{-10}	bath	1.2×10^{-12}			3.5×10^{-6}	3×10
Zr 93	6.6×10^{-13}	10	fish	6.6×10^{-13}					3.4×10^{-7}	2×10^{6}
Tc 99	1.2×10^{-13}	3	fish	1.1×10^{-13}	beach	4.7×10^{-15}	fishing tackle	5.6×10^{-16}	6.9×10^{-7}	1×10^{6}
I 129	1.3×10^{-10}	3	fish	1.3×10^{-10}	beach	2.4×10^{-15}	fishing tackle	2.8×10^{-16}	2.8×10^{-2}	3×10^{7}
Cs 135	1.6×10^{-10}	8×10^{3}	beach	8×10^{-11}	fish	7.3×10^{-11}	fishing tackle	9.3×10^{-12}	1.2×10^{-4}	2×10^{6}
Cs 137	4.4×10^{-10}	10	fish	4.4×10^{-10}	beach	1.5×10^{-12}	fishing tackle	1.1×10^{-12}	2.6×10^{-6}	3×10
Ra 226$^{1)}$	2.9×10^{-8}	10	fish	2.9×10^{-8}	fishing tackle	1.2×10^{-15}	beach	3.9×10^{-16}	4.2×10^{-4}	2×10^{2}
Ra 226$^{2)}$	1.8×10^{-7}	1×10^{5}	fish	1.8×10^{-7}	fishing tackle	6.2×10^{-10}	beach	2.7×10^{-10}	5×10^{-3}	1×10^{5}
Ra 226$^{3)}$	7.5×10^{-11}	6×10^{3}	fish	7.5×10^{-11}	fishing tackle	1.9×10^{-13}	beach	6.6×10^{-14}	1.2×10^{-2}	4×10^{5}
Th 229	6.9×10^{-10}	1×10^{4}	fishing tackle	5×10^{-10}	fishing tackle	1.7×10^{-10}	fish	1.2×10^{-11}	3.1×10^{-5}	1×10^{4}
Th 229$^{4)}$	2.2×10^{-12}	4×10^{4}	fishing tackle	1.6×10^{-12}	beach	5.6×10^{-13}	fish	4.2×10^{-15}	7.7×10^{-6}	2×10^{5}
Th 230	3.4×10^{-11}	6	fish	3.4×10^{-11}	-		-		1.7×10^{-8}	4×10
U 233	2.3×10^{-11}	100	fish	2.3×10^{-11}					9.1×10^{-7}	2×10^{5}
U 234	2.3×10^{-11}	100	fish	2.3×10^{-11}					1.3×10^{-6}	3×10^{5}
Np 237	4.3×10^{-11}	60	fish	4.2×10^{-11}	fishing tackle	7.9×10^{-3}	beach	1.4×10^{-13}	2×10^{-5}	3×10^{6}
Pu 239	1.1×10^{-11}	90	fish	1.1×10^{-11}	-		-		1.3×10^{-7}	9×10^{2}
Pu 240	1.1×10^{-11}	90	fish	1.1×10^{-11}	fishing tackle	1.8×10^{-18}	beach	2×10^{-18}	1.2×10^{-7}	6×10^{2}
Am 241	6×10^{-12}	30	fish	5×10^{-11}	beach	9.3×10^{-12}	fishing tackle	5.5×10^{-13}	1.9×10^{-6}	2×10^{2}
Am 243	6.4×10^{-11}	60	fish	5.4×10^{-11}	beach	1×10^{-11}	fishing tackle	5.9×10^{-13}	3×10^{-6}	6×10^{2}

*) Time to reach 90% of maximum annual dose after inflow into biosphere.

1) Dose not include decaying Th-230.

2) Refers to doses due to Ra 226 generated by the decay of Th 230 released at the rate of 10^{-6} Ci/year.

3) Refers to doses due to Ra 226 generated by the decay chain U 234 → Th 230 → Ra 226 due to the release of U 234 at the rate of 10^{-6} Ci/year.

4) Refers to doses due to Th 229 generated by the decay of U 233 released at the rate of 10^{-6} Ci/year.

In the consequence analysis in Chapter 6.9, the results are given as the maximum generation dose, which is the same as the maximum individual dose for a period of 30 years. The collective dose is also given in units of manrems per megawatt electrical output and year of operation, converted from total manrems during a 500-year period. A figure of $3 \cdot 10^5$ MWe year for the entire repository containing 10 000 canisters has been used in the conversion.

6.8.2 Medical effects of ionizing radiation

When ionizing radiation is absorbed by body tissue, the tissue is damaged. Some of the damage can be repaired, while some cannot. The proportion of irreparable damage increases sharply with increasing radiation dose. The biological effects of the radiation are dependent upon a number of factors:

- The radiation dose, i.e. the quantity of absorbed energy per unit weight.
- The range of the radiation in tissue.
- The ionization density.
- The distribution of the radiation dose to different organs.
- The duration of the radiation.

If the radiation dose is expressed in "rems", the biological effects of various types of radiation are taken into account. A distinction is made between acute damage and delayed damage.

Acute damage

The ionizing radiation can kill the cells in the tissue. If a large number of cells are killed without being replaced, acute radiation damage is sustained. Sometimes, cells killed by radiation can cause indirect damage even if they are replaced by regeneration of the damaged tissue.

If the whole body receives radiation doses of more than 100 rems within the course of a few hours, changes in blood values occur. The effects of the radiation increase sharply with increasing radiation doses. At 200 rems, half of all irradiated persons are afflicted with nausea and vomiting caused by damage to the mucous membranes of the intestines. In 10%, the damage is so severe that it leads to death in the absence of medical treatment. At whole-body radiation doses of around 300 rems, the mortality rate without medical treatment is 50%.

Local irradiation of the skin with more than 500 rems causes reddening of the skin. More serious skin damage occurs at doses of more than 1 000 rems.

Foetuses are more sensitive to radiation during the first months of pregnancy than adults. Although foetal damage in human beings has not been observed after ionizing radiation doses of less than 150 rems, a number of experiments on different animal species have shown that foetal organ development is affected by much smaller doses. There is every reason to assume that this is also the case for man. The limit dose for such foetal damage is assumed to lie between 10 and 50 rems.

Radiation doses high enough to cause acute somatic damage can only occur in connection with serious nuclear mishaps. In such cases, it is primarily the personnel in reactor or reprocessing plants who may be afflicted.

Delayed damage

Radiation doses which are too small to cause acute damage can still give rise to delayed effects such as cancer and genetic damage.

Genetic (hereditory) damage can result if a reproductive cell which has previously sustained chromosomal damage caused by irradiation is involved in a fertilization. Genetic damage may manifest itself in widely differing forms, varying from an increased susceptibility to infections to congenital deformities and death.

Radiation-induced cancer does not appear immediately after irradiation, but instead manifests itself in the form of an increased cancer frequency after a number of years. This makes it very difficult to assess the cancer risk associated with an irradiation. The values for the relationship between radiation dose and cancer risk which are usually specified are drived mainly from studies of people who have been exposed to high doses. These values have then been used to estimate the cancer risk for low radiation doses. Such extrapolations have given a total cancer risk in the low dose range of between one and two cases out of every 10 000 persons, all of whom have been exposed to a radiation dose of 1 rem /6-47, 6-48/. If the radiation dose has been absorbed over a long period of time, it is believed that the carcinoma risk per rem is less.

The corresponding risk figure for genetic damage caused by long-term exposures is 1.7:10 000 per generation, of which 0.6:10 000 can be expected to appear in the first generation /6-48/. The risk of genetic damage per rem is estimated to be 3 times as great for short-term exposures. However, due to the normal high frequency of genetic birth defects, an observable increase of such defects is not likely to be obtained even after high doses. UNSCEAR /6-48/ has estimated that 10% of all births exhibit genetic defects of one type or another.

International standards and health risks

The fundamental recommendation from the International Commission for Radiological Protection (ICRP) is that all radiation doses shall be kept as low as possible in the light of economic and social considerations. All standards and limit values completely eliminate the risk of acute injuries. But as was pointed out above, the risk of delayed radiation damage cannot be completely excluded, even at very low radiation doses. But it is impossible to say with certainty whether very low radiation doses entail any risk or not.

Among the conservative assumptions on which radiation protection recommendations are based is the assumption that there is a risk which increases in direct proportion to the radiation dose. This assumed linear relationship between radiation dose and risk does not reflect any biologically/scientifically observed or calculated phenomenon in the low-dose range. It is the result of a formal extrapolation of observed high-dose effects. There are, on the other hand, scientific grounds for assuming that such a linear extrapolation constitutes an upper limit for the health risk.

6.8.3 Analysis of variation

The reliability of the dose calculations is dependent upon the design of the model system, the choise of paths of exposure, the numerical approximation involved in the calculations and the uncertainty in the empirical and derived data.

The parts of the model are the regional, intermediary (Baltic Sea) and global ecosystems. They are based on experiences from previous radioecological calculation models /6-41, 6-42, 6-43/.

The 13 paths of exposure which are taken into account by the model include relevant food chains containing the important nutrients which may contain radioactive elements. These radioactive elements may have been deposited directly on the plant, absorbed via the root system or become concentrated in animal products. The selected paths of exposure also include the doses which originate from activity in the air, ground and water. The paths of exposure have been selected to cover all relevant nuclides. The assumption that the beach consists entirely of sediment with the same concentration of radioactive elements as the sediment under the water can, in some cases, lead to considerable overestimates of external exposure. This applies especially in the case of thorium-229.

The numerical method which is used in the model has been checked with regard to the uncertainty involved in the approximations which are made. Error analysis has shown that uncertainty due to numerical approximation does not exceed 20% of the dose values. In most cases, the uncertainty is less than 5%.

The transfer parameters for each nuclide, which describe the exchange between the reservoirs (compartments) in the model system and uptake via the food chains, have been given as intervals. These intervals are defined by maxima and minima in empirical and calculated data from the literature. In cases where the interval spans a relatively large range, the dose calculations have been based on the values which give the higher dose load. The doses may therefore differ from the results in Tables 6-4, 6-5 and 6-6 if other parameters within the specified interval are used.

Variations in the following parameters significantly affect the doses to the critical group in the well alternative and the lake alternative of the nuclides cesium-135, radium-226 and uranium-233, among others. These nuclides make significant contributions to the total dose.

- Exchange between soil and groundwater or between sediment and water can result in a 50% reduction in the dose for all of the three nuclides mentioned above.
- The concentration factors for accumulation in fish are of importance. Due to natural variations in the concentration factors for fish between different ecosystems in fresh water as well as uncertainty in the measurements, the dose for cesium-135 can vary in relation to the given values by a factor of five in both directions. As far as radium-226 and uranium-233 are concerned, the choice of concentration factor for fish does not have a significant effect on the dose load in the well alternative. In the lake alternative, the doses can vary for the same reason as in the case of cesium-135 by a factor of four for radium-226 and two for uranium-233.
- Variations in the diet of the critical group also affect the doses. The critical group is meant to represent a limited number of individuals who can be exposed to higher-than-average doses due to, among other things, their eating habits. A standard diet has been established for the critical group. However, the foodstuffs which give the highest dose contributions per unit weight vary for different nuclides. Other equally representative diets may therefore entail different dose loads. In the cases of cesium-135, radium-226 and uranium-233, water, fish and milk are the most important paths of exposure (see Table 6-4, 6-5). Water consumption will not vary much from the assumed 440 litres per year. The relatively high rate of fish consumption of 50 kg per year

> and person has been assumed. The fish are assumed to come from a given lake. If consumption were to be reduced by half, the dose from cesium-135 would be nearly halved while the doses from radium-226 and uranium-233 would be reduced by less than one-third. Moderate variations in consumption of dairy products, for example a doubling, do not have any significant effect on the dose load.

The water turnover rate in the coastal region affects the dose load in the Baltic Sea alternative for both the critical group and the collective dose to the population. The sensitivity analysis /6-36, 6-42/ shows that the doses can vary by a factor of two.

As is evident from the consequence analysis in Chapter 6.9, the highest calculated doses are obtained for the critical group which uses water from an activity-bearing well. The concentration of radioactive elements in this water depends on the amount of leakage from the final repository and on the amount of dilution. Dilution is calculated on the basis of the amount of precipitation which falls over the ground surface immediately above the final repository (see volume II, section 5.1.4). Fish consumption from the nearby lake is an important path of exposure for certain nuclides which contribute to the dose (see Table 6-4).

During the time periods encompassed by the calculations, the local ecosystem in particular can undergo changes which have a considerable effect on the exposure picture. The consequences of certain major changes such as drying-up of a nearby lake or parts of the Baltic Sea have been analyzed qualitatively. In both cases, the change can give rise to new paths of exposure, owing to the fact that the sediments are used in agriculture.

Some elements become concentrated in the sediment from the lake or the Baltic Sea. Drying-up of these bodies of water and changes in the paths of exposure do not lead to any significant increase in the annual doses of the radioactive nuclides which are responsible for the dominant dose to the critical group or the collective dose to the population, owing to the fact that uptake via agricultural products cultivated on the sediment does not result in as high doses as fish consumption.

The nuclide cesium-135 is an exception. The individual dose to people living in the areas around the Baltic Sea which may be exposed due to the land elevation can be up to 10 times as high as the dose given by the calculations based on no changes in the Baltic Sea level. But the contribution to the collective dose from the Baltic Sea area is less than one-fifth of the total dose load. An increase in the exposure of a future Baltic Sea population through cesium-135 would therefore lead to a doubling of the total dose commitment over the long run.

6.9 CONSEQUENCE ANALYSIS OF SLOW DISPERSAL

6.9.1 General premises

On the basis of the data and the assumptions discussed in Chapters 3, 5 and 6.1 - 8, an analysis is carried out in this section of the consequences of a slow dispersal of the radioactive elements from the waste bodies in the final repository.

The calculation procedure for this analysis is schematically illustrated in Figure 6-5. The calculations are aimed at determining the maximum individual and collective doses for the different cases so that comparisons can be made with the grounds for evaluation given in Chapter 2.

Two reference cases have been studied in the analysis: a) initial undetected canister damage to a single canister, and b) slow decomposition of all canisters, owing to the reasons given in Chapter 6.2.

The concentration of radioactive elements in the glass and their decay with time has been obtained from the ORIGEN computer program, as described in greater detail in Chapter 3.

As explained in Chapter 6.6, three different recipients are analyzed: a well, a lake and the Baltic Sea.

The results of the GETOUT computer program have been used to calculate the inflow of radioactive nuclides to the primary recipient (see Chapter 6.5). A simplified parameter study has also been carried out independently of this calculation, providing some verification of the results.

The radioactive nuclides flowing into the primary recipient are diluted during their transport from the repository to the recipient by a certain quantity of water. This dilution has been calculated for the local conditions existing at Finnsjö Lake, whereby it is assumed that half of the groundwater flow from the final repository goes to Finnsjö Lake and half to the valley bottom where the well is assumed to be located. The groundwater is diluted in $5 \cdot 10^5$ m^3 water for the well case and $2.5 \cdot 10^7$ m^3 for the lake case. These local conditions are not particularly favourable for the well case. The dilution volume which is used is therefore regarded as being on the low side (see Chapter II:5).

On the basis of the obtained concentrations of nuclides in primary recipients, the resulting radiation doses are calculated using the BIOPATH computer program as described in Chapter 6.6 - 6.8. In practice, these computer runs have been made with an emission rate of one μCi/year and nuclide in the inflow to the primary recipient. The dose values obtained in this manner were then proportioned on the basis of the inflows obtained from the GETOUT calculations. All 13 paths of exposure (see 6.7.4) have been taken into account in the dose calculations. The relative importance of the three most important paths of these 13 is shown by tables 6-4, 6-5 and 6-6.

A reference case entailing canister penetration after 1 000 years

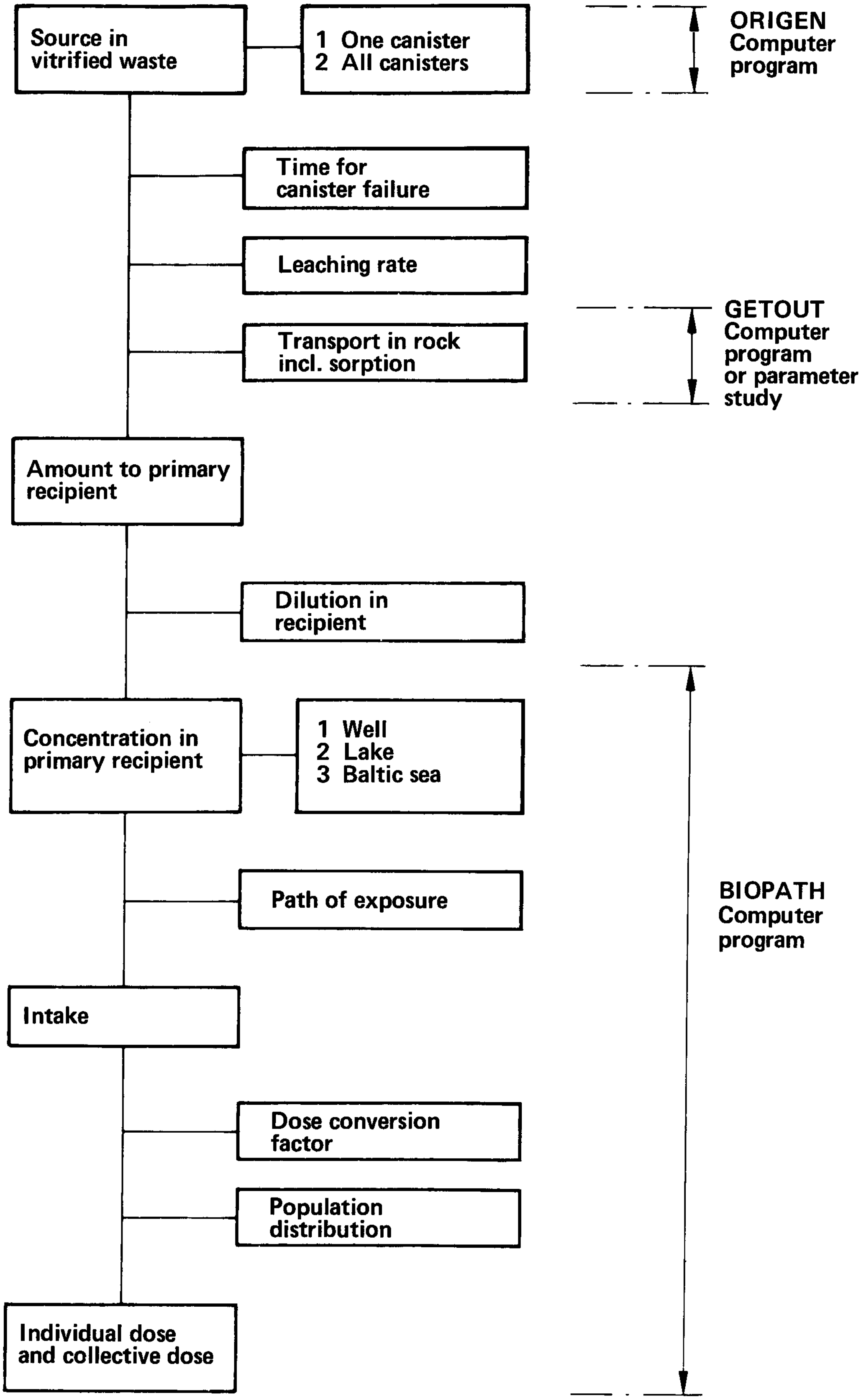


Figure 6-5. Schematic illustration of calculations of radiation doses from the radioactive elements which may be released from a final repository for high level waste.

and a glass leaching period of 30 000 years has been used in the calculations. (See 6.2 and 6.3.) The transit time in impervious rock has been set at 400 years and the retardation factors for various nuclides given in 6.5 have been used.

A case involving a single waste canister which is damaged at the time of deposition has also been studied. It is hereby assumed that the glass body is completely unprotected, i.e. that 10 m^2 glass surface (see 6.2.6) is directly accessible for leaching and that the glass is leached completely in 3 000 years. The transit time for the water and other premises are the same as in the reference case.

6.9.2 Results for the reference case - slow decomposition of the canisters

Tables 6-7, 6-8 and 6-9 present the results for the basic case. These tables apply to the three cases of primary recipient (well, lake, Baltic Sea) and each table gives:

- the maximum inflow of different nuclides in curies per year (Ci/year)
- the time at which this inflow takes place
- the maximum calculated dose to a nearby resident for each nuclide in rems per 30 years (one generation) calculated over the 30-year period which gives the highest calculated values

Table 6-7 Maximum individual doses in critical group (nearby residents) and maximum collective doses in the case of canister decomposition after 1 000 years. The calculations apply for a well as the primary recipient.

	Maximum inflow to recipient		Maximum dose	
Nuclide	Time (years)	Activity (Ci/year)	Nearby residents (rems/30 years)	Collective dose[a] (manrems/ MW(e) year)
Zr 93	4×10^6	3×10^{-3}	7×10^{-6}	2×10^{-6}
Tc 99	6×10^3	5	0.07	0.003
I 129[b]	6×10^3	1×10^{-4}	0.002	0.006
Cs 135	4×10^5	2×10^{-2}	0.02	0.006
Ra 226	5×10^4	1×10^{-4}	0.07	2×10^{-4}
Th 229	9×10^4	3×10^{-4}	0.001	7×10^{-4}
Th 230	5×10^4	1×10^{-5}	7×10^{-5}	1×10^{-8}
U 233	5×10^4	3×10^{-2}	0.06	1×10^{-4}
U 234	3×10^4	7×10^{-3}	0.01	3×10^{-5}
Np 237	2×10^5	9×10^{-2}	0.3	0.003
Pu 239	6×10^5	5×10^{-7}	1×10^{-6}	1×10^{-9}
	Maximum total dose [c]		0.4	0.007
	Time for maximum total dose		200 000 years	500 000 years

a) Calculated for the most unfavourable 500-year period and 10 000 MWe for 30 years

b) Based on 1% in the vitrified waste

c) The total dose is not equal to the sum of the above tabulated values, since the maxima for the different nuclides occur at different points in time.

Table 6-8 Maximum individual doses in critical group (nearby residents) and maximum collective doses in the case of canister decomposition after 1 000 years. The calculations apply for a lake as the primary recipient.

Nuclide	Maximum inflow to recipient		Maximum dose	
	Time (years)	Activity (Ci/year)	Nearby residents (rems/30 years)	Collective dose[a] (manrems/ MW(e) year)
Zr 93	4×10^{6}	3×10^{-3}	2×10^{-7}	2×10^{-6}
Tc 99	6×10^{3}	5	0.003	0.003
I 129[b]	6×10^{3}	1×10^{-4}	6×10^{-5}	0.006
Cs 135	4×10^{5}	2×10^{-2}	0.01	0.006
Ra 226	5×10^{4}	1×10^{-4}	0.003	2×10^{-4}
Th 229	9×10^{4}	3×10^{-4}	8×10^{-5}	7×10^{-4}
Th 230	5×10^{4}	1×10^{-5}	2×10^{-7}	1×10^{-8}
U 233	5×10^{4}	3×10^{-2}	0.002	1×10^{-4}
U 234	3×10^{4}	7×10^{-3}	5×10^{-4}	3×10^{-5}
Np 237	2×10^{5}	9×10^{-2}	0.01	0.003
Pu 239	6×10^{5}	5×10^{-7}	3×10^{-8}	1×10^{-9}
	Maximum total dose[c]		0.02	0.007
	Time for maximum total dose		200 000 years	500 000 years

a) Calculated for the most unfavourable 500-year period and 10 000 MWe for 30 years

b) Based on 1% in the vitrified waste

c) The total dose is not equal to the sum of the above tabulated values, since the maxima for the different nuclides occur at different points in time.

Table 6-9 Maximum individual doses in critical group (nearby residents) and maximum collective doses in the case of canister decomposition after 1 000 years. The calculations apply for the Baltic Sea as the primary recipient.

Nuclide	Maximum inflow to recipient		Maximum dose	
	Time (years)	Activity (Ci/year)	Nearby residents (rems/30 years)	Collective dose[a] (manrems/ MW(e) year)
Zr 93	4×10^{6}	3×10^{-3}	6×10^{-8}	2×10^{-6}
Tc 99	6×10^{3}	5	2×10^{-5}	0.005
I 129[b]	6×10^{3}	1×10^{-4}	5×10^{-7}	0.006
Cs 135	4×10^{5}	2×10^{-2}	1×10^{-4}	0.005
Ra 226	5×10^{4}	1×10^{-4}	9×10^{-5}	7×10^{-5}
Th 229	9×10^{4}	3×10^{-4}	5×10^{-6}	1×10^{-5}
Th 230	5×10^{4}	1×10^{-5}	1×10^{-8}	3×10^{-10}
U 233	5×10^{4}	3×10^{-2}	2×10^{-5}	5×10^{-5}
U 234	3×10^{4}	7×10^{-3}	5×10^{-6}	2×10^{-5}
Np 237	2×10^{5}	9×10^{-2}	1×10^{-4}	0.003
Pu 239	6×10^{5}	5×10^{-7}	2×10^{-10}	1×10^{-10}
	Maximum total dose[c]		2×10^{-4}	0.006
	Time for maximum total dose		200 000 years	500 000 years

a) Calculated for the most unfavourable 500-year period and 10 000 MWe for 30 years

b) Based on 1% in the vitrified waste

c) The total dose is not equal to the sum of the above tabulated values, since the maxima for the different nuclides occur at different points in time.

- the calculated collective dose to the population of the earth (10^{10} people) for the most unfavourable 500-year period, specified per nuclide as manrems per MW electrical output and year. In calculating the collective dose expressed in these units, it has been assumed (as throughout the rest of the report) that the waste comes from 30 years of operation of 13 reactors with a combined annual capacity of 10 000 MWe
- the maximum total individual dose and the time at which it occurs. The total dose cannot be obtained by adding together the maximum values for each nuclide which are given in the table, since these maximum values occur at different times
- the maximum total collective dose for a 500-year period.

Figures 6-6 and 6-7 present the corresponding results with respect to individual doses as a function of time. Observe that the scales are logarithmic on both axes. Figure 6-6 presents a comparison between the different nuclides for the well case. Figure 6-7 illustrates a comparison between the total doses for the cases of a well, a lake and the Baltic Sea as the primary recipient.

The results show that the dominant nuclides in the well case are neptunium-237, uranium-233, radium-226 and technetium-99. Neptunium-237 and cesium-135 dominate in the well and Baltic Sea cases. This applies to individual doses. With respect to collective doses, the largest contributions come from iodine-129, cesium-135, neptunium-237, thorium-229 and technetium-99.

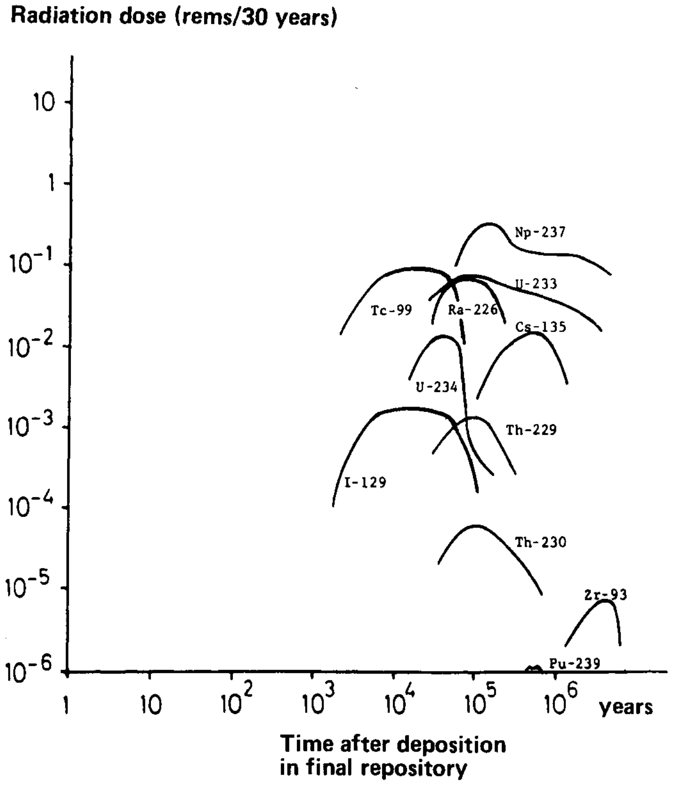


Figure 6-6. Maximum calculated individual doses to critical group (nearby residents) from different nuclides. The calculations apply for a well as the recipient and a glass leaching period of 30 000 years.

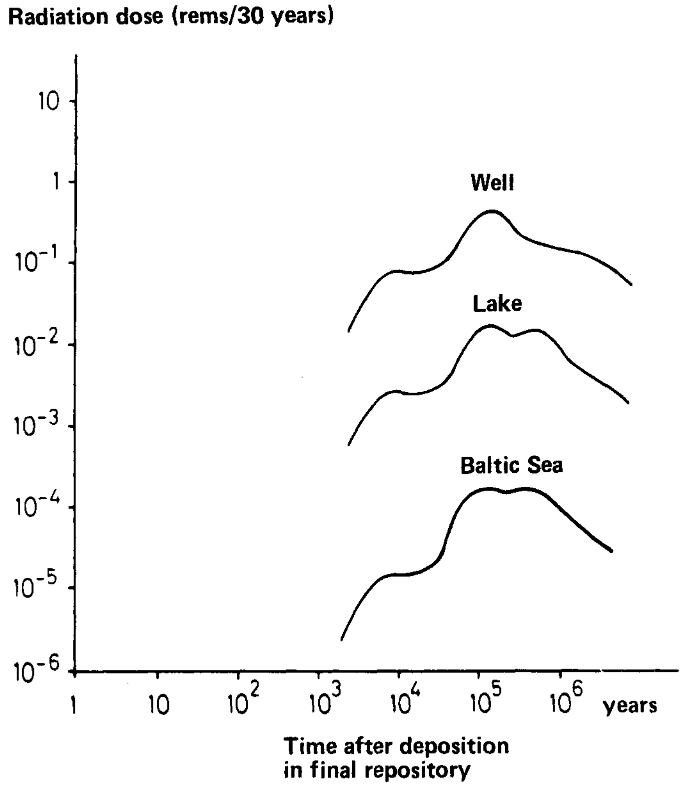


Figure 6-7. Maximum calculated individual doses to critical group (nearby residents) for different primary recipients. The calculations assume a glass leaching period of 30 000 years.

As expected, the case with a well as the primary recipient gives the highest calculated individual doses while the collective dose is nearly identical for different primary recipients. No radiation doses appear until after more than 1 000 years, since the lead-titanium canister remains intact for at least 1 000 years and the transit time for water is at least 400 years. This means that the medium-long-lived nuclides strontium-90 and cesium-137, with half-lives of 28.1 and 30.0 years, respectively, have sufficient time to decay completely. As was mentioned in chapter 3.7, and is evident from Figure 3-9, these nuclides dominate the toxicity of the waste during the first 300 years.

It will further be noted that no calculated radiation doses are obtained from the nuclides plutonium-240, americium-241 and americium-243 with half-lives of 6 760, 433 and 7 650 years, respectively. According to Figure 3-9, these nuclides dominate toxicity during the following 25 000 years. The reason for this is that the elements americium (Am) and plutonium (Pu) are retarded due to sorption effects by factors of 84 000 and 1 100, respectively, in impervious rock in relation to the water (see Chapter 6.5 and Table 6-1). Note, however, that neptunium-237 is a daughter product of americium-241 (see Figure 3-3, chain 4N+1).

The earliest appearing radiation doses are obtained from the long-lived fission products technetium-99 and iodine-129, with half-lives of 210 000 and 17 million years, respectively. These nuclides are not retarded at all; they are dispersed with the transport velocity of the water. These elements produce an initial increase of the calculated radiation doses up to about 6 000 years.

The long-lived fission products zirconium-93 and cesium-135 and all actinides (isotopes of thorium (Th), uranium (U), neptunium (Np) and plutonium (Pu)) are retarded considerably in relation to the water. These elements produce a combined calculated dose maximum after approx. 200 000 years.

In addition to the collective doses given in the tables for the most unfavourable 500-year period, the collective dose for the first 10 000 years has also been calculated. Approximately 30 000 manrems, i.e. 0.1 manrem per MWe and year for 30 years of operation at 10 000 MWe, is obtained for the well case. The largest contribution comes from technetium-99.

6.9.3 Results for the case of initial canister damage

Table 6-10 and Figure 6-8 present the most important calculation results for the case of an initially damaged canister. Only individual doses for a well as the primary recipient have been calculated for this case.

The results show that the calculated maximum radiation doses from the nuclides which leak out from the initially damaged canister are approximately 1/6 000th of the doses obtained for the reference case of slow canister decomposition. The essential diffe-

Table 6-10. Maximum individual doses in critical groups (nearby residents) in the case of initial canister damage. The calculations apply for a well as the primary recipient with a glass leaching period of 3 000 years.

Nuclide	Maximum inflow to recipient		Maximum dose
	Time (years)	Activity (Ci/year)	Nearby residents (rems/30 years)
Zr 93	4×10^{6}	3×10^{-7}	7×10^{-10}
Tc 99	400	5×10^{-3}	7×10^{-5}
I 129[a]	400	1×10^{-7}	2×10^{-6}
Cs 135	5×10^{5}	2×10^{-6}	2×10^{-6}
Ra 226	3×10^{4}	2×10^{-8}	1×10^{-5}
Th 229	6×10^{4}	3×10^{-8}	1×10^{-7}
Th 230	3×10^{4}	2×10^{-9}	8×10^{-9}
U 233	1×10^{5}	5×10^{-6}	9×10^{-6}
U 234	3×10^{4}	4×10^{-6}	7×10^{-6}
Np 237	2×10^{5}	1×10^{-5}	4×10^{-5}
Pu 239	5×10^{5}	1×10^{-11}	3×10^{-11}
	Maximum total dose[b]		7×10^{-5}
	Time of maximum total dose		400 years

a) Based on 1% in the vitrified waste

b) The total dose is not equal to the sum of the above tabulated values, since the maxima for the different nuclides occur at different points in time.

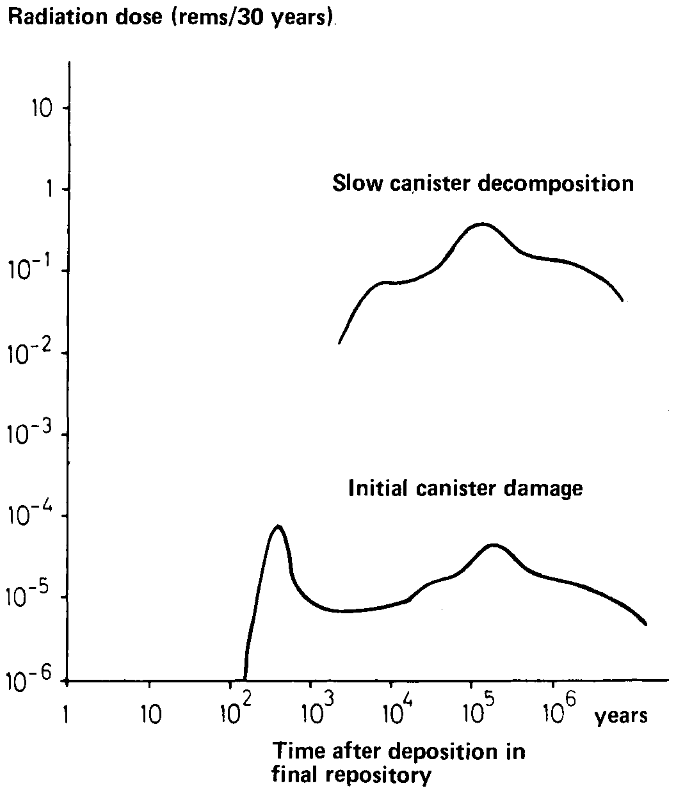


Figure 6-8. Maximum calculated individual doses to critical group (nearby residents) for the two main cases of canister failure. The cases are a) slow decomposition of all canisters during the period from 1 000 to 6 000 years, and b) initial damage to one canister.

rence between these cases is that the earlier leaching of technetium-99 and iodine-129, which are transported at the same velocity as the water, lead to the appearance of low levels of these elements after only about 200 years. After 1 000 years, the slow decomposition of all canisters will contribute doses well in excess of those which originate from the initial canister damage.

The consequences of a few initially damaged canisters will at worst be proportional to their number. In other words, three damaged canisters will give rise to three times as much of a dose increment as one damaged canister. In normal cases, however, the local consequences will not be dependent upon the number of damaged canisters, due to the large extent of the repository and the randomness of the damages.

6.9.4 Analysis of variation

The most important input data, whose values are essential for obtaining the results reported above, are discussed below. As a rule, the case of a well as a primary recipient is used as an example. It has been found that this case gives the highest calculated radiation doses. The different factors are explained in the order followed by the calculation flow chart in Figure 6-5.

The content of radioactive elements

Assuming other conditions to be 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:

- time from discharge of fuel from reactor to reprocessing
- degree of separation of uranium and plutonium as well as certain fission products in reprocessing

It has been assumed in the calculations that 0.1% uranium, 0.5% plutonium and 1% iodine-129 end up in the high-level waste. As is evident from Chapter 3, 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) uranium-234 and radium-226 by about 30%. The plutonium content of the waste stream is of greater importance. The value of 0.5% chosen here is above the expected value as well as values obtained in actual practice. (See Chapter 3.4.)

It is assumed that reprocessing takes place 10 years after discharge of the fuel from the reactor. Of the important nuclides, the level of neptunium-237 (which is formed by neutron absorption and by the decay of plutonium-241, 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) neptunium-237 decreases by about 20%, whereas if reprocessing takes place after many more years, the level of neptunium-237 increases by max. 75% (see Chapter 3.5.)

Life of the canister

A canister life of 1 000 years has been assumed in the reference case. Some of the experts in the reference groups believe that the canister has a service life (i.e. a period during which the canister remains completely intact) of at least 500 years. A comparison between the cases of initial canister damage in Table 6-10 and slow canister decomposition after 1 000 years shows that the main difference at different canister lives is a shift in the time for calculated radiation doses from technetium-99 and iodine-129. The reason is the long water transit time and the retardation of other elements in relation to the water. The nuclides strontium-90 and cesium-137 decay during the time required for transport to the primary recipient.

As long as the canisters are fully intact, they constitute a redundant barrier to the escape of the medium-long-lived waste products. If the canisters remain intact for 100 000 years or more, they also provide a redundant barrier to americium-241, -243 and plutonium-240, -239.

Glass leaching rate

Table 6-11 shows how different assumptions regarding the glass leaching rate affect the maximum inflow of nuclides to the primary recipient. Four different leaching times are shown: 3 000,

Table 6-11 Maximum inflow to primary recipient according to GETOUT runs, with varying leaching times. The table gives the time of the concentration maximum. Canister failure has been assumed to occur after 1 000 years and the inflow is specified per damaged canister.

	T_{leach} = 3000 years		T_{leach} = 30 000 years		T_{leach} = 300 000 years		T_{leach} = 3 000 000 years	
Nuclide	C_{max} (Ci/years)	T_{max} (years)	C_{max} (Ci/years)	T_{max} (years)	C_{max} (Ci/years)	T_{max} (years)	C_{max} (Ci/years)	T_{max} (years)
Zr 93	3.1×10^{-7}	4.0×10^{6}	3.1×10^{-7}	3.9×10^{6}	2.8×10^{-7}	4.1×10^{6}	6.8×10^{-8}	4.5×10^{6}
Tc 99	4.7×10^{-3}	3.7×10^{3}	4.7×10^{-4}	3.8×10^{3}	4.7×10^{-5}	3.8×10^{3}	4.7×10^{-6}	4.0×10^{3}
I 129[a)]	3.8×10^{-7}	3.8×10^{3}	3.8×10^{-8}	3.8×10^{3}	3.8×10^{-9}	3.8×10^{3}	3.8×10^{-10}	3.8×10^{3}
Cs 135	2.4×10^{-6}	4.6×10^{5}	2.3×10^{-6}	4.2×10^{5}	7.4×10^{-7}	5.2×10^{5}	7.4×10^{-8}	5.3×10^{5}
Ra 226	1.7×10^{-8}	2.8×10^{4}	1.1×10^{-8}	5.4×10^{4}	2.8×10^{-9}	2.2×10^{5}	2.8×10^{-10}	2.1×10^{5}
Th 229	2.7×10^{-8}	6.4×10^{4}	2.6×10^{-8}	8.5×10^{4}	1.3×10^{-8}	3.3×10^{5}	1.4×10^{-9}	5.3×10^{5}
Th 230	1.6×10^{-9}	2.8×10^{4}	1.4×10^{-9}	5.4×10^{4}	3.7×10^{-10}	2.2×10^{5}	3.7×10^{-11}	2.1×10^{5}
U 233	4.8×10^{-6}	1.4×10^{5}	3.2×10^{-6}	5.4×10^{4}	1.5×10^{-6}	3.0×10^{5}	1.6×10^{-7}	5.2×10^{5}
U 234	3.8×10^{-6}	2.5×10^{4}	7.2×10^{-7}	3.1×10^{4}	7.2×10^{-8}	3.0×10^{4}	7.2×10^{-9}	3.0×10^{4}
Np 237	1.0×10^{-5}	1.6×10^{5}	8.7×10^{-6}	1.7×10^{5}	1.2×10^{-6}	2.0×10^{5}	1.2×10^{-7}	1.9×10^{5}
Pu 239	1.2×10^{-11}	4.7×10^{5}	4.8×10^{-11}	5.5×10^{5}	5.0×10^{-12}	5.5×10^{5}	5.5×10^{-13}	5.5×10^{5}

a) Based on 3% in the vitrified waste.

30 000, 300 000 and 3 million years. The first two times were used for the cases of initial canister damage and slow canister decomposition. The last value has been calculated in accordance with Chapter 6.3.10 - 11 in order to illustrate possible leaching on the basis of the water flow past the canisters.

In the case of nuclides which are transported as fast as the water (technetium-99 and iodine-129), the inflow is proportional to the leaching rate. In the case of nuclides which are severely retarded compared to the water, a more complex dependence on the leaching rate is obtained. An example is provided by cesium-135, with a half-life of 3 million years and a retardation factor of 840 in impervious rock. The following inflows and changes in inflow are obtained for the four different cases for this nuclide:

Leaching period	Inflow	Change
3 000 years	$2.4 . 10^{-6}$	
30 000 years	$2.3 . 10^{-6}$	- 4%
300 000 years	$7.4 . 10^{-7}$	-68% (factor 3)
3 million years	$7.4 . 10^{-8}$	-90% (factor 10)

A leaching period of 3 million years reduces the inflow and thereby the doses from all of the important nuclides by a factor of 20 or more in relation to the value of 30 000 years used for the reference case. The highest total dose is reduced by a factor of 60. See Table 6-12.

A leaching period shorter than 30 000 years cannot be expected, in view of the limited water flow rate.

Table 6-12 Maximum individual doses in critical group (nearby residents) and maximum collective doses in the case of canister decomposition after 1 000 years. The calculations apply for a well as the primary recipient with a glass leaching period of 3 million years.

	Maximum inflow to recipient		Maximum dose	
Nuclide	Time (years)	Activity (Ci/year)	Nearby residents (rems/30 years)	Collective dose a) (manrems/ MW(e) year)
Zr 93	5×10^{6}	7×10^{-4}	2×10^{-6}	4×10^{-7}
Tc 99	1×10^{4}	5×10^{-2}	7×10^{-4}	3×10^{-5}
I 129 b)	1×10^{4}	1×10^{-6}	2×10^{-5}	6×10^{-5}
Cs 135	5×10^{5}	7×10^{-4}	5×10^{-4}	2×10^{-4}
Ra 226	2×10^{5}	3×10^{-6}	0,002	6×10^{-6}
Th 229	5×10^{5}	1×10^{-5}	7×10^{-5}	4×10^{-5}
Th 230	2×10^{5}	4×10^{-7}	2×10^{-6}	3×10^{-10}
U 233	5×10^{5}	2×10^{-3}	0,003	5×10^{-6}
U234	4×10^{4}	7×10^{-5}	1×10^{-4}	3×10^{-7}
Np 237	2×10^{5}	1×10^{-3}	0,004	5×10^{-5}
Pu 239	6×10^{5}	6×10^{-9}	1×10^{-8}	2×10^{-11}
	Maximum total dose c)		0,007	2×10^{-4}
	Time for maximum total dose		200 000 years	600 000 years

a) Calculated for the most unfavourable 500-year period and 10 000 MWe for 30 years

b) Based on 1% in the vitrified waste

c) The total dose is not equal to the sum of the above tabulated values, since the maxima for the different nuclides occur at different points in time.

Transit time in rock

The transit times in rock for the various nuclides are obtained from the water transit time of 400 years and the retardation factor given by Table 6 1. As concluded in the geology section, it is not improbable that sites may be found with considerably longer water transit times to the primary recipient. A transit time of 10 000 years is mentioned as a reasonable value.

The retardation factors which are used are based on measurements. Due to certain circumstances in connection with these measurements, the retardation factors for the elements plutonium and neptunium are probably too low (see 6.5).

Table 6-13 illustrates the effects of different assumed nuclide transit times for the interval between 1 000 and 1 million years. Drastic changes in the calculation results are only obtained for those nuclides for which the transit time in the reference case is substantially longer than half-life of the nuclide. In order to illustrate this, calculated values for relatively short transit times for the nuclides plutonium-239, plutonium-240, americium-241 and americium-243 are included in the table. The transit time for plutonium in the reference case is 400 000 years and for americium 34 million years. Thus, these elements are effectively retained by the rock. In order for appreciable calculated doses to be obtained from the americium isotope, their transit time must be shorter by a factor of at least 1 000. A reduction of the transit time by a factor of 10 is required for plutonium. Avail-

Tabell 6-13 Parameter study of individual doses in critical groups in the case of slow canister decomposition for various total nuclide transports to the primary recipient. Well alternative. Glass leaching period 30 000 years.

Nuclide	Maximum weighted whole-body dose (rems/30 years)				
	10^3 år	10^4 år	3×10^4 år	10^5 år	10^6 år
Tc 99	$\underline{6 \times 10^{-2}}$	$\underline{6 \times 10^{-2}}$	6×10^{-2}	5×10^{-2}	2×10^{-3}
I 129[a]	$\underline{2 \times 10^{-3}}$	$\underline{2 \times 10^{-3}}$	2×10^{-3}	2×10^{-3}	2×10^{-3}
Cs 135	5×10^{-2}	5×10^{-2}	5×10^{-2}	$\underline{5 \times 10^{-2}}$	$\underline{4 \times 10^{-2}}$
Ra 226	8×10^{-2}	0.2	$\underline{0.5}$	$\underline{0.9}$	0.2
Th 229	8×10^{-3}	3×10^{-2}	$\underline{0.1}$	$\underline{0.3}$	0.8
Th 230	1×10^{-3}	2×10^{-3}	$\underline{4 \times 10^{-3}}$	$\underline{8 \times 10^{-3}}$	2×10^{-3}
U 233	1×10^{-2}	2×10^{-2}	$\underline{5 \times 10^{-2}}$	$\underline{0.1}$	$\underline{0.3}$
U 234	5×10^{-3}	5×10^{-3}	$\underline{5 \times 10^{-3}}$	$\underline{4 \times 10^{-3}}$	$\underline{5 \times 10^{-4}}$
Np 237	0.7	0.7	0.7	$\underline{0.7}$	$\underline{0.5}$
Pu 239	4	4	3	$\underline{0.5}$	$\underline{3 \times 10^{-7}}$
Pu 240	3	1	0.2	$\underline{2 \times 10^{-4}}$	$\underline{3 \times 10^{-9}}$
Am 241	0.4	0.1	3×10^{-2}	1×10^{-4}	$\underline{0}$
Am 243	13	5	1	3×10^{-3}	$\underline{4 \times 10^{-7}}$

a) Based on 1% in the vitrified waste

NOTE: Hypothetical nuclide transit times. Underlined values indicate intervals for expected values on the basis of calculated transit time. The dose values in this table are not based on the inflows calculated by means of GETOUT. They therefore deviate considerably in many cases from the corresponding values in Table 6-7, since decay or generation from parent nuclides during the transit period are not taken into account.

able experimental data indicate that the transit times given above for plutonium are, if anything, on the low side.

Primary recipient

The difference between different primary recipients is illustrated by Tables 6-7, 6-8 and 6-9 and by Figure 6-7. The maximum calculated individual dose for the well case is approximately 20 times higher than for the lake case, which is in turn approximately 100 times higher than for the Baltic Sea case. The well case is considered to be rather improbable. If the final repository is situated near the coast, the Baltic Sea case is the most likely. If, on the other hand, the repository is located inland, it should be possible to orient the repository so that the primary recipient is a lake.

Paths of exposure

The predominant paths of exposure in a well case are via drinking water (neptunium-237, uranium-233, radium-226 and technetium-99). Fish is the dominant path for cesium-135, while iodine-129 is ingested via meat, drinking water and milk. Meat also contributes part of the dose from neptunium-237. In the lake case, the dose from fish consumption is dominant for a number of nuclides (see Chapter 6.8). Fish consumption has been assumed to be 50 kg/year, which is more than twice the average consumption in Sweden. The choice of concentration factors for different nuclides is based on available data from the literature. The concentration factors are variable, depending on the type of lake (salt content and trace metals) and the type of fish. For cesium, the concentration factor varies between 500 and 10 000. A figure of 2 000 has been chosen for the calculations. For radium, concentration factors in marine environments vary between 15 and 60. Swedish results for lakes indicate a concentration of less than 10. A value of 15 has been used in the calculations for lake water and 50 for seawater.

It would not seem that changes in dietary habits and life style could lead to an increase in individual radiation doses, but collective doses could increase if a community is established near the point of inflow to the primary recipient.

The effects of drying-up of the recipients (including the Baltic Sea), which can occur as a result of land uplift, have been taken into account. Thorium-229 and cesium-135 sediment to some extent and can lead to some increase in the radiation dose. (See chapter 6.8.)

Calculation verification

Many of the most important results of the computer runs with the computer programs mentioned in Figure 6-5 have been checked by means of calculations using independent methods. Good agreement has been obtained between the results. A simple check can be carried out using the method illustrated in Table 6-14.

The first line gives the emission rate obtained from tables in /3-1/. Knowing the leaching rate per year and the nuclide transit time, it is then possible to calculate the inflow to the primary recipient. This is given in the second line, and can then be compared with corresponding values from the GETOUT model. For certain nuclides, e.g. technetium-99 and iodine-129, good agreement is obtained, while for others, such as radium-226, the difference is considerable. This is due to the fact that the parent nuclides thorium-230 and uranium-234 are retarded considerably more than radium-226. As a result, all of the radium-226 does not reach the primary recipient, but rather only that portion which comes from thorium which is present in or near the recipient.

The concentration and intake per year can be calculated on the basis of the inflows and the amount of dilution in the recipient. The individual doses (weighted whole-body dose) are obtained with the aid of the tables showing the relative importance of the paths of exposure /6-4, 6-5 and 6-6/ and the dose conversion factors (Table 6-2). These doses are then compared with the BIOPATH results from Tables 6-7 and 6-8.

6.9.5 Conclusions

The preceding analysis shows that the following factors are of great importance for preventing the slow dispersal of radioactive substances from the final repository:

Table 6-14. Table of simplified calculations of radiation doses to nearby residents for the most dominant nuclides.

	NUCLIDE						
Parameter	Tc 99	I 129	Cs 135	Ra 226	Th 229	U 233	Np 237
Emission rate for 10 000 canisters (max. within 30 000 years), Ci (see ref. /3-1/)	140 000	3.8a)	2 500	20	500	750	6 400
Leaching time 30 000 years (max. inflow into recipient, Ci/year) (see chap. 6.3)	4.7	1.3×10^{-4}	0.083	6.7×10^{-4}	0.017	0.025	0.21
Max. inflow obtained from GETOUT, Ci/year (see Table 6-7)	4.7	1.3×10^{-4}	2.3×10^{-2}	1.1×10^{-4}	2.6×10^{-4}	3.2×10^{-2}	8.7×10^{-2}
To recipient area (50%)	2.4	6.5×10^{-5}	1.2×10^{-2}	5.5×10^{-5}	1.3×10^{-4}	1.6×10^{-2}	4.4×10^{-2}
Recipient	WELL	WELL	LAKE (+WELL)b)	WELL	WELL	WELL	WELL
- dilution, m^3/year (see Chap. 6.7)	5×10^5	5×10^5	2.5×10^7	5×10^5	5×10^5	5×10^5	5×10^5
- concentration Ci/m^3	4.7×10^{-6}	1.3×10^{-10}	4.8×10^{-10}	1.1×10^{-10}	2.6×10^{-10}	3.2×10^{-8}	8.7×10^{-8}
Path of exposure	WATERc)	WATER	FISHd)e)	WATER	WATER	WATER	WATER
- relative importance, % (see Tables 6-4, 6-5)	49	26	65b)	60	88	83	78
- level Ci/kg	4.7×10^{-9}	1.3×10^{-13}	9.6×10^{-10}	1.1×10^{-13}	2.6×10^{-13}	3.2×10^{-11}	8.7×10^{-11}
Intake Ci/year	2.1×10^{-6}	5.7×10^{-11}	4.8×10^{-8}	4.8×10^{-11}	1.1×10^{-10}	1.4×10^{-8}	3.8×10^{-8}
Dose conversion factor, rems/Ci (see Table 6-2)	5.5×10^2	3.4×10^5	7.3×10^3	2.8×10^7	3.4×10^5	1.1×10^5	2.0×10^5
Individual dose,							
rem/year	1.2×10^{-3}	1.9×10^{-5}	3.5×10^{-4}	1.3×10^{-3}	3.7×10^{-5}	1.5×10^{-3}	7.6×10^{-3}
rem/30 years	3.6×10^{-2}	5.7×10^{-4}	1.0×10^{-2}	3.9×10^{-2}	1.1×10^{-3}	4.5×10^{-2}	0.23
Total individual dose (incl. other paths of exposure) rem/30 years	0.07	2.2×10^{-3}	1.6×10^{-2}	0.07	1.3×10^{-3}	0.05	0.29
Calculation with BIOPATH, rem/30 years (see Tables 6-7 and 6-8)	0.07	2.2×10^{-3}	2.0×10^{-2}	0.07	1.4×10^{-3}	0.06	0.30

a) based on 1% in waste

b) exposure via fish from lake (65%) and via well water (35%)

c) consumption 440 l/year

d) consumption 50 kg/year

e) concentration factor 2 000

- impervious rock with a low rate of water flow
- long flow time for the water to the nearest primary recipient
- sorption of certain radioactive elements in rock fissures with flowing water
- low leaching rate for the waste glass
- long service life of lead-titanium canister
- favourable geographic and topographic location to provide adequate dilution of the substances which eventually leak out to the primary recipient

The analysis also shows that all of these conditions do not have to be fulfilled simultaneously. A series of conservative assumptions have been made for the reference case which has been analyzed. These assumptions lead to an overestimate of the calculated radiation doses. The most important of these assumptions are:

- the glass leaching rate has been calculated on the basis of experimental studies with virtually unlimited water flow. Leaching in the final repository is expected to be substantially lower due to a low rate of water flow through the repository. Preliminary analyses indicate an overestimate by a factor of approximately 100 of the quantity of dissolved elements per year (see section 6.3)
- retardation of the elements neptunium and plutonium has been calculated using retardation factors which are probably at least a factor of 10 less than what can be expected in rock fissures (see Section 6.5)
- the water flow time in impervious rock from the final repository to the primary recipient has been set at 400 years. Age determinations carried out on groundwater samples indicate a flow time which is considerably less. The value of 10 000 years has been discussed as a reasonable value in the appropriate siting of the final repository
- the postulated primary recipient, a well, is assumed to be situated in such a manner that a relatively limited dilution of escaping elements is obtained.

In order to explore the degree of overestimation of the calculated doses for the well case, a calculation has been carried out using a glass leaching time of 3 million years, a water flow time of 10 000 years and retardation factors for neptunium and plutonium increased by a factor of 10. The results are shown in Table 6-15 and should be compared with Table 6-7 and 6-12.

Table 6-15. Maximum individual doses in critical group (nearby residents) in the case of canister decomposition after 1 000 years. The calculations apply to a well as the primary recipient with a glass leaching period of 3 million years, a water transit time of 10 000 years and a delay factor of 10 for neptunium and plutonium.

	Maximum inflow to recipient		Maximum dose
Nuclide	Time (years)	Activity (Ci/year)	Nearby residents (rems/30 years)
Zr 93	-	0	0
Tc 99	1.6×10^{4}	5×10^{-2}	7×10^{-4}
I 129a)	1.6×10^{4}	1×10^{-6}	2×10^{-5}
Cs 135	8×10^{6}	2×10^{-5}	2×10^{-5}
Ra 226	6×10^{5}	9×10^{-7}	5×10^{-4}
Th 229	3×10^{6}	6×10^{-6}	3×10^{-5}
Th 230	6×10^{5}	1×10^{-7}	6×10^{-7}
U 233	5×10^{6}	8×10^{-4}	0.001
U 234	5×10^{5}	2×10^{-5}	4×10^{-5}
Np 237	3×10^{6}	3×10^{-5}	9×10^{-5}
Pu 239	-	0	0
	Maximum total dose b)		0.001
	Time of maximum total dose		5 million years

a) Based on 1% in the vitrified waste.

b) The total dose it not equal to the sum of the above tabulated values, since the maxima for the different nuclides occur at different points in time.

The following conclusions can be drawn from the reported analysis:

1 Slow dispersal of the radioactive substances from a final repository cannot give rise to any radiation doses to nearby residents until after at least 1 000 years.

2 The maximum calculated radiation dose to a nearby resident does not exceed 0.4 rem in 30 years. The maximum dose will probably be less by a factor of at least 100.

3 The maximum calculated dose to nearby residents does not occur until after 200 000 years or more.

4 Initial damage to a few isolated canisters does not give rise to any measurable radiation doses.

5 The collective dose to the total population of the earth during the most unfavourable 500-year period is calculated to be less than 0.01 manrem per MWe and year (of reactor operation).

7 INFLUENCE OF EXTREME EVENTS

7.1 BEDROCK MOVEMENTS

7.1.1 General

Bedrock movements of appreciable magnitude could damage a final repository by creating new paths for the groundwater flow and/or by damaging the canisters. Limited canister damage does not alter the premises of the safety analysis, since a case with initial canister damage has been considered.

A number of investigations have been conducted within the KBS project for the purpose of determining the probability of bedrock movements which could affect the safety of a final repository.

Seismic activity in Sweden is very low, and there has never been an earthquake which has caused major damage, as far as is known.

The faults which have been observed in the Swedish bedrock are largely the result of the tectonic and seismic events of approximately 1 800 million years, where the movement from millennium to millennium has been on the order of a few mm. However, larger fault movements have been observed and reported in areas with special zones of movement /7-1, 7-2, 7-3/, for example in northwest Skåne and in Norrbotten County. The land uplift which followed the retreat of the inland ice cap and is still proceeding is probably the primary cause of these recent bedrock movements /7-4/.

Bedrock movements are discussed in greater detail in Chapter II:7.

The possibilities of a new glaciation are discussed in Chapter II:7.7. It is concluded that a new ice age could not affect the integrity of the final repository.

7.1.2 Earthquakes

The Department of Seismology at the University of Uppsala was commissioned by KBS to compile all available data on Swedish earthquakes /7-5/. The data were obtained from previous reviews by Båth /7-11/, which were based on the subjective observations of the population, and from the instrumental observations of recent years. These latter observations, which are presented in

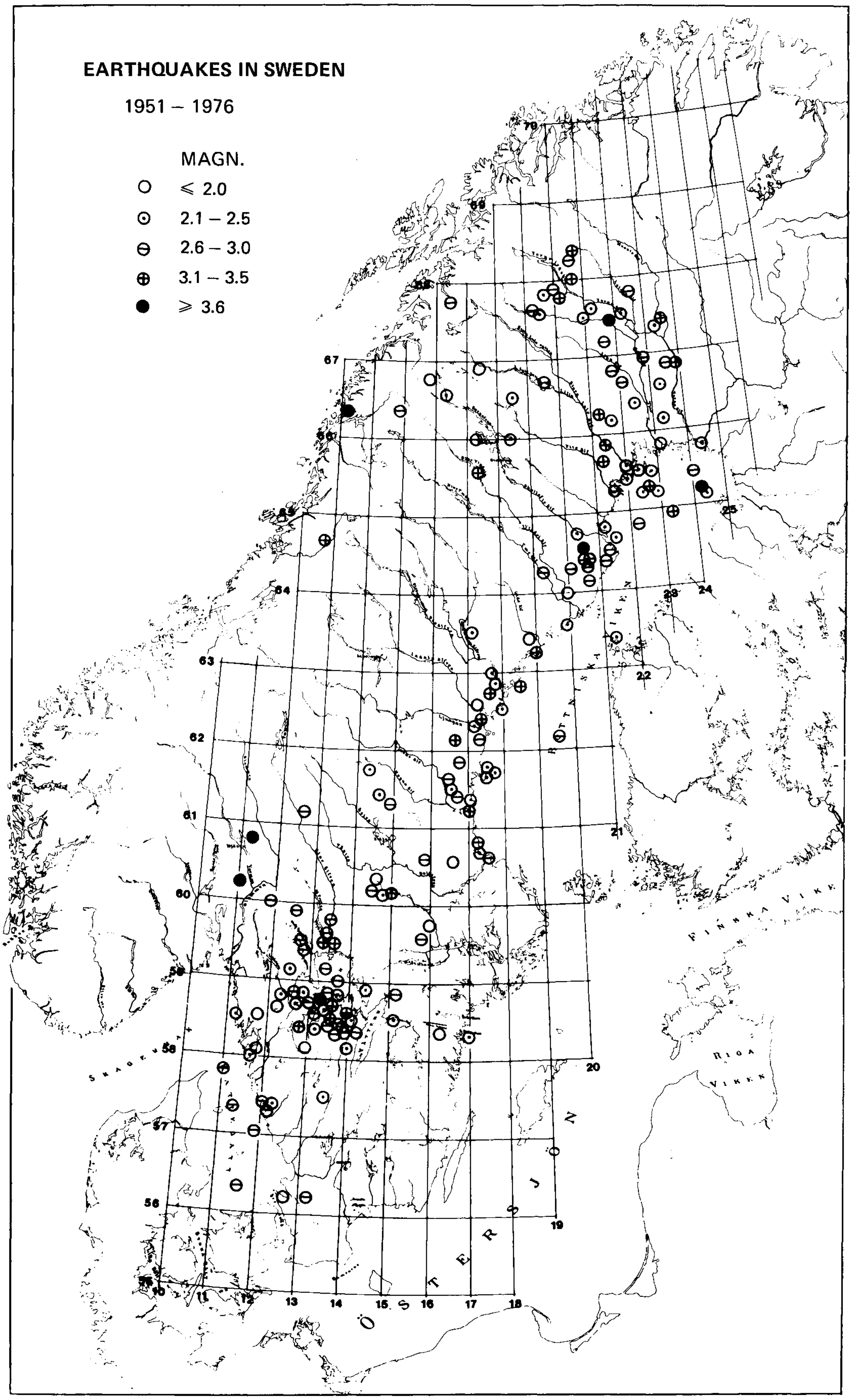


Figure 7-1. Earthquakes registered in Sweden during the period 1951–1976. (Professor Markus Båth, Department of Seismology at the University of Uppsala).

Figure 7-1, show that seismic activity in Sweden is concentrated within certain belts. Outside of these belts, there are large areas where no seismic activity at all has been observed. Magnitudes greater than 3.5 are rare, even within the most active areas.

The following relationships between magnitude and displacement is approximated in /7-5/:

Magnitude	Displacement
3.5	0.3 cm
4.0	0.6 cm
4.5	1.5 cm
5.0	3.6 cm

7.1.3 Occurrence of new cracks and consequences hereof

The Swedish bedrock exhibits a pattern of fracture zones of varying size /7-3/. As is mentioned in Chapter II:7, geological observations show that new fractures and faults will occur in already existing joint planes. This is true regardless of whether the external forces which cause the movements are of tectonic or glacial origin. To put it simply, the general rule of the weakest link in a chain breaking applies here as well. The stress and deformation conditions existing in a rock mass which is subjected to external forces have been illuminated in a study which confirms these conclusions /7-7/.

An investigation concerning the probability that a final repository will be affected by a fault movement was commissioned by KBS /7-6/. This study concludes that this probability for a repository one square kilometre in size is less than 10^{-9} per year for the country, on the average.

A special study has analyzed how vertical rock displacements of varying magnitude affect long canisters which are stored horizontally in clay-filled tunnels /7-8/. If the canisters are stored vertically in boreholes, as is proposed here, the effects of such displacements on the stresses in the canister material will be reduced.

7.1.4 Importance for the safety of the final repository

In summary, the studies carried out by KBS of bedrock movements which could have an adverse effect on the safety of the final repository have shown that:

- the probability of such movements in the Swedish bedrock is very low
- within areas which are surrounded but not intersected by fracture zones, the probability of new flow paths (cracks in the rock) opening is extremely low
- sections of rock which are found during the construction of final repository to have a high fracture content should not be utilized for storage
- neither the proposed buffer layer nor the canister will be damaged, even if an earthquake of considerable proportions

by Swedish standards should be sustained by the final repository.

The probability of damage to part of the final repository as a result of bedrock movements is thus extremely low, somewhere around once in several million years. If such damage should nevertheless occur, it will probably affect only a few per cent of the canisters. The consequences of such damage are deemed to be no more serious than the consequences of slow canister decomposition.

7.2 METEORITE IMPACTS

If a meteorite should hit the surface of the earth directly above a final repository, a crater would be created which could weaken the geological barrier or, at worst, eliminate it completely.

According to Hartman /7-9/, who has studied meteorite impacts which have occurred over the past 2 000 million years, the probability of a meteorite hitting a given area of 1 km^2 and creating a crater with a depth of about 100 m is in the neighbourhood of 10^{-13} per year. Historical experience also confirms the assumption that a meteorite impact is not a risk which need be considered in this context.

7.3 ACTS OF WAR AND SABOTAGE

In the long time perspective which must be applied to the final repository, acts of war cannot be considered to be "extreme events". On the other hand, the possibility that acts of war might lead to serious consequences for the safety of a terminally sealed final repository at a depth of some 500 metres in the Swedish bedrock must be considered to be remote.

Ground detonations of nuclear devices of 10-15 megatons create craters in the rock with a depth of roughly 110 - 180 m /7-10/. Thus, the geological barrier would not be broken through, but may well be weakened. In such a situation, however, this would be of subordinate importance, since any release of radioactivity from the final repository would represent only a fraction of the radioactivity caused by the bomb, which would remain in the area for a long period of time.

Wartime damages to the final repository and the encapsulation station during the deposition stage are, of course, a possibility. But the probability is low, since these facilities are not likely to be primary targets for military actions. Furthermore, the consequences of bomb hits and similar occurrences will be limited compared to the situations which would otherwise be the result of such acts of war.

Safeguards against sabotage as described in Section III:7.4 will be provided during the intermediate storage, encapsulation and final deposition stages. After the final repository has been closed and sealed, effective acts of sabotage are impossible.

Compared to other installations which experience has shown to be targets for sabotage in terrorist actions, the facilities described here are less attractive to potential saboteurs and are roughly comparable to other industrial plants where environmentally hazardous material is handled.

7.4 FUTURE DISTURBANCE BY MAN

It is conceivable that the knowledge of where the final repository is located may be lost in the distant future and that man at that time may, for some reason, perform drilling or rock work which results in contact with the waste. Since the final repository is situated in one of our most common types of rock which does not contain any valuable minerals, it is not likely that the site will be considered for mining operations. The depth and low water content of the impervious rock selected for this purpose makes it highly improbable that deep wells will be drilled for water at this site in the future. No reason can be seen for seeking out such great depths for the construction of rock storage vaults or the like.

The loss of knowledge of the location of the final repository would presuppose that our current civilization would be destroyed as a result of some catastrophic event such as a global war of extermination or a new ice age. If the country is then repopulated again, the risks mentioned here would arise, but only after the new population had achieved a level of technological development which permitted advanced rock work. In such a case, it is probable that such a civilization would also possess the capability to detect the radioactivity in the final repository and would act accordingly to avoid damage to the repository.

8 SUMMARY SAFETY EVALUATION

8.1 HANDLING, STORAGE AND TRANSPORTATION OF SPENT FUEL AND VITRIFIED WASTE

Spent fuel and vitrified waste will be handled, stored and transported in accordance with international and national standards and regulations.

A good deal of experience has been gained both in Sweden and abroad in the handling and storage of spent fuel.

It is assumed that the central storage facility for spent fuel will be situated in a rock cavern with approximately 30 m of rock cover. The facility is designed with the primary objective of minimizing radiation doses to the personnel and preventing the release of radioactive substances to the environment.

The various measures which are adopted to ensure the radiological safety of the personnel - such as carefully designed radiation shielding, remote-controlled handling and ventilation, systems for monitoring direct radiation and airborne activity etc. - permit handling and storage to be affected with adequately low dose loads.

The fuel storage pools are designed as free-standing, thick-walled units of concrete, lined with stainless steel. The cooling system is designed to maintain the temperature at 25-30°C. A number of back-up systems for cooling and a make-up water system can maintain cooling in the event of a failure of the primary system.

The consequences of a dropped transport cask or fuel cassette have been analyzed and found to give rise to doses of less than 0.1 mrem in the environment, with a probability of about 0.0004 per year.

Normal activity releases during operation consist primarily of small quantities of tritium and krypton-85 and give rise to insignificant radiation doses (0.0001 mrem/year).

Vitrified high-level waste will be handled and stored in accordance with the principles which have been established in practice at the Marcoule plant in France, from which good experiences have been obtained.

Spent fuel and vitrified high-level waste will be transported in accordance with international regulations in casks which can withstand severe transport accidents. The consequences of hypothetical leakage in connection with such severe accidents and failures have been analyzed. The probability of the most severe type of accident, ship collision with fire, has been calculated to be about $3 \cdot 10^{-6}$ per year.

The probability of such an accident with severe consequences is even lower. The consequence of a release has been calculated to be max. 6 rems for the maximally exposed individual, and a collective dose of max. approx. 30 000 manrems if the release takes place in the vicinity of a large city. This means that no acute radiation damage will be sustained and there will be a risk for max. 6 cases of radiation-induced cancer, even in this severe hypothetical case.

8.2 LONG-TERM EFFECTS OF THE FINAL STORAGE OF VITRIFIED HIGH-LEVEL WASTE

The high-level waste from the reprocessing of spent nuclear fuel is isolated by means of vitrification and encapsulation in a lead-titanium canister which is emplaced in sound rock at a depth of 500 m embedded in a buffer material (90% quartz sand and 10% bentonite). The safety analysis of such a final disposal reveals the following.

During the period of at least 1 000 years during which the lead-titanium canister is fully intact, the elements strontium-90 and cesium-137 decay almost completely. Initially undetected damage to one or two canisters would not give rise to any measurable increase in the radiation level. The slow decomposition of the canister could lead to a slight increase of the radioactivity level in the environment after several thousand years. This slight increase would stem mainly from the nuclides neptunium-237, technetium-99, radium-226 and uranium-233 as well as cesium-135 and iodine-129. The elements plutonium and americium are retained in rock fissures etc.

For the severest identified case - a deep drinking water well in the vicinity of the final repository - it is calculated that the individual dose in the future could increase by max. 0.4 rem over a 30-year period, which would be reached after around 200 000 years.

The calculated maximum dose loads (see Figure 8-1) are much lower than the maximum permissible radiation dose recommended by the ICRP 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 load will be approximately equal to the target value recommended by the National Institute of Radiation Protection as the goal which should be aimed at in the design of nuclear power plants.

The most probable dose load is less than one percent of the above-specified maximum value. This is due to the fact that the decomposition of canisters and leaching of glass at the low water flows which are found at 500 m down in impervious rock can be expected to take place at a considerably slower rate than has been

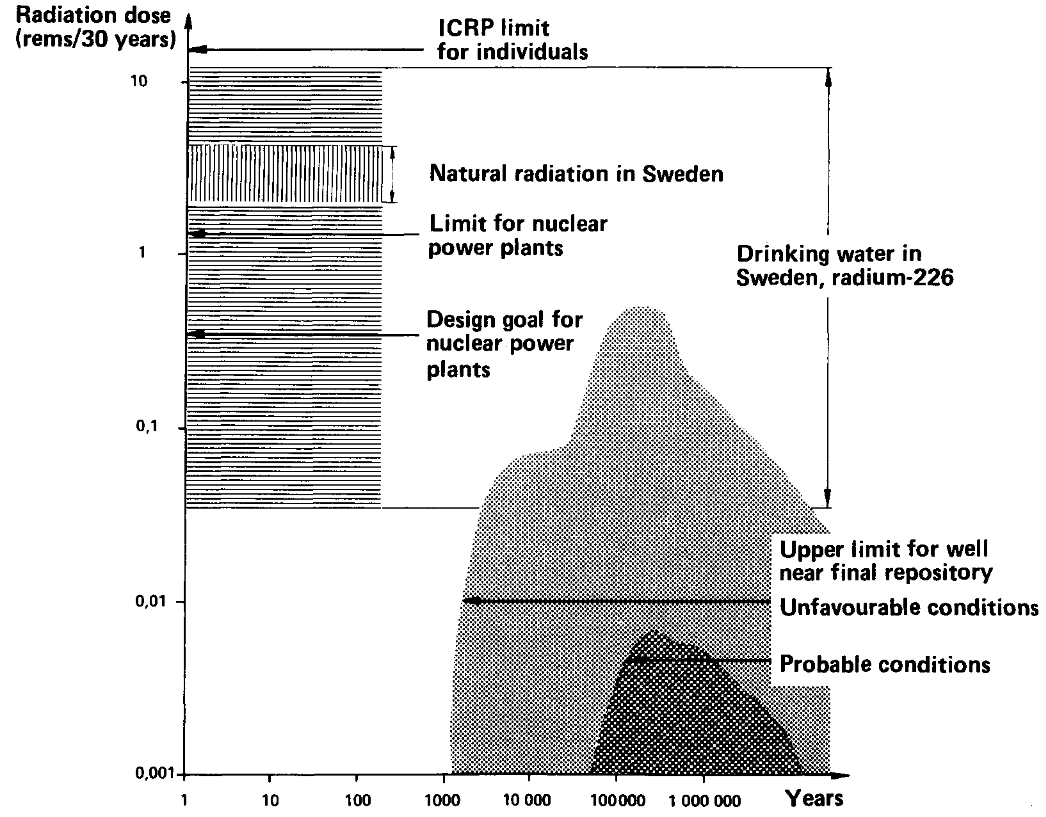


Figure 8-1. Calculated upper limit for radiation doses to people who live near the final repository (critical group). The calculations pertain to the slow decomposition of the canister with a well as the primary recipient. For purposes of comparison, the dose load from several natural radiation sources as well as a number of established dose limits have also been plotted in the diagram.

assumed in the calculations. Furthermore, the retardation factor for neptunium and the water transit time used in the calculations have both been conservatively chosen.

The regional and global dose load to large populations groups has been calculated for the most unfavourable 500-year period in the future. In the very long run, a maximum 500-year dose of around 2 000 manrems can be obtained, which corresponds to 0.0007 manrem per MWe and year of operation.

Even for the most unfavourable cases with conservatively chosen data in the calculations, the health risks are extremely low, or nil. The risk for nearby residents can be illustrated by means of the following table, which gives the calculated increase of the cancer risk.

Recipient	Present risk in Sweden	Risk increase due to leaching from final repository	
		probable	maximum
Well	0.08	1×10^{-6}	0.0001
Lake	0.08	4×10^{-8}	4×10^{-6}
Baltic Sea	0.08	4×10^{-10}	4×10^{-8}

The calculated collective doses correspond to 0.4 cases of cancer and 0.4 cases of genetic defects for the entire population of the world over a period of 500 years. The present number of deaths due to cancer in Sweden is approximately 20 000 per year. Of all births, only approximately 3% exhibit natural genetic defects, which means a current level of 3 000 cases per year in Sweden. The values specified for medical effects are calculated on the basis of the internationally accepted relationships between radiation dose and maximum medical effects. This may, however, lead to an overestimation of the actual medical effects at the low dose values and dose rates which apply here.

The calculated increase in the level of radioactive elements in the recipients to which the waste products could possibly be dispersed are comparable to the natural levels of such elements. Neptunium-237 can be compared with uranium and cesium with potassium. Table 8-1 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.

Table 8-1. Levels of radioactive elements in water.

Radioactive element	Levels in natural water (pCi/l) in Sweden		Maximum calculated increase in level in primary recipients near the final repository a) (pCi/l)	
	Drinking water	Sea-water b)	Well	Lake
Radium-226	0.1-40	0.3	0.1	0.002
Uranium	0.1-1 500 c)	3	30	0.6
Neptunium-237	-	-	90	2
Potassium-40 d)	ca 20	330	-	-
Cesium-135 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) Potassium-40 and cesium-135 are biologically comparable, but have slightly different dose factors (24 000 as compared to 7 300 rems/Ci, respectively).

Figure 8-1 also presents the range of variation for the radiation doses which can be obtained from radium-226 in drinking water in Sweden /2-7/. These radiation doses have been calculated using the same dose factor which is used elsewhere in the report.

Even in the case where a series of unfavourable assumptions have been made, the calculated changes in the radiation environment are considerably less than normal naturally occurring variations. These natural variations do not have any currently demonstrable effect on either man or the ecological systems. The calculated maximum values for radiation doses caused by leakage from a final repository are below the limit value for nuclear energy installations specified by the radiation protection authorities. The proposed method for the final storage of high-level waste glass is therefore judged to be absolutely safe.

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