Analysis of a Post-Closure Safety Assessment Methodology for Radioactive Waste Disposal Systems in South Africa

by

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CHAPTER 1

INTRODUCTION

1.1 GENERAL

All users of radioactive materials in the nuclear industry, agriculture, research and medicine generate waste, much in the same way as other human activities. The largest volumes of radioactive waste, however, are produced by activities related to the nuclear fuel cycle. In this cycle, radioactive waste is a by-product from the mining and milling of uranium ores, the uranium enrichment and conversion process, and nuclear fuel fabrication. In addition to the nuclear fuel cycle, waste generated from the operation of nuclear reactors, reprocessing of spent fuel, and decommissioning of nuclear facilities contribute considerably to the total volume of radioactive waste.

Radioactive waste, like all other waste, does not have any economic value and must ultimately be disposed. Although radioactivity is a natural phenomenon that decreases exponentially with time, radioactive waste can pose considerable risks to the natural environment of the earth. Special precautions must therefore be taken with the waste, from its creation to its disposal—even to the time the activity has decayed to the natural level—by public demand. This is clearly a management problem, hence the name *radioactive waste management.* In its broadest sense, the term includes the activities of the waste generators, the operators (managers) of waste disposal facilities and regulatory authorities if the waste is subject to regulatory control.

During the first decade of the nuclear era, scientists and licensing authorities essentially handled the problem of radioactive waste management, since they were directly confronted with the necessity to develop reasonable and safe solutions. After decades of research, study and testing, there exists today a large number of technological solutions with which radioactive waste can be managed safely (IAEA, 1992). However, radioactive waste management and waste disposal, in particular, are no longer matters for scientists alone, but requires the co-operation of scientists, politicians, licensing authorities, industry and the public at large. The international community has consequently developed a set of principles, summarized in Table 1–1, through the International Atomic Energy Agency (IAEA) with the main objective:

'to deal with radioactive waste in a manner that protects human health and the

Table 1-1 Fundamental principles agreed upon by the international community for the management of radioactive waste (IAEA, 1995*a*).

Principle	Description
Protection of Human Health	Radioactive waste shall be managed in such a way as to secure an acceptable level of protection to human health.
Protection of the Environment	Radioactive waste shall be managed in such a way as to secure an acceptable level of protection to the environment.
Protection beyond National Borders	Radioactive waste shall be managed in such a way as to assure that possible effects on human health and the environment beyond national borders will be taken into account.
Protection of Future Generations	Radioactive waste shall be managed in such a way that predicted impacts on the health of future generations will not be greater than the relevant levels of impact that are acceptable today.
Burdens on Future Generations	Radioactive waste shall be managed in such a way that it will not impose undue burdens on future generations.
National Legal Framework	Radioactive waste shall be managed within an appropriate national legal framework including clear allocation of responsibilities and provision for independent regulatory functions.
Control of Radioactive Waste Generation	Generation of radioactive waste shall be kept to the minimum practicable.
Radioactive Waste Generation and Management Interdependencies	Interdependencies among all steps in radioactive waste generation and management shall be appropriately taken into account.
Safety of Facilities	The safety of the facilities for radioactive waste management shall be appropriately assured during their lifetime.

environment, now and in the future, without imposing undue burdens on future generations.' IAEA (1995a).

However, it is clear that these principles can only be implemented successfully if member countries create their own national legal frameworks, and associated organizational structures, to dispose and manage their radioactive wastes appropriately (IAEA, 1995*a*).

The majority of countries in the world today require that the long-term safety of any hazardous waste disposal site must be demonstrated convincingly prior to its implementation. However, to achieve this one must have a framework, also known as the *disposal system*, that describes the disposal method and site in detail (AECL, 1994*a*). This includes the potentially affected geology and accessible environment (e.g. air, land, water, people, plant and animal life) surrounding the site. This procedure is commonly referred to as a *safety or performance assessment* of the site, which will now be described in more detail.

1.2 DEFINITION OF THE TERM SAFETY ASSESSMENT

According to Cho *et al.* (1990), the general objective of a safety assessment should be to determine what impact the disposed waste would have on individuals and their environment as a function of time. This implies that one must determine how radioactive materials may escape from the disposal site and along which paths can it migrate and what effect it will ultimately have on human beings. This exercise is generally referred to as a *safety assessment.*

The main idea behind a safety assessment is clearly to investigate, quantify and explain the effects that a proposed (or selected) radioactive waste disposal system will have on its surroundings. Although considerable attention has been given to this problem over the years, there does not seem to be a universal view of what should be done in this regard, how it should be done and for what reason.

The fundamental principles for radioactive waste management in Table 1-1 can be used to divide the time during which a radioactive waste disposal site will remain active into two periods.

(a) The *pre-closure period* – The time from the moment the site was developed until it was closed and perhaps a number of years after that. This period is sometimes also referred to as the operational period.

(b) The *post-closure period* – This period should theoretically begin the moment the site is closed and extend to the time that the waste does not pose any further thread to humanity and the environment. For this purpose, the post-closure period can be further divided into institutional and post-institutional control periods. However, the emphasis in the fundamental principles of Table 1-1, is clearly more centred on later times (cf. the reference to future generations). The term will consequently be interpreted here as an unspecified period beginning at a time when nobody is interested in maintaining the site, or know of the site.

This division suggests that the safety assessment of the site can also be divided into two phases—a *pre-closure safety assessment* and a *post-closure safety assessment.* Since there will be people that operate and manage the site during the pre-closure period, but not necessarily during the post-closure period, a pre-closure assessment may differ considerably from a post-closure assessment. For example, the operators of the site will certainly be able to monitor and investigate the site and take corrective actions where necessary. Risks to humans during the pre-closure period are also different than during the post-closure period. For example, the risk is zero that a container will drop on an operator or that workers will be exposed directly to radiation after the site is closed. A pre-closure assessment of the site could therefore be based on sound scientific principles and procedures, but not a post-closure assessment. In this case, one will have to rely mainly on assumptions and guesses of what the situation will be in the post-closure period. Since a considerable amount of work has already been done on the pre-closure safety assessment of radioactive waste disposal sites this thesis will concentrate exclusively on the post-closure safety assessment. However, this does not suggest that the two *periods* should be treated independently. In fact, a major disadvantage of previous safety assessment analyses in South Africa is that no attention was paid to the influence that pre-closure and operational activities might have on a post-closure safety assessments of the sites.

The IAEA (1993*a*) defines performance assessment formally as:

> *'...an analysis to predict the performance of a disposal system, or sub-system, followed by a comparison of the results of such an analysis with appropriate standards or criteria.'*

A performance assessment becomes a *safety assessment* when

> *'...the system under consideration is the waste disposal system and the performance measure is the radiological impact or some other global measure of impact on the safety of humans and the environment.'*

The Scientific Committee 87-3 established by the National Council on Radiation Protection (NRCP) used these definitions to establish suitable guidelines and concepts for conducting a post-closure safety assessment for radioactive waste disposal facilities. In their findings

(Kennedy, 1997), the committee portrayed a post-closure safety assessment as a multi-disciplinary, iterative process focussed on regulatory compliance rather than an analysis of a disposal system for the purpose of predicting its actual behaviour. With this in mind, they defined a post-closure safety assessment as: *'the iterative process involving site-specific, prospective evaluations of the post-closure phase of the system'* with three primary objectives:

(a) determine whether *reasonable assurance* of compliance with quantitative performance objectives can be demonstrated,

(b) identify data, design and other needs to reach *defensible decisions about regulatory compliance,* and

(c) identify waste acceptance criteria—i.e. a list of the waste types for which the repository is intended—related to the quantities of wastes that need to be disposed.

1.3 THE SITUATION IN SOUTH AFRICA

Radioactive waste in South Africa is generated mainly by two agencies: the Atomic Energy Corporation of South Africa Ltd. (AEC) at Pelindaba, and the Electricity Supply Commission ESKOM. The AEC is involved in many of the activities related to the nuclear fuel cycle, while ESKOM is responsible for the operation of the Koeberg Nuclear Power Plant near Cape Town in the Western Cape Province, as indicated in Figure 1-1. Users of radioisotopes in industry, medicine and research, the mines and mineral processing plants also produce various types of radioactive wastes.

Two sites are currently used for the disposal of radioactive waste in South Africa. The first site, Thabana (previously known as Radiation Hill), is situated at Pelindaba near Pretoria in the North-West Province (see Figure 1-1). This site has been in operation since 1969 and consists of a variety of earth trenches used for the disposal of uranium-contaminated waste and some plutonium. A stainless steel engineered borehole is also used for the disposal of ^{60}Co sources. The second site is the National Radioactive Waste Disposal Facility at Vaalputs near Springbok in the Northern Cape Province (see Figure 1-1). This site came into operation in 1986 and is currently being used for the disposal of low- and intermediate level waste from Koeberg in near-surface trenches.

The Council for Nuclear Safety (CNS), a statuary body established by the Nuclear Safety Act of 1993 and earlier acts control the disposal of nuclear waste in South Africa exclusively. This body has already granted a license for the disposal of radioactive waste at Vaalputs in 1986, after the site went through a detailed screening, selection and characterization

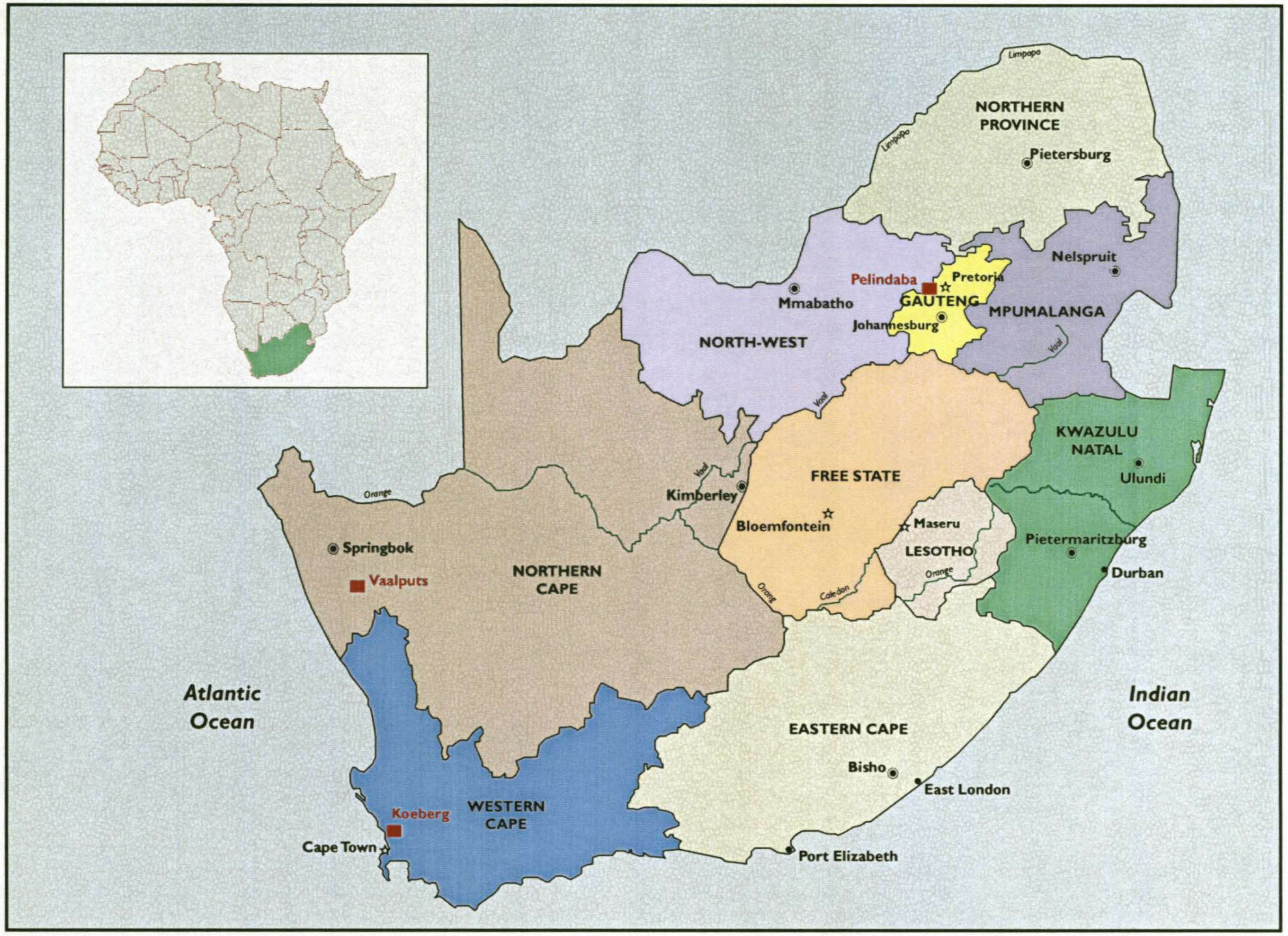


Figure 1-1 Locality map of the two radioactive waste disposal facilities at Vaalputs and Pelindaba, as well as the Nuclear Power Plant at Koeberg.

process (Corner and Scott, 1980; Moore *et al.*, 1987). However, very little is known about the selection criteria used for issuing a license to permanently store active solid waste at Thabana by the then Safety Committee of the Atomic Energy Board (Niebuhr *et al.*, 1968).

One difficulty with the current Vaalputs license is that it does not address the post-closure safety of the facility adequately. The methods used to demonstrate the safety characteristics of the site also did not conform to the current internationally recognized standards for performing the post-closure safety assessment of a radioactive waste disposal facility. This view was confirmed by an Expert Mission of the International Atomic Energy Agency (IAEA) to Vaalputs (IAEA, 1998*a*). They concluded that, although there do exist documents that stipulate procedures to ensure the operational safety of the site, and to protect the environment in the short term, very little has been done to ensure its post-closure safety. Particular points raised by them include the following:

(a) The isolation strategy—the safety principles on which waste disposal at Vaalputs is based—is not documented adequately.

(b) The waste acceptance criteria do not specify the nuclides that can be accepted for disposal.

(c) The nature of the containment barriers and their function in achieving waste isolation is not documented adequately.

(d) No provision was made for a policy to control the disposal site and nearby areas in the post-closure period.

The strategies followed to obtain licenses for Vaalputs and Thabana were probably appropriate and sufficient at the times they were granted, but not today. One particular disadvantage of the present management systems is that they fail to address the consequences of today's actions on future generations. This applies especially to spent fuel produced by the reactors at Pelindaba and Koeberg and the disposal practices at Thabana. All the spent fuel at Koeberg is currently stored in reactor pools, while some are stored in reactor pools and the rest in dry storage at Pelindaba. Since a permanent solution for this problem has not receive much attention in South Africa, the country can be accused of leaving the problem to future generations to solve (Hambleton-Jones *et al.*, 1998). This violates one of the fundamental principles of radioactive waste management discussed above.

As far as Thabana is concerned, there is confusion whether it should be considered as a storage or a disposal facility. The license granted to the AEC in 1969 was for the permanent storage of radioactive waste *without the intention of retrieval* (Niebuhr *et al.*, 1968). The AEC has not obtained a license from the CNS for the disposal of radioactive waste at the site, and no assessment of the site's suitability as a disposal site has ever been undertaken. It is therefore not certain whether the site is suitable for the disposal of radioactive waste. Moreover, it is very difficult to determine what long-term consequences the past and current practices have on humans and the environment, since there is very little information on the practices used previously to dispose radioactive waste at Thabana. Considerable uncertainties therefore exist about the total inventory and its future, that is, should the site be upgraded to an approved disposal facility or rehabilitated and closed. In the meantime, Thabana has to be considered as a disposal facility, for it will be necessary to quantify the current inventory of the waste, its movement through the disposal system and its potential impact on future generations, and to undertake the necessary remedial actions if an when necessary.

1.4 PURPOSE OF THE STUDY

The previously described situations call for the definition and implementation of a post-closure safety assessment strategy for radioactive waste disposal facilities in South Africa, based on the well-established and accepted international methodology. However, there does not exist, at least at this moment, a document that describes the methodology, its implications and the steps necessary to implement it in South Africa. In this thesis an attempt is made to develop such a structured methodology for radioactive disposal sites in South Africa and other parts of Africa, in such a way that it can also be understood by interested members of the public.

The aim of the thesis is not to present new priciples of radioactive waste management, except for a new management strategy to dispose spent nuclear sources in boreholes, but rather to use the existing information to develop such a structured methodology. This could only be achieved by using the information from various organizations, regulatory authorities and other interested parties. Many of the principles discussed in this thesis are therefore not new. However, it is believed that the structured approach advanced here—the integration of radioactive waste management, post-closure safety assessment of radioactive waste disposal sytems and the associated nuclear liabilities—is original.

A post-closure safety assessment of radioactive waste disposal systems is an extensive exercise that requires input from various scientific, engineering, social and economic disciplines, which cannot be adequately covered by an individual. The thesis therefore concentrates exclusively on the broad principles of such a post-closure assessment and not its detailed technical implementation. Readers interested in the more technical aspects are referred to (Kozak, *et al.*, 1999), where the practical implementation of a preliminary version of the methodology is described in detail.

1.5 SCOPE OF THE STUDY

Radioactive waste management is the concluding part of all activities related to the use of radioactive materials in the nuclear industry, agriculture, research and medicine. Traditionally, the management and other activities related to the pre-closure phase of a waste disposal site were considered separately from any post-closure assessment of the impact that these activities may have on humans and their environment in the future. However, the discussion in Section 1.2 shows that this is not desirable, at least as far as radioactive waste is concerned, and that the two periods should be considered simultaneously within an integrated framework for the management of radioactive waste.

It is obvious that one must have a detailed knowledge of all the various factors that may influence the disposal, before such a framework can be designed. In the case of radioactive waste these factors include the nature and properties of radioactive materials, the origin and types of radioactive waste and the method that will be used to dispose the waste. The effects that waste have on humans and the natural environment will vary with the process that generates it and how it is treated before disposal. It is therefore advantageous to have a scheme which can be used to classify the waste, albeit qualitatively. The scheme commonly used in the nuclear industry is discussed in Chapter 2, together with the nature and properties of radioactive materials and the origin and types of radioactive waste.

In the early days, radioactive waste were often stored for an interim period to allow for some decay, and then disposed of by dispersion and dilution in natural reservoirs (IAEA, 1993*b*). However, the production of larger quantities (and more dangerous) of radioactive waste made it necessary to develop new management strategies that take advantage of the properties and nature of radioactive waste, discussed in Chapter 2. Some of the more important strategies that have been advanced over the years for the management of radioactive waste and their relation to the integrated management of radioactive waste are discussed in Chapter 3. Although there are a number of strategies that can be followed the discussion shows that disposal is the only suitable method for the long-term management of radioactive waste. However, to use this approach one must ensure that the disposed waste wil not affect humans and their environment adversely. A number of the proposed practices, their associated management practices and so-called nuclear liabilities are also discussed.

A disposal strategy can only be implemented if it can be demonstrated that it satisfies the principles of radioactive waste management. This objective is best achieved by performing an integrated post-closure safety assessment of the disposal system. However, this can only be achieved if one has a good knowledge of the components of the system and how they are incorporated in a safety assessment, as described in Chapter 4. These components, which can be conveniently divided into internal and external components, provide the necessary information to characterise the movement of radionuclides through what is called the near field, geosphere and biosphere of the system.

A few disposal concepts for radioactive waste that are of particular interest to South Africa are discussed in Chapter 5. This includes a concept for the disposal of low- and intermediate-level waste at Vaalputs and one for the disposal of various categories of waste at Thabana. A permanent solution for the high-level waste generated by Koeberg and the AEC has not received much attention in South Africa. In fact, not even a conceptual plan of the disposal concept that will be used in South Africa has yet been formulated. The conceptual geological disposal concept prepared by Atomic Energy of Canada Ltd. for the disposal of spent fuel in Canada is consequently used as an example of what can be expected, if one wants to implement such a concept. The main reason for choosing this site is that its geological characteristics are very similar to that at Vaalputs, where such a facility may likely be implemented. The discussion concludes with the description of the borehole concept, recently proposed for the disposal of spent nuclear sources in South Africa and other African countries. These examples show that design of a disposal concept will very much depend on the properties and characteristics of the waste that is to be disposed.

An attempt is made in Chapter 6 to clarify the often misuse of the term model in safety assessments, before discussing a broader perspective of the basic principles of a post-closure safety assessment. The reason for this is that models play a very important role in the safety assessment methodology. The proposed methodology, which is based on various guidelines already accepted or considered by the international community, exhibits some unique characteristics, when compared with similar methodologies for the disposal of other hazardous waste.

The assessment of a radioactive waste disposal site was historically often seen as an attempt to predict the behaviour of the site far into the future with deterministic, predictive models. These models are based on the physical principles that underlie a specific phenomenon and should therefore be able to predict the future behaviour of the phenomenon. Unfortunately, this requires information on parameters whose behaviour cannot be determined with certainty far into the future. It is therefore impossible to apply the models in the historical sense of a safety assessment. The purpose of the heuristic and phenomenological models discussed in Chapter 7 for the evaluation and migration of radionuclides through the near field, geosphere and biosphere, is thus not to predict the future, but rather as an aid to assess whether a site will comply with regulatory or community imposed safety constraints. However, it is important to note that a safety assessment is not an exact procedure. It will therefore be impossible to demonstrate complete compliance with the imposed constrains, even with the technological aids available today. Nevertheless, it is believed that by using the assessment methodology proposed here, it will be possible to demonstrate the safety of a site with reasonable assurance.

A major contributor to the inexact nature of a safety assessment is the uncertainties that are inherently part of the analysis, as discussed in Chapter 8. These uncertainties can be conveniently divided into uncertainties related to the unknown future state of the disposal system, data and parameter uncertainties, and model uncertainties. The chapter is concluded with a discussion on how to treat these uncertainties in a post-closure safety assessment.

The safety assessment methodology developed herein can be described as a decision tool to determine the conditions under which compliance with safety objectives can be reasonably assured and consequently very much resembles system analysis. However, no attempt has been made in the past to formally include system analysis theory into the post-closure safety assessment of radioactive waste disposal systems. It is also not the purpose of this thesis to do so either. What is discussed, in Chapter 9, is a decision analysis framework that will aid in making more reliable decisions in such an assessment and also reduce nuclear liabilities.

CHAPTER 2

HISTORICAL OVERVIEW AND PROPERTIES OF RADIOACTIVE WASTE

2.1 RADIOACTIVITY

The discovery of radioactivity dates back to 1896 when the French physicist Henri Becquerel tried to relate the emanations from the fluorescent mineral pitchblende to the X-rays, discovered in 1895 by the German physicist W.C. Röntgen. Between 1896 and 1898, Becquerel and his students Pierre and Marie Curie discovered various naturally occurring radioactive elements such as uranium, radium and thorium, as indicated in Table 2-1. Today, the application of radioactive materials range from medical, industrial and research techniques, to the generation of nuclear energy and the manufacturing of nuclear weapons.

Despite the often tragic consequences of errors made in handling and applying radioactive materials during the following decades, the general fascination with radioactivity did not abate in any way. Even as late as the nineteen-thirties, charlatans were promoting the use of radioactive toothpaste, radium hair-tonic and salve and cloth impregnated with radium. Mineral water with a high radon content was also frequently prescribed as being good for the health (IAEA, 1991). However, advances in both the theoretical principles and practical applications of radioactivity have since led to the introduction of radiation protection regulations that assure high levels of safety if applied correctly.

The discussion on the properties of radioactive waste will start with a review of the nature and effects of radioactive material in Section 2.2, as discussed by Chapman and McKinley (1988). This discussion will provide greater clarity on the principles and methodologies applied in the management of waste generated from nuclear related activities.

Radioactive waste is generated from a variety of sources and depending on the origin and type of radioactive waste, will exhibit different properties. Section 2.3 is consequently devoted to a discussion on the origin and types of radioactive waste generated from various sources. The different properties of radioactive waste suggest that not all waste should be treated the same, but instead to define categories of waste with similar properties that can be managed accordingly. This led to different classification schemes for radioactive waste that will be discussed in Section 2.4.

Table 2-1 A brief chronicle of the discovery and applications of radioactivity between 1896 and 1992 (Issler, 1990; USDOE, 1998; IAEA, 1991).

1896	Becquerel discovers that invisible emanations from the native oxide uranium, pitchblende, affected photographic plates and called them radioactive rays.
1898	Marie and Pierre Curie isolate radium from pitchblende, which was much more effective in darkening photographic plates than pitchblende itself, and later the radioactive elements actinium, polonium and thorium.
1899	Rutherford shows that there are three types of radioactive rays, which he called α-, β- and γ-rays, names that persist today.
1901	Marie Curie provided a physician at a Paris hospital with a radium source to be used for medical treatment. The source was to be applied to a malignant surface tumour. Two years later the first successful treatment was reported.
1904	The first attempt to treat a tumour inside the body was made by inserting a glass capsule containing radium into the patient.
1911	Rutherford's model of the atom distinguishes between the nucleus and the electron cloud of an atom. Georg von Hevesy conceives the idea of radioactive tracers. The idea is later applied to medical diagnosis and the transport of radioactivity in groundwater.
1927	Herman Blumgart, a Boston physician, first uses radioactive tracers to diagnose heart disease.
1938	Two German scientists, Otto Hahn and Fritz Strassmann discover that bombarding uranium atoms with neutrons produces barium atoms. Lize Meitner explains this in terms of nuclear fission.
1939	Enrico Fermi discovers the possibility of a chain reaction. Hans Bethe identifies nuclear fusion as the energy of the sun.
1942	Fermi demonstrates the first self-sustaining nuclear chain reaction in the Chicago reactor.
1944	The first nuclear reactor begins operation in the USA and the first disposal in Oak Ridge, Tennessee.
1945	The USA explodes the first atomic device at a site near Alamogorda in New Mexico on the 24th of July, and drops the first atomic bomb on the 6th of August on Hiroshima and the second on the 16th of August on Nagasaki.
1951	The first usable electricity from nuclear fission is produced.
1955	Arco, Idaho becomes the first U.S. town to be powered by nuclear energy.
1957	Radiation is released when the graphite core of the Windscale Nuclear Reactor in England catches fire. The International Atomic Energy Agency (IAEA) is formed to promote the peaceful uses of nuclear energy.
1963	The United States and Soviet Union sign the Limited Test Ban Treaty, which prohibits underwater, atmospheric and outer space nuclear tests.
1966	The large number of utility orders for nuclear power reactors makes nuclear power a commercial reality in the USA.
1968	The Nuclear Non-proliferation Treaty—calling for halting the spread of nuclear weapons capabilities—is signed.
1972	Computer axial tomography, commonly known as CAT scanning, is introduced.
1977	United States president, Jimmy Carter, bans the recycling of used nuclear fuel from commercial reactors.
1979	The Three Mile Island nuclear power plant near Harrisburg, Pennsylvania suffers a partial core meltdown. Minimal radioactive material is released.
1986	The Chernobyl Nuclear Reactor melts down in the Soviet Union. Massive quantities of radioactive material are released in the atmosphere over Europe.
1987	Yucca Mountain, Nevada, is designated as the prime site for the first geological repository in the United States.
1992	One hundred and ten commercial nuclear reactors are operating in the United States.

2.2 NATURE AND EFFECTS OF RADIOACTIVE MATERIALS

Matter is composed of atoms. The nucleus of such an atom is composed of positively charged protons and neutral neutrons. However, an atom is electrically neutral, since the positive charge of the protons is balanced by the negative charges of the identical number of electrons, which surround the nucleus. These electrons are responsible for the chemical properties of the atom.

All atoms with the *same number of protons* are chemically identical and called an *element.* The number of protons in a nucleus (*atomic number*) is consequently often represented by the chemical symbol for the element, e.g. U for uranium, or H for hydrogen. However, atoms of the same element sometimes contain different numbers of neutrons. These atoms are known as *isotopes* and denoted by the chemical symbol of the element superscripted to the left with the sum of its protons and neutrons (*mass number*), e.g. ^{238}U. Situations often arise though where it is necessary to describe a specific atom of an element, or *nuclide*, more precisely. In such cases the number of protons is added as a subscript to the left of the symbol and the number of neutrons as a subscript to the right of the symbol, e.g., ${}^{238}_{92}U_{146}$.

The majority of the known nuclides are inherently unstable and transform spontaneously into other nuclides by emitting an α-, a β, or a γ-ray, or split into two or more nuclides—a process called *fission*. These processes are called *radioactive decay*, hence the name *radionuclides* for nuclides that exhibit one or more of these processes spontaneously. The α- and β-rays emitted by radionuclides are particles, the nucleus of the ^4He nuclide and positively and negatively charged electrons ($e^{\pm}$) respectively, while γ-rays are, like light, electromagnetic radiation with a very short wavelength.

The radionuclide that decays first is commonly referred to as the *mother* or *parent* nuclide, and the product nuclide as the *daughter*. Many daughter nuclides are often also unstable and decay again, thereby creating what is known as a *decay chain.*

Numerous experiments have shown that the rate at which a sample of radionuclides disintegrates is directly proportional to the number of atoms, $N(t)$, in the sample at the time t, that is

$$\mathrm{D}N(t) = -\lambda N(t) \tag{2.1}$$

where λ is known as the *decay constant*, that varies from isotope to isotope. This property is conventionally characterized by another constant, known as the *half-life* of the radionuclide and denoted by the symbol $t_{1/2}$. The half-life is, as the name implies, the time required for a large number of atoms of a particular radionuclide to decrease by 50%, in other words from their original number, N_0 at $t = 0$, to $N_0/2$ at $t = t_{1/2}$. The half-life should therefore be related to the decay constant, as it is indeed. To show this, one merely has to integrate Equation (2.1) over time to obtain

$$N(t) = N_0 \exp(-\lambda t)$$

and then replace t by 0 and $t_{1/2}$ and N with N_0 and $N_0/2$ respectively, which yields

$$\lambda = \frac{\ln 2}{t_{1/2}}$$

The half-lives of the radionuclides known today vary from a fraction of a second to millions of years. For example the half-life of ^{219}Pa is $5.3 \cdot 10^{-8}$ s, while that of ^{128}Te is $8.0 \cdot 10^{24}$ years.

The *activity or* rate at which a radioactive material decays, is measured in terms of the number of nuclei which decay or disintegrate each second. The SI-unit for this quantity is the *becquerel* (Bq), defined as a decay rate of 1 nucleus per second. The unit, unfortunately, says nothing about the type of radiation, its energy, or its ability to interact with matter. This led to the introduction of the term *absorbed dose*, defined as the energy absorbed by a body exposed to radioactive radiation, with SI-unit the *Gray* (Gy), defined as the energy absorbed by 1 kg of the material.

Experience over the years has shown that the effect radioactive radiation has on an individual differs from organ to organ and also with the individual's age. This property is described by the term *equivalent dose* , obtained by multiplying the dose for a particular organ with a suitable factor, that depends on the biological effect the type of incident radiation may have on the organ. However, one is usually not so much interested in damage to a particular organ in the management of radioactive waste, but rather to the individual as such. The *effective dose,* defined as the sum of the dose equivalents for each part of the human body, weighted by factors that take the susceptibilities of the different organs to radiation-induced damage into account, is such a quantity. The *sievert* (Sv) is the SI-unit for both the equivalent dose and the effective dose.

There are essentially three ways in which an individual may be exposed to radioactive radiation. The first two are the inhalation of radionuclides in the form of gasses or airborne particles, and the ingestion of radioactive foods or drinks (especially water), as is the case with other hazardous substances. The third one, damage caused by exposing the body directly to the radiation, however, is unique to radionuclides.

The main reason why radioactive radiation is biologically harmful, is its ability to split and kill individual cells, or prevents them from reproducing normally, thereby causing the development of malignant tumours and other medical abnormalities. This ability depends in the first place on the strength with which the different rays interact with matter. As illustrated in Figure 2-1, α-particles interact very strongly with matter and are easily stopped by a sheet of paper, while β-particles (which interact less strongly with matter), can be stopped by a thin layer of metal. The γ-particles, however, are very penetrating and can only be stopped by thick shielding materials, such as lead or concrete. Elements that emit β- and γ-rays are therefore the main sources of concern in the management of radioactive waste, particularly in those cases where the waste poses an exposure risk to the public. This does not mean though that α-emitting nuclides are not important in radioactive waste management. On the contrary, these nuclides often cause the greatest damage to internal organs when inhaled or ingested by an individual.

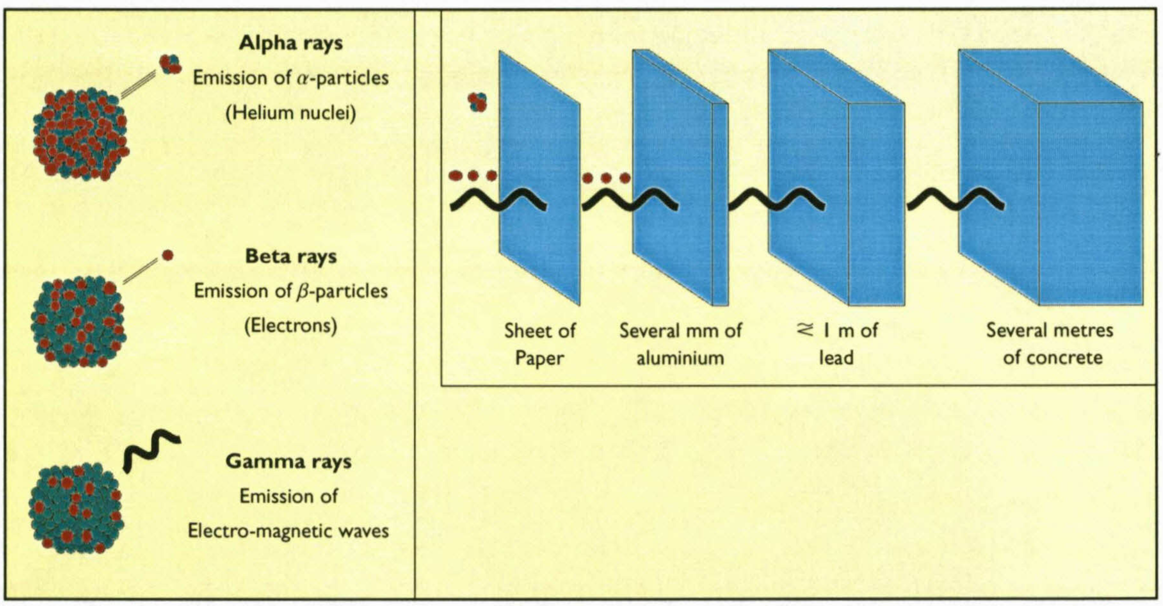


Figure 2-1 The three types of radiation emitted by radionuclides and the shielding required to stop them (after Issler, 1990).

2.3 ORIGIN AND TYPES OF RADIOACTIVE WASTE

2.3.1 General

As mentioned in Chapter 1, radioactive waste is generated mainly by processes related to the nuclear fuel cycle, the operation of nuclear reactors and the decommissioning of nuclear facilities, with smaller quantities produced by industry, research institutions and the medical profession. The different origins of radioactive waste can cause a variation in its physical state (solid, liquid or gaseous), activity and type. Activity levels range from extremely high for spent fuel, to very low for radioisotope applications in laboratories, hospitals and universities. Equally broad is the spectrum of the half-lives of radionuclides contained in the waste. Since the waste generated by one process may pose a different risk to humans and the environment than that generated by another, it may be worthwhile to briefly review these processes before continuing this discussion on radioactive waste.

2.3.2 The Nuclear Fuel Cycle

The term *nuclear fuel cycle* is used to denote all processes connected with nuclear power generation, including the mining and milling of fissile materials, enrichment, production, utilisation and storage of nuclear fuel, optional reprocessing of spent fuel, and the processing and disposal of the resulting wastes (IAEA, 1993*a*). There are essentially three processes in the nuclear fuel cycle where waste is generated (IAEA, 1992): the mining, milling and refinement of uranium ore, the production (enrichment, or conversion) of reactor fuel elements, and the production of spent fuel in nuclear reactors.

At the front end of the fuel cycle, uranium is mined, milled, chemically processed and usually enriched in the isotope ^{235}U to a concentration of between 3 and 3.5%, to produce nuclear fuel. In the enrichment process the solid uranium oxide (U_3O_8) is first converted chemically to the gaseous uranium hexaflouride (UF_6), by a process called fluorination, and the enriched fraction of UF_6 converted to uranium dioxide (UO_2). This is then used to produce the fuel elements for nuclear reactors. Small amounts of liquid and solid wastes, contaminated with uranium, are produced during this process. The calcium fluoride, generated by the conversion of UF_6 to UO_2, is especially important in the regard.

The radioactive waste generated by the mining and milling of uranium and thorium ores, are usually deposited on tailings dams at the mines. These materials contain relatively low concentrations of long-lived radionuclides, of which uranium's daughter elements, thorium, radium and radon are the most important.

The waste generated by a nuclear reactor can be broadly divided into two classes, the solid and solidified waste, containing corrosion, activation and fission products, and the spent fuel. The first class arises from the cleaning of the reactors cooling systems, fuel storage ponds and the decontamination of equipment. This waste consists mainly of paper, filters, ion exchangers, contaminated clothing and equipment, floor sweepings and concrete.

The spent fuel is sometimes reprocessed to regain various isotopes, particularly ^{239}Pu and ^{241}Pu. These isotopes are formed when the ^{238}U in the fuel cells capture a thermal neutron.

The main reason for this is that these isotopes are also fissionable and can therefore also be used in the production of new fuel elements instead of the ^{235}U. Indeed, they are often more economical to use than ^{238}U, since their fission efficiency is higher , and they differ chemically completely from uranium, so that they can be separated chemically from the uranium. Unfortunately, this process generates waste containing considerable quantities of uranium fission products and highly energetic actinides.

2.3.3 Decommissioning of Nuclear Facilities

All nuclear facilities, especially nuclear reactors have a finite operational life and need to be decommissioned at some point in time (IAEA, 1992). Since the facilities and their sites are often highly contaminated with radioactive materials, they must be dismantled and decontaminated, before the site can be used for other uses, thereby generating what is known as decommissioned waste. In the case of a nuclear reactor the volume of this waste, which consists mainly of the construction materials of the dismantled facility, its hardware and contaminated soil, is often much larger than the volume of waste produced by the nuclear fuel cycle. The decommissioning of these plants is consequently often delayed, because of the lack of suitable disposal facilities.

2.3.4 Other Forms of Nuclear Waste

The volume of radioactive waste generated by the industry, research laboratories and the medical profession is small compared to the nuclear fuel cycle, but still may pose a considerable health risk to humans if not disposed properly. Of considerable importance in this regard are spent sealed radiation sources, or sealed sources for short, defined by the International Standards Organisation (ISO) as IAEA (1991):

> *'A radioactive source sealed in a capsule or having a bonded cover, the capsule or cover being strong enough to prevent contact with and dispersion of radioactive material under normal conditions of use and wear for which it was designed.'*

The applications of sealed sources found in the industry include: belt, density, level and thickness gauges, industrial radiography, moisture detectors, the sterilisation and preservation of food, and roentgen fluorescent analysis. Research applications include calibration sources, electron capture detectors, densitometers, tritium targets and eliminators for static electricity. Sealed sources are also used widely in the medical field for bone, brachytherapy, teletherapy and clinical radiotherapy (IAEA, 1991). Since the physical dimensions of these sources are small, although some of them may contain highly active isotopes, they are very susceptible to theft and misuse, with a corresponding potential radiation hazard to humans.

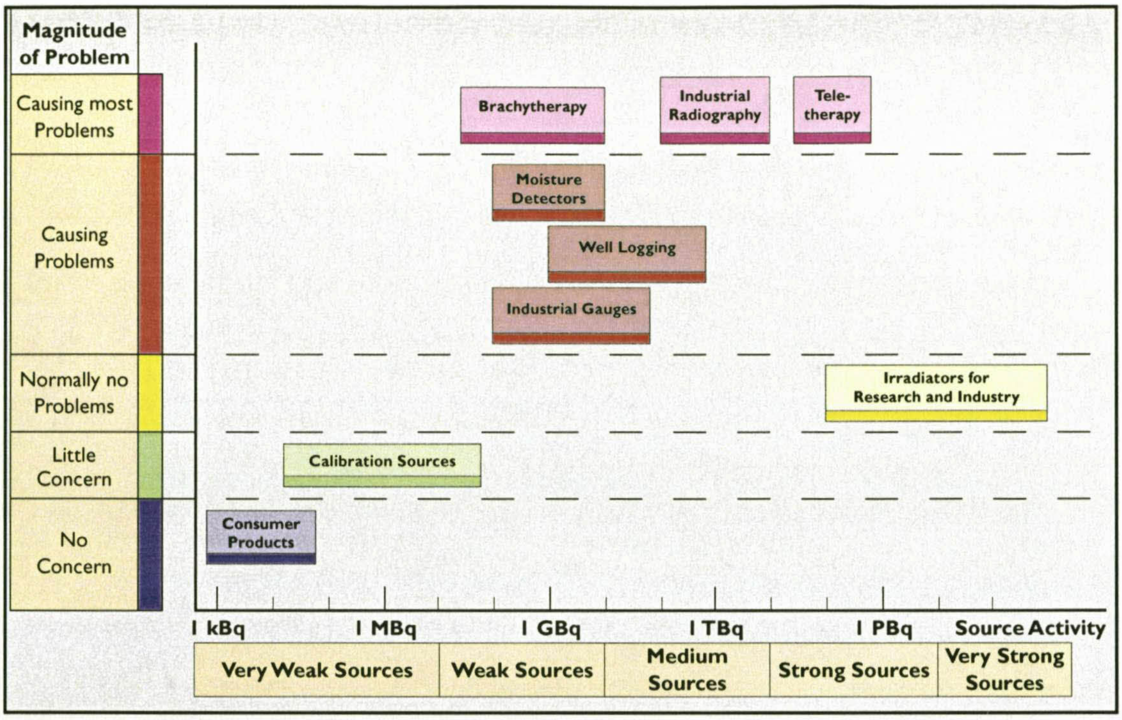


Figure 2-2 Activity range for some important applications of sealed sources and the magnitude of problems associated with spent sources (after IAEA, 1991).

Figure 2-2 gives an overview of sealed sources, their main areas of application and the relative magnitudes of the problems associated with the different types of sources.

2.4 CLASSIFICATION OF RADIOACTIVE WASTE

2.4.1 General

All wastes that contain radioactive materials should be regarded as radioactive from the physical point of view. However, the activities of the radioactive materials in some of the wastes are so low that they do not present radiological hazards to the environment. The IAEA (1993*a*) therefore define radioactive waste for *legal and regulatory purposes* as:

> *'waste that contains radioactive materials with activities higher than the clearance levels established by the regulatory body, and for which no further use is foreseen.'*

The preceding discussion indicates that it will be wrong to treat the various forms of radioactive wastes on the same footing. The physical, chemical, biological and radiological properties of radioactive waste have consequently been used to devise criteria, summarised in Table 2-2, for the qualitative and quantitative classification of the waste. Since these classification schemes are very useful in the management of radioactive waste, they are discussed in more detail below.

Table 2-2 Important properties of radioactive waste used as criteria to classify them (IAEA, 1994*a*).

General	Specific
Origin	
Criticality	
Radiological properties	Half life
	Heat generation
	Intensity of penetrating radiation
	Activity of the radionuclides
	Surface contamination
	Dose factors of the relevant radionuclides
Physical properties	Physical state (solid, liquid or gaseous)
	Size and weight
	Compactability hazards
	Dispensability
	Volatility
	Solubility, miscibility
Chemical properties	Potential chemical hazard
	Corrosion resistance/corrosiveness
	Organic content
	Combustibility
	Reactivity
	Gas generation
	Sorption of radionuclides
Biological properties	Potential biological hazards

2.4.2 Qualitative Classification

There are a number of ways in which radioactive waste can be classified qualitatively. One approach is to group the waste in terms of its origin, physical state (solid, liquid or gas), or activity. A scheme based on the last property is particularly useful, as it allows one to classify the waste semi-qualitatively into *low-level* (LLW), *intermediate-level* (ILW) and *high-level* (HLW) waste. This *activity classification system* is consequently the one most widely used today.

In the activity classification LLW is waste whose activity is so low that it does not require shielding during its handling and transportation, while ILW is waste that requires shielding, but will not generate a significant amount of heat. High-level waste is, as its name implies, waste whose activity is so high that it can generate a significant amount of heat (≥ 2 kW m^{-3}). The best known forms of high-level waste include: the highly radioactive liquids, containing fission products and some of the actinides, generated by the chemical reprocessing of spent fuel, and spent fuel itself, if declared a waste.

The IAEA (1994*a*) tends to differentiate further between short-lived, long-lived and alpha bearing low-level and intermediate-level wastes. As used here, the terms short-lived and

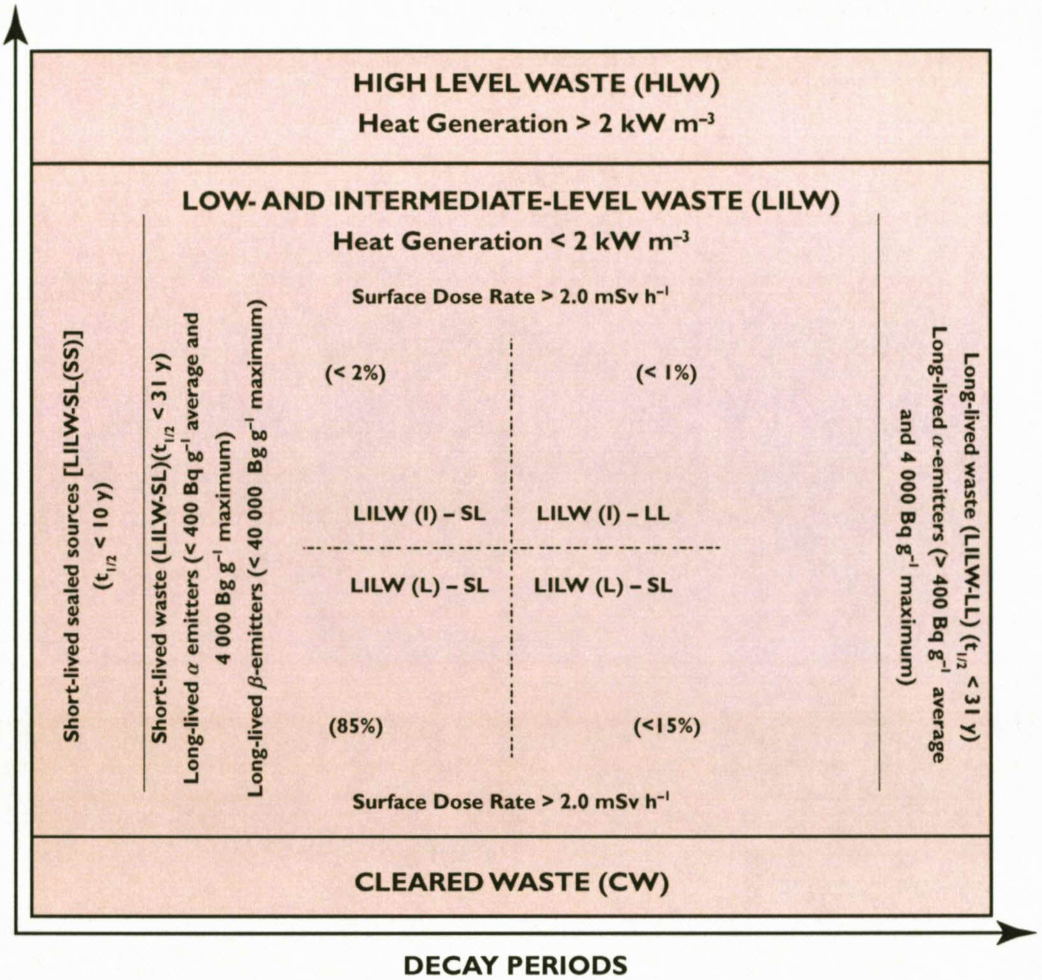


Figure 2-3 The qualitative radioactive waste classification scheme proposed for the AEC. The figures in brackets represent the estimated percentage of AEC waste (AEC, 1997*a*).

long-lived waste refer respectively to radioactive waste that will, or will not, decay to acceptable activity levels, within the time period during which administrative controls are expected to last. Alpha bearing waste, on the other hand, is simply radioactive waste that contains one or more α-emitting radionuclides, usually actinides, in quantities above the acceptable limits established by the national regulatory body.

2.4.3 Quantitative Classification

In many cases, the classification of radioactive waste is related to the safety objectives set by a regulatory body for their management. These safety objectives are often formulated in numerical terms, such as dose rate or activity level, which requires a quantitative classification system (IAEA, 1994*a*).

The quantitative radioactive waste classification scheme at the AEC, illustrated schematically in Figure 2-3, recognises three main categories of waste: *cleared waste* (CW), *low-* and *intermediate-level waste* (LILW) and *high-level waste* (HLW) (AEC, 1997*a*). Cleared waste contains so little radioactive material that it cannot be considered as *radioactive*, and might be cleared from nuclear regulatory control. That is to say, although still radioactive from a physical point of view, this waste may be safely disposed of, applying conventional techniques and systems, without specifically considering its radioactive properties (IAEA, 1994*a*).

In the AEC classification LILW can contain a wide range of radionuclides and radionuclide concentrations that may require special packaging, conditioning and disposal options. Low- and intermediate-level waste are consequently further subdivided into the sub-classes indicated in Figure 2-3, and summarised below.

(a) Short-lived waste, sealed sources [LILW-SL(SS)].

(b) Short-lived waste; low dose rate [LILW(L)-SL].

(c) Short-lived waste, intermediate dose rate [LILW(I)-SL].

(d) Long-lived waste; low dose rate [LILW(L)-LL].

(e) Long-lived waste, intermediate dose rate [LILW(I)-LL].

However, only the qualitative classification scheme radioactive waste will be used in the discussions that follow.

CHAPTER 3

MANAGEMENT OF RADIOACTIVE WASTE

3.1 INTRODUCTION

Radioactive waste were often stored in the early days for an interim period to allow for some decay, and then disposed of by dispersion and dilution in natural reservoirs (IAEA, 1993*b*). However, the production of larger waste quantities, such as the by-products of the nuclear fuel cycle, made it necessary to develop new management strategies. Which strategy to follow will, however, depend largely on the properties of the waste and the internationally accepted fundamental principles for the management of radioactive waste, as outlined in Chapter 1. This chapter will consequently be devoted to an overview of an integrated radioactive waste management approach in Section 3.2, followed by a discussion on some of the more important strategies that have been advanced over the years for the management of radioactive wastes, in Section 3.3. This is followed by a discussion of the main objectives for radiation protection in Section 3.4, and a brief overview of the present status of disposal practices for low-level, intermediate-level and high-level wastes, as well as spent sealed sources, in Section 3.5.

All facilities associated with nuclear activities eventually have to be decontaminated, decommissioned and the radioactive waste produced by the activities managed in a way that obey the fundamental principles of radioactive waste management, as discussed in Chapter 1. The cost and responsibility for these operations create what is called *nuclear liabilities* (OECD/NEA, 1996), which is discussed in Section 3.6.

3.2 AN INTEGRATED APPROACH TO RADIOACTIVE WASTE MANAGEMENT

3.2.1 General

Radioactive waste management is the concluding part of all activities related to the production of nuclear fuel, the generation of nuclear power, research and development, and the many applications of radioisotopes, as discussed in Section 2.3 (IAEA, 1992). According to the IAEA glossary (IAEA, 1993*a*), the term integrated approach refers to

> *'a logical and preferably optimized strategy of a radioactive waste management programme as a whole, from waste generation to disposal, so that the interac-*

tions between the various stages of waste management are taken into account and that decisions made at one stage do not foreclose certain alternatives at a subsequent stage.'

This definition is a confirmation of the eighth principle in Chapter 1—*radioactive waste generation and management interdependencies*—stating that interdependencies among all steps in radioactive waste generation and management shall be taken into account. In many cases the waste generating process is chosen before planning how the waste will be managed. It has been recognized, however, that a careful consideration of the waste generation from the beginning can help considerably in solving problems that may develop afterwards (IAEA, 1992).

As mentioned in Section 1.2, the pre-closure and post-closure assessments have to be treated differently, but the periods themselves are interrelated. For example, in the pre-closure period, safety of the operators has to be assured and their radiation exposure kept at a minimum. After closure of the repository, the main concern is to minimise the exposure of the population to radiation. However, the treatment process of waste can determine the suitability of waste for storage and the selection of a disposal option; conversely, the characteristics of a chosen site or repository can determine requirements on waste specifications. It therefore makes sense to consider the various stages of an integrated radioactive waste management process as part of a structured methodology for a post-closure safety assessment. This includes waste generation, waste pre-treatment, waste treatment and conditioning, interim storage, and disposal. These steps, taken prior to disposal, are very important, since they determine the quality and quantity of waste to be disposed and the method that can be used for the disposal. They and the sources and types of radioactive waste generated in the nuclear industry are discussed in more detail below. Two other, equally important, components in an integrated approach to radioactive waste management, that which will not be discussed in this thesis, are the need for a national legal framework (IAEA, 1995*c*) and the role of the regulatory body (Environmental Agency *et al.*, 1997).

3.2.2 Waste Pre-treatment

Waste pre-treatment comprises all technical and administrative operations to collect, separate, characterize, classify, package and document radioactive waste, according to the national waste classification procedures, taking the radiological content and characteristics of the waste into account. The purpose of pre-treatment is to try to change the characteristics of the waste in such a way that it can be handled more safely and economically in the future. It may also include volume reduction and decontamination of contaminated materials. Note, however, that this may lead to the production of secondary wastes, such as contaminated filters, spent resins and sludge, which may have to be separated and pre-treated again.

3.2.3 Waste Treatment and Conditioning

Depending on the level of treatment done during the previous step, waste treatment encompasses the operations intended to benefit safety and/or economy by changing the characteristics of the waste. This is done through volume reduction, removal of radionuclides from the waste and change in composition in order to achieve the preferred approach of concentration and containment, rather than dilution and dispersion in the environment.

After treatment, the waste is still not in a condition that it can be handled and transported or disposed safely. To achieve this the waste is almost always immobilised first—a process known as *conditioning*. The simplest way to achieve this is to mix the waste with a solidifiable material, and place the mixture into a container that also serves as a mould during the infilling and solidification of the waste mixture, or *waste form*. Solidifiable materials often used for this purpose include synroc, glass or borosilicate for HLW, and cement, bitumen or resins for LLW and ILW. The key parameters that need to be considered in selecting the solidifiable material include: the distribution stability of the radionuclides within the waste form and the rate at which the material will decay physically and chemically. The containers used for this purpose, of which an example is shown in Figure 3-1, may vary from the common 200 L steel drums to precisely engineered thick-walled containers, depending on the nature and activities of the waste.

The use of waste containers has considerable advantages. They can facilitate the handling of the waste form considerably, facilitate transport, provide radiation shielding during emplacement operations and isolate the waste form from the groundwater at the disposal site. Since the last property is particularly important, waste containers are usually designed to protect the waste from the groundwater for a pre-defined period. It is therefore important that properties such as the mechanical strength, resistance to corrosion, wall thickness, the waste form and site characteristics be taken into account when designing such containers.

3.2.4 Organisational Responsibility

The responsibility for radioactive waste management can be given to a central organisation or a group of organisations. ENRESA in Spain, for example, have responsibilities covering the management of all the waste including the decommissioning of nuclear facilities (OECD/NEA, 1996). These responsibilities include

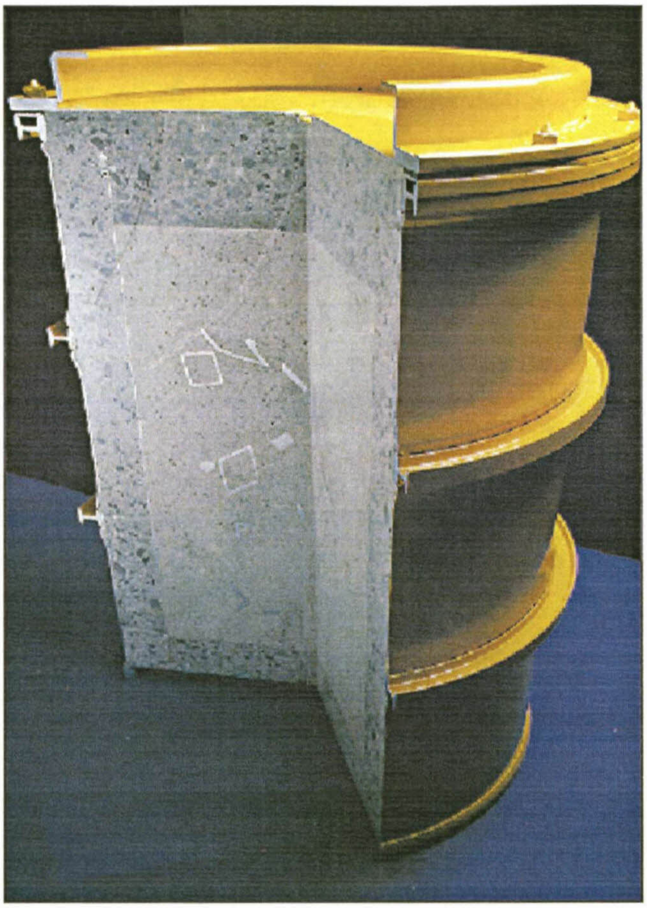


Figure 3-1 Model of a typical container used for the immobilisation of low- and intermediate-level waste (NAGRA, 1992).

(a) Handling and conditioning of radioactive waste;

(b) Siting, design, construction, operation and closing of interim and final disposal facilities for high, low and intermediate level waste;

(c) Management of activities derived from the decommissioning of nuclear and radioactive installations;

(d) Setting-up of systems for collection, transfer and transportation of radioactive waste;

(e) Conditioning, when required, of the tailings arising from uranium mining and milling, in a safe and definitive manner;

(f) Ensuring the long-term management of any waste disposal facility;

(g) Carrying out the necessary technical, economical and financial studies taking into account the deferred cost of radioactive waste management.

Integrated structures within one organisation like ENRESA will enhance the capability to recognise interdependencies between the various steps and to avoid conflicting requirements that could compromise operational and long-term safety (IAEA, 1995*a*).

3.3 MANAGEMENT STRATEGIES

3.3.1 Fuel Recycling

The radioactive waste produced on earth is rarely of any economic value, except for the spent fuel discharged from nuclear reactors. This fuel contains substantial quantities of materials, notably plutonium and uranium that can be used again as sources of nuclear energy. This process, generally referred to as *fuel recycling,* is illustrated schematically in Figure 3-2. However, fuel recycling can only be achieved if one can extract the plutonium and uranium from the spent fuel.

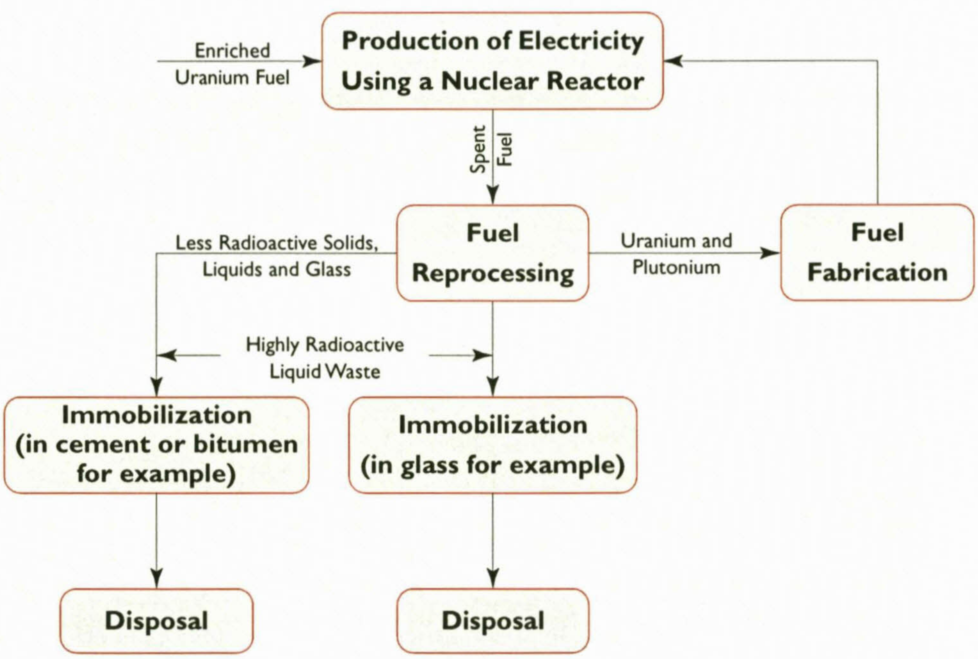


Figure 3-2 The different stages in the recycling of spent fuel used.

Fuel reprocessing is a costly process (AECL, 1994*a*) and produces highly active liquid waste containing fission and activation products, in addition to less active solid, liquid and gaseous wastes. Since the highly active liquid waste is usually more hazardous than the spent fuel itself, be it only for a limited time, special precautions have to be taken in handling it. One method used for this purpose is to dissolve the highly active waste in molten glass. This solution is then cast into blocks, which are allowed to solidify—a process called *vitrification*. The less active wastes are likewise dissolved in cement or bitumen which is then poured in metal drums to solidify (IAEA, 1993*b*).

Fuel recycling can be performed safely and practically on an economical basis in situations where there is a demand for spent fuel. The method can therefore be used to manage spent fuel to some extent, but it does not provide an effective solution to the problem of radioactive waste in general.

3.3.2 Storage of Radioactive Waste

Storage is another method often used in the management of radioactive waste. In this method the waste is placed in an isolated and controlled facility *with the intent to retrieve the waste* for exemption, reprocessing or disposal later (IAEA, 1993*a*). Storage is thus a temporary measure that can be applied at any stage of the management process. Nevertheless, there are sound technical and economical reasons why one would store radioactive waste. Spent fuel is, for example, usually stored in water pools, because of the excellent cooling and shielding properties of water. However, dry storage containers, in the form of concrete structures, have been developed for the long-term storage of spent fuel in Canada, Germany and USA (IAEA, 1992).

Another advantage of storage is that it can be used to reduce the quantity of volatile radionuclides in the waste, the strength of its radiation field and the production of decay heat. Waste consisting of short-lived radionuclides is consequently often stored until the activity has decayed to such an extent that the waste can be released within authorised limits to the environment. However, storage must be often applied today, not for technical, but socio-political reasons. This applies in particular to high-level waste, the disposal of which is often vehemently opposed by the public.

Although storage is feasible and safe, it remains a temporary solution and does not replace the need for the development and application of a long-term solution to the problem of radioactive waste (IAEA, 1992).

3.3.3 Disposal of Radioactive Waste

The term disposal of radioactive waste is commonly used to denote the emplacement of the waste in an approved, specified facility, also known as a *repository*, *without the intention of retrieving* it (IAEA, 1993*a*). Extreme care must therefore be exercised in the design and construction of such a repository and where it is placed, if one wants to satisfy the fundamental principles of radioactive waste management in Table 1–1.

Although there are other options, the disposal strategies in use today are usually limited to *near-surface disposal* and *geological disposal*. Near-surface disposal is, per definition, the disposal of waste on or below the earth's surface. The repositories used in near-surface disposal systems usually consist either of an engineered disposal vault, situated below or above the earth's surface, or a trench dug into the surface. Earth trenches are normally backfilled with a suitable material and sealed off with an engineered, compacted clay cap of a few metres thick, to prevent infiltrating rainwater from reaching the waste. These repositories are conventionally used for the disposal of low- and intermediate-level wastes.

A special case of near-surface disposal, that is becoming popular, is *borehole disposal*. In this method the waste is emplaced in a borehole drilled to depths between 40 m to 100 m. One example where this method is used, is the Greater Confinement Disposal (GCD) Facility at the Nevada Test Site in the USA, which is used for the disposal of high specific-activity LLW (Price *et al.*, 1996). Another example is the BOSS (Borehole disposal Of Spent Sources) disposal concept that is currently under development in South Africa for the disposal of spent sources (Van Blerk *et al.*, 1999). These facilities are often considered as near-surface facilities, but may be more appropriately called intermediate depth disposal systems, since the boreholes are considerably deeper than most existing near-surface facilities.

Geological disposal involves the isolation of the waste, through a system of engineered and natural barriers, at depths up to several hundred metres, in geologically stable formations, with low permeabilities. No such permanent repositories have yet been constructed, but the experimental facilities usually consist of a vault excavated a few hundreds of metres deep in the host rock, and divided into separate disposal chambers. Access to the chambers is achieved through vertical shafts, extending from the soil surface to horizontal tunnels that connect the disposal chambers with one another.

Although geological disposal can be used for the disposal of all radioactive waste, near-surface and intermediate depth disposal are more economical in many cases. This is especially true in the case of the waste containing only short-lived radionuclides, or low levels of long-lived radionuclides. Geological disposal is consequently mainly reserved for the disposal of high-level waste and waste containing long-lived radionuclides.

Another disposal option that has been investigated extensively, but is not considered viable today, is the disposal of radioactive waste in the sea. Here, one should distinguish between disposal in the sea, the seabed and sub-seabed. Two interesting variants of this option that have been proposed recently is to dispose the waste in the vicinity of subducting plates, or in stable mudflats on the sea floor (Hollister and Nadis, 1998).

It follows from the preceding discussion that much can be gained if site-specific characteristics, such as the shielding power of the natural geological formations, are taken into account when designing a disposal facility. This power, unfortunately, varies considerably from one formation to the next, with the result that it may be possible to dispose one type of waste safely in one geological formation, but not in another. This means that one has to design a set of so-called *waste acceptance criteria* for a repository, to ensure that the waste will not pose unacceptable risks to future generations and the environment.

The introduction of waste acceptance criteria does not mean that society will not have to implement a long-term institutional control of the repository, but that the risk posed by the waste to humans and the natural environment will be limited if the controls fail. For example, the possibility that radioactive wastes can escape from stable, low-permeable formations is small. Disposal is therefore, at least in principle, able to meet one of the main objectives of radioactive waste management, in Chapter 1—to minimize the burden of radioactive waste produced today on future generations. However, this does not imply that the present generation, which derives a significant benefit from the use of radioactive materials, can rely solely on the geological environment to protect future generations from the adverse effects of radioactive waste.

The disposal of radioactive waste is, despite its advantages, a controversial subject on the local, national and international level. It is therefore of the utmost importance to have a legal framework and well established procedures to decide when and where the method can be implemented. Such an approach has an additional advantage in that it can assist the decision-makers in the evaluation of alternatives, and also ensure that all the relevant factors are addressed in the final decision. This approach will be particularly attractive in a society able to assign a proper mandate and responsibilities to the authorities, and partakes actively in the evaluation of the strategy and its implementation (Forsström and Westerlind, 1997). Implementation of the international objectives for the disposal of radioactive waste, that will now be discussed, can contribute significantly in developing such an approach.

3.4 RADIATION PROTECTION OBJECTIVES

The aim of all protection objectives for the disposal of radioactive waste is to protect humans and the natural environment, now and in the future, from adverse effects of the wastes. This implies that future generations should be afforded at least the same degree of radiation protection as given to the public today and that the safety of wastes should not depend on the active maintenance of the disposal system beyond a certain period.

One approach to achieve this aim is to have a clear and consistent set of radiation protection goals to ensure that the risk to any member of the public, when exposed to the radiation from radioactive wastes, will be as small as possible. The current international set of protection goals is embodied in the so-called 'Framework for Radiological Protection', which is based on recommendations of the International Commission on Radiological Protection (ICRP, 1977). The goals form the basis of the regulations for radiological protection implemented in many countries and the safety standards of the IAEA (IAEA, 1989, OECD/NEA, 1977).

The basic components of the ICRP recommendations, as set out in Paragraph 112 of their Publication 60, can be briefly summarized as follows (ICPR, 1997):

(a) **Justification of a Practice** – No practice involving exposures to radiation should be adopted, unless it produces sufficient benefit to the exposed individual, or society, to offset the radiation detriment it causes.

(b) **Optimization of Protection** – All reasonable steps should be taken to adjust the protection for any particular source of radiation, so as to maximize its net benefit to the society, subject to the prevailing economic and social factors.

(c) **Application of the Individual Dose Limit** – A limit should be applied to the dose an individual may receive from exposures to radiation sources, other than medical exposures.

To achieve this goal the ICRP (1997) recommended that the following procedures be followed in the radiological protection of the public against waste disposal practices.

(a) The control of public exposure to waste disposal should be exercised by using constrained optimization of protection. To allow for exposures to multiple sources, the maximum value of the constrained exposure to a single source should preferably be less than 0.3 mSv a^{-1}, while the total exposure should not exceed 1 mSv a^{-1}.

(b) In situations where environmental monitoring is used to monitor the confinement of radioactive wastes, derived restrictions should be developed for application to the monitoring results. Since environmental monitoring is often used to assess the combined implications of all the relevant practices, these restrictions should be based on a dose to the critical group approaching 1 mSv a^{-1}.

The IAEA initiated in 1977 an integrated program for the geological disposal of radioactive waste. One part of the program was to recommend basic requirements for the protection of human health and the natural environment, and another to establish criteria for the underground disposal of solid radioactive waste (IAEA, 1983). Two of the recommendations that emerged from the program have interesting consequences for the disposal of high-level waste (IAEA, 1989):

(a) **Radiological Safety** – Ensure the long-term radiological protection of humans and the environment in accordance with the current internationally accepted radiation protection principles.

(b) **Responsibility to Future Generations** – Isolate high-level waste from the environment over long time-scales, without imposing undue constraints on future generations, or require that they maintain the integrity of the disposal system.

The aim of the radiological safety objective is clearly not only to protect human health and the natural environment from the potential harmful effects of the waste at the time a disposal concept is implemented, but also far into the future (AECL, 1994*a*).

There are a number approaches that can be followed to satisfy the responsibility to future generations (AECB, 1987). One is to restrict the disposal options to those that are technically and economically feasible, and socially acceptable, and do not rely on long-term institutional controls to ensure safety in the future.

Although not stated explicitly in these recommendations, the basic principles of the protection of the environment are clear (IAEA, 1992). The ecological balance of areas should not be disturbed unnecessarily and the pressure on rare and endangered species should not be increased. Known toxic materials should also not be released into the environment without careful assessment of their potential for re-concentration or accumulation.

3.5 WASTE DISPOSAL PRACTICES

3.5.1 Low- and Intermediate Level Waste

Most low- and intermediate-level wastes around the world are disposed of in near-surface disposal facilities, consisting of unlined trenches and pits or engineered concrete structures. These facilities are usually not more than 20 m deep. Currently, however, there is a tendency to dispose these wastes in deeper facilities. In the United Kingdom, for example, no further shallow land repositories will be constructed, except for the expansion of the Drigg facility (IAEA, 1993*b*).

Depending on factors such as the characteristics of the waste, the site, the sources, climatic conditions and legislative requirements, different repository designs have been adopted in different countries. In some countries, the conditioned waste is placed directly in contact with the geological material, while others use special engineered structures, such as earth-mounded concrete bunkers and concrete vaults below the surface (Phillip and Clifton, 1989). Special caps, such as the one at the Intrusion Resistant Underground Structure facility (IRUS) at Chalk River in Canada (Dolinar *et al.*, 1996) are also used sometimes. Since the success of a near-surface disposal facility will ultimately depend on the capability of the system to prevent the migration of radionuclides, the precise structure of the repository may not be that important. What is important is that the repository *should minimize the contact time between waste andpercolating water or groundwater*. Systems with engineered barriers are consequently becoming more common (IAEA, 1993*b*).

Deep disposal facilities for LLW and ILW have been or are being constructed and operated in some countries. These facilities are still near-surface, although it makes more sense to refer to them as intermediate depth disposal. In this case, disposal is performed in low permeable geological formations, at depths of tens to hundreds of metres. Waste acceptance criteria for these facilities can be quite different from near-surface disposal, since the desired isolation capabilities of the system are much greater. Examples of such facilities include:

The Asses salt mine located in a salt diapir in Lower Saxony, Germany, where disposal took place between 1967 and 1978 at a depth of 500 m for LLW and 700 m for ILW (Brewitz, 1986).

The salt dome Morsleben facilities in the former German Democratic Republic (Ebel and Richter, 1986).

The Forsmark facility in Sweden, which is excavated in crystalline rock underneath the Baltic Sea with a rock cover of 60 m (Carlsson *et al.*, 1990).

The Waste isolation Pilot Plant (WIPP) repository in the USA, constructed in a bedded salt formation, about 660 m below the surface, for the disposal of transuranic waste from defence activities (USDOE, 1990).

The Konrad iron mine in Germany, where the overlying formations consist mainly of argillaceous sediments with a total thickness of several hundreds of metres. (Holtz, 1986).

The Olkiluoto repository in Finland, constructed in hard rock, at a depth of 50 to 100 m (IAEA, 1993*b*).

Another example of intermediate depth disposal is the GCD Facility at the Nevada test Site referred to above. In this facility the waste is emplaced in the bottom 15.2 m of a borehole, 3 m in diameter and 36.6 m deep, while the rest is back-filled with native alluvium (Price *et al.*, 1996)

An alternative waste disposal concept is deep-sea disposal. This concept is technically feasible with negligible environmental impact, in part because of the enormous dilution potential of the sea. However, there is an international non-binding moratorium by the London Dumping Convention on sea disposal of radioactive waste since 1983 (IAEA, 1993*b*).

3.5.2 High-Level Waste

There is no real urgency to dispose HLW and spent fuel, since vitrified HLW and spent fuel can be stored safely for many years (IAEA, 1993*b*). However, as mentioned in Section 3.3.2, storage is not a permanent solution. Various concepts for the disposal of HLW are consequently being developed at the moment. Three concepts for the disposal of spent fuel that have received some international attention recently (AECL, 1994*a*), include:

Transporting the waste into outer space.

Transmuting some or all of the radioactive nuclides in the waste to stable nuclides.

Geological disposal in stable, rarely disturbed geological mediums, such as glaciers, deep-sea sediments, or rock below the seabed, and sediments or rock on land.

To transport an entire bundle of spent fuel (or reprocessed waste for that matter) into space will be prohibitively expensive. Moreover, one cannot dismiss the probability of a launch failure and the associated radiological risk it may pose for some parts of the earth.

Although transmutation is, strictly speaking, not a disposal method, it is included in this discussion because it is the only method that has the potential to eliminate radionuclides from the waste quickly. The transmutation technology available today is, unfortunately, not very suitable for this purpose. Nevertheless, the method has considerable potential; particularly if it can be applied in conjunction with the reprocessing of spent fuel.

The geological disposal of radioactive waste in glaciers and other ice sheets on earth is feasible, but will probably not be acceptable to the public. Sea disposal of high-level waste is also prohibited by the London Dumping Convention, despite the same attractive features, such as the high dilution capacity of the sea (IAEA, 1993*b*). A related method, but not sea disposal per se, is the one recently proposed by Hollister and Nadis (1998). The essence of their proposal is to dispose the waste at depths ≥4 000 m in the geological stable and biological unproductive mudflats, poised in the middle of the larger tectonic plates on earth. However, it may take time before the international community accepts this proposal, if ever. Geological disposal on land is therefore the only remaining alternative.

No repository for the geological disposal of HLW or spent fuel is in operation yet. However, many countries are investigating potential host rocks (IAEA, 1993*b*). Several countries (Denmark, France, Germany, the Netherlands, Spain, the USA and the former USSR) are considering salt rock, while others (Argentina, Canada, Finland, France, India, Japan, Spain, Sweden, Switzerland, the UK and the USA) consider the more conventional igneous, metamorphic and sedimentary rocks. Countries such as Belgium and Italy also investigate argillaceous sediments, with their low permeabilities, high sorption capacities (for most radionuclides) and high plasticity.

Some countries have already developed underground research laboratories for this purpose (Kickmaier and McKinley, 1996), while others have developed conceptual repositories (AECL, 1994*a*). One site that has been investigated in considerable detail is the Yucca Mountain Site in Nevada, where the waste will be disposed in unsaturated tuff. The repositories for the envisaged sites will all be deep-seated and surrounded by multiple artificial barriers, to ensure that any release of radioactivity to the environment will occur at an acceptably low rate.

3.5.3 Spent Sealed Source Disposal

A concept for the disposal of the spent sealed sources discussed in Section 2.4.3 requires special consideration for a number of reasons. Large numbers of spent sealed sources exist in many countries, including the Russian Federation (former USSR) and developing countries. In some cases, no other nuclear activities exist in the country and therefore these sources are the only radioactive waste that needs to be stored and disposed. Even countries with nuclear fuel cycle facilities and with either operational or planned near-surface repositories find spent sources troublesome to manage, as they need to be handled separately from other types of radioactive waste. For example, some of these sources have very high activities and cannot be disposed in near-surface facilities. However, the small volume of this type of waste does not justify a geological disposal concept.

This situation prompted the AEC to introduce the BOSS disposal concept (Van Blerk *et al.*, 1999), which uses a borehole drilled down to a 100 m as repository and stainless steel containers as waste packages. In this concept, the bottom 50 m is used as disposal zone, while the remaining 50 m are back-filled to prevent intrusion. This BOSS disposal concept, which is still under development, will be discussed in more detail in Chapter 5. The initial safety assessment of the concept indicates, nonetheless, that it is robust and provides a viable solution for the disposal of spent sealed sources (Kozak, *et al*, 1999). The IAEA, however, has not yet adopted an official position on the disposal of spent sources in boreholes.

3.6 NUCLEAR LIABILITIES

3.6.1 Definition of Term Nuclear Liabilities

The term nuclear liabilities is a fairly new concept that has not been applied very much in the nuclear industry. There is therefore no universal consensus how it should be interpreted and applied in the industry. The definition that will be used in this thesis is given in OECD/ NEA (1996) who defines nuclear liabilities as

> *"...costs, which an organisation is expected to have to meet in the future as a consequence of its current or past nuclear operations".*

This definition contains carefully selected phrases with specific implications that need to be interpreted very carefully. The term *cost* in the definition clearly refers to cost expected to be incurred in future. This cost can be expressed in terms of either current money values (CMV) or net present values (NPV). In the CMV approach future liabilities are evaluated in terms of what it would cost to meet it today. This figure is then adjusted annually for inflation and periodically revised to take account of technological or regulatory changes affecting the costs. The value ascribed to a future liability is clearly independent of the time at which the expense will actually occur in this approach. In the NPV approach the current cost of a liability is estimated and then projected onto the expected time frame. An estimate is then made of the net present value of the cash needed at the time, based on an assumed discount rate. The estimated costs in the NPV approach can of course also be revised to account for inflation and technological change. However, timing is crucial in this method, since the later a liability is incurred the more will it be discounted.

The role of the *organisation* in the definition will largely depend on the legal requirements in a country. In principle, one can distinguish between generators of waste and operators of waste. In South Africa the generator is the producer of waste and is also the legal owner of the material, with the important implication that it should also bear the financial burden for all consequences flowing from the ownership of the waste. The operator, on the other hand is deemed to be the manager of the waste. His main responsibility is therefore to ensure that the waste is safely handled on behalf of the waste generator.

The phrase, *have to meet,* stresses the obligation resting on the liable party to manage its waste in accordance with legal requirements. The liable entity is also required to make provision for the future management of these liabilities.

How the term *future* in a liability assessment framework is interpreted depends on certain legal requirements and assumptions. For example, if the liability of a certain organisation is considered to be discharged at the time the waste is disposed, then the State or an equivalent institutional control body should accept any liabilities that may occur after disposal.

The term *consequence* in the definition simply acknowledge that any consequences of nuclear activities is a liability that needs to be accounted for.

Current or past nuclear operations suggest that one should distinguish between current liabilities based on current activities, and historic liabilities based on past activities. Current liabilities can be considered as liabilities created after a previous liability assessment. The current activities should always be managed in an optimised way to ensure a minimised current liability. The historic liabilities, incurred from past operations when inadequate provisions were made for such obligations, can be reduced through carefully designed liability reduction programmes. This can be enhanced, for example, through new waste management techniques that may result in more effective waste management processes.

3.6.2 Sources of Nuclear Liabilities

The nuclear industry can be conventiently divided into five sectors: mining and milling, fuel cycle activities, nuclear power plants, research centres and production of radioactive isotopes, for the purpose of identifying the various sources of nuclear liabilities.

Future liabilities in the mining and milling of uranium and thorium ores arise primarily from the need to ultimately decommission the mine, mill, surface facilities and the waste management areas. Although the volumes of waste in this sector are very large, the level of radioactivity is rather low, with the result that the liabilities associated with the waste are relatively small.

The nuclear fuel cycle sector encompasses all nuclear fuel processing plants, including facilities for the enrichment, conversion, production and reprocessing of fuel, as well as the conditioning of spent fuel. Nuclear liabilities are incurred through the large volumes of depleted uranium tailings, arising from the enrichment of uranium, and the decommissioning of the fuel production plants. Liabilities also arise from the management of the produced waste, such as reprocessing, vitrification, storage and disposal. Research centres and owners of nuclear power plants, such as electricity utilities, face similar liabilities with the addition of liabilities associated with the management and disposal of the spent fuel, the low- and intermediate level waste produced by the plants and the decommissioning of the plants.

The nuclear liabilities of centres that produce high-level or long-life radioactive isotopes (e.g., ^{60}Co, ^{99}Te, and ^{241}Am), for the medical profession, research, industry and agriculture, arise from waste produced during the production of the sources, the management and disposal of spent sources and the decommissioning of the producing facilities.

It must be remembered though that liabilities may also arise after the waste disposal system has been closed, or even abandoned. Since the activity of radioactive waste decreases with time one can expect that the associated liabilities will also decrease with time, at a differential rate that increases slowly with time at first, but later more rapidly. Nonetheless, this 'post-closure' liability cannot be neglected and must be carefully considered in the post-closure assessment of a radioactive waste disposal system.

3.6.3 Nuclear Liabilities Management

It is clear from the preceding discussion that the major sources of nuclear liabilities are the costs of cleaning contaminated sites and facilities, and the costs associated with the management of the radioactive waste. These costs will clearly depend not only on present decisions, but also on decisions taken earlier in the management of the radioactive waste. For example, a site previously selected for the temporary storage of certain types of waste, may eventually be used as a disposal site for all types of waste, which can place considerable liabilities on the owner in future. Moreover, all activities performed within the safety assessment framework will produce costs that must be accounted for in the calculation of nuclear liabilities. It therefore makes sense to consider nuclear liabilities and the management of nuclear liabilities as an integral part of radioactive waste management, if one does not want to violate the eigth principle of radioactive waste disposal in Table 1–1. This applies in particular to the 'post-closure' liability introduced above. What one should guard against, though, is to regard nuclear liabilities as more important than the fundamental object of a post-closure assessment—the reasonable assurance that the disposed waste will not affect future generations and the environment adversely.

CHAPTER 4

COMPONENTS OF A RADIOACTIVE WASTE DISPOSAL SYSTEM

4.1 INTRODUCTION

The discussion in Chapter 3 has shown that disposal is the only method available for the long-term management of radioactive waste. However, disposal as a selected strategy brings about certain responsibilities and need to be dealt with within the framework set by the principles of radioactive waste management. As shown by the discussion in Chapter 1, this objective is best achieved by performing an integrated safety assessment of the disposal system. As mentioned there, the discussion in this chapter will mainly concentrate on the components of a disposal system and how they are incorporated in a safety assessment.

The components of a disposal system can be conveniently divided into two classes: *internal* and *external components*. The internal components are those components that are situated within the spatial and temporal boundaries of the system, while the external components are situated outside these boundaries. These components can often be further divided into a number of subsystems or components, that are linked to one another through various internal and external components or as it is more commonly known, *features, events and processes* (FEPs), as presented in Figure 4-1.

The external components in Figure 4-1 are usually the components over which people has the least control. The discussion in Section 4.2 shows that these components are mainly the result of natural and anthropogenic events induced on a regional scale.

The earth's surface has been used as a natural disposal site for anthropogenic wastes since times immemorial, without any disastrous effects on the environment. One reason for this is that the surface can easily absorb and decompose the waste, but only to a certain limit. Unfortunately, this limit is often exceeded in situations where large volumes of hazardous waste are confined to a small area, such as modern waste repositories. Some of the remaining wastes may therefore dissolve in water that infiltrates the site to ultimately reach an underlying aquifer, which may transport the dissolved wastes over vast distances and areas. This migration of the dissolved wastes is fortunately a natural phenomenon that can be restricted, to some extent, by applying one or more of the following procedures.

(a) Select a site with characteristics that will reduce the movement, for example where the infiltration is small and the underlying geological formations are not very permeable.

(b) Study the physical, chemical, biological and radiological properties of the waste that need to be disposed of in detail, and try to reduce its mobility by further processing or conditioning.

(c) Reduce the mobility of the waste with suitable engineered barriers.

A more detailed discussion of these procedures can be found in Section 4.3.

4.2 EXTERNAL COMPONENTS

4.2.1 Repository Factors

The repository factors in Figure 4-1 are those factors that need to be considered as part of

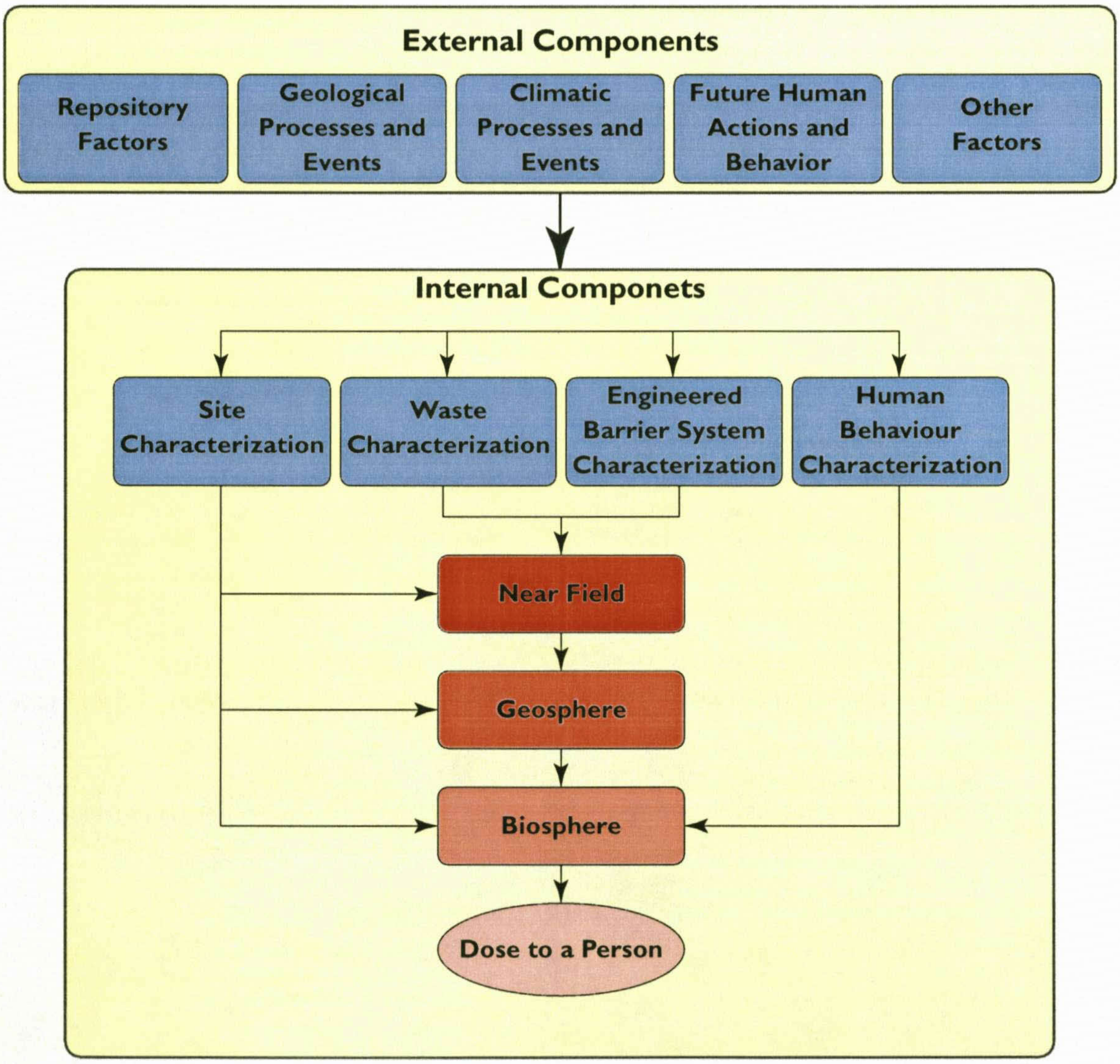


Figure 4-1 Conceptual representation of the different components of a disposal system, and the flow of information between them.

the planning, design, preparation, operation and final closure (e.g. capping) of the repository. Some of the more important of these factors for a safety assessment are summarized below, as a guideline for future investigators.

- The waste type(s) and volume(s) allocated to the repository.
- The design and physical dimensions of the repository.
- The scheduling and planning of events and activities that will occur during repository excavation, construction, waste emplacement and closure;
- Quality control, assurance procedures and tests that have to be performed during the design, construction and operation of the repository, and the form in which the waste will be delivered.
- The configuration in which the waste will be emplaced, and whether back filling should be continuous or not in the case of earth trenches.
- The retention of records of the waste emplaced in the repository, and the placement of markers at or near the site.
- Any special design, emplacement, operational or administrative measures needed to enable (or ease) the retrieval of the waste, if such an option is applied or considered.
- The monitoring system needed to ensure that the repository will meet the design criteria and that it will operate safely in the long-term.
- The cessation of the waste disposal operations at the repository and the subsequent closure operations, including the capping and rehabilitation, if applicable.

4.2.2 Geological and Climatic Processes and Events

The time scales of importance to the safety assessment of radioactive waste disposal systems range from 1 year to 10^6 years. It is therefore important to consider what potential influence geological and climatic changes, a few examples of which are given in Table 4-1, may have on the long-term performance of a disposal system. Many of these processes occur over geological time scales and may therefore not influence disposal systems for short-lived radionuclides significantly.

4.2.3 Future Human Actions and Behaviour

One factor that is frequently ignored in the design of a disposal system, but may have a major influence on its long-term performance, is the time span and degree of reliance that can be placed on institutional control of the facility. Closely related to this factor are future human actions and regional practices that can change the engineered barrier system and geological barriers—thereby affecting the performance of the disposal system adversely. These actions, of which a few examples is given in Table 4-2, may either be inadvertent or deliberate and may occur during the institutional control period or thereafter.

4.2.4 Other Factors

There are a number of extra-terrestrial and other perceivable factors that can have a considerable impact on the long-term performance of disposal systems. Two of these are comet impacts and the development of new species that threaten the existence of mankind, especially in and around repositories. Although such events cannot be completely discarded, the probability that they will occur is small. Moreover, the effects of some of the events, such as comet impacts, will be far more devastating than just a disrupted radioactive disposal facility, and can only be combated by society as a whole. They will consequently not be considered further here.

4.3 INTERNAL COMPONENTS

4.3.1 General

As mentioned above, the internal components are those components that are situated within the spatial and temporal boundaries of the disposal system. According to Figure 4-1 these components consist of the site information, physical, chemical, biological and radiological properties of the waste, engineered barrier system information and human behaviour characteristics that could influence the performance of the disposal system. These components are used in combination with the external components to define the migration of dissolved waste through what is known as the near-field, geosphere and biosphere. It is therefore necessary to characterize these internal components to quantify the dose a future individual may receive from the disposed waste.

There is no standard approach to the selection of a suitable site for radioactive waste disposal. Seeking the best site is not always possible or mandatory. What is required is that the site must meet the applicable national regulatory requirements. This means that the potential operator must demonstrate to the regulatory authority that the chosen site meets all the necessary regulations, and in some countries that concepts offering obvious advantages have not been overlooked. To achieve this the potential operator must obtain enough information to evaluate the suitability and long-term performance of the proposed disposal facility (IAEA, 1993*b*). This process is referred to as *site characterization* in the literature and

Table 4-1 Examples of long-term geological and climatic factors that may influence the safety of a disposal site adversely.

Changes in the sea level, caused by geological and climatic changes	Hydrothermal activities associated with high groundwater temperatures
Ecological responses to a change in climate	Long-term, global changes in the climate (from past, present and future evidence)
Extreme weather effects in tropical and desert areas	Periglacial effects
Erosion and sedimentation on the geological scale	Physical deformation of geological structures
Geomorphological changes	Seismic events and their potential to occur
Hydrological and geohydrological responses to climatic changes	Tectonic movement and oregeny
Hydrological and geohydrological responses to geological changes	Volcanic and magmatic activities

Table 4-2 Examples of future human actions that can affect the post-closure safety assessment of a radioactive disposal facility adversely and therefore should be included in a regional site characterization.

Mining and other underground activities
Changes in the social and institutional structures
Non-intrusive site investigations, eg for human resources
Future drilling activities (in the repository or its surroundings)
Surface activities (excavations, site development, and archaeology)
Future actions needed to remedy problems with the waste repository
Knowledge of the location of the repository (inadvertent or deliberate intrusion)
Humankind's influence on the climate (emission of greenhouse gasses, deforestation)
Groundwater and surface water management activities (groundwater withdrawal, dam construction, irrigation schemes)

commences during the site selection phase.

4.3.2 Site Selection

Siting of radioactive waste disposal sites can be described as the identification of potential disposal sites (site screening and selection) followed by detailed investigations at one or more preferred sites (site characterization). Site screening is the identification of a number of potential areas or sites that may be technically suitable and acceptable to the public for disposal and that warrants the expenditure of resources for detailed investigations. The first step in the selection of a site for the disposal of radioactive waste is to decide what disposal method will be used. A site covered with unconsolidated sands will obviously not be suitable for near-surface disposal, but may be suitable for geological disposal, if underlain by thick impermeable rocks at suitable depths. Site characterization provides the technical information needed to design the disposal concept and to assess the performance of the disposal system. One or more of these sites are then selected as the candidate site and characterized in detail, with the view to convince the regulatory authority that the site is suitable for a radioactive waste repository. Table 4-3 lists a few geological and surface features that can be useful in the regional selection of a site. More detailed discussions on the stages of site selection for geological and near-surface disposal facilities can be found in IAEA (1994*b*) and IAEA (1994*c*) respectively.

Before leaving this discussion of site selection it must be pointed out that it will be difficult to get a site selected without the active participation and co-operation of the local populations and other interest groups in all four phases of the site selection. This objective, also referred to as *confidence building* (see Section 6.6.7), may not be easy to achieve, particularly in the

Table 4-3 Examples of geological and surface factors that should be included in a regional site characterization.

Geological resources of the area
Evolution of the geological structures in the area
Characteristics of the aquifer systems in the area
Regional hydrology and geohydrology of the area
Characteristics of the regional weather and climate
Characteristics of the erosional and depositional evolution of the area
Properties and characteristics of large-scale geological discontinuities
Characteristics and evolution of the area's coastal and marine features
Chemical and geochemical characteristics of the regional geology and groundwater
Characteristics of terrestrial water bodies (lakes, rivers and streams) and their evolution
Characteristics of terrestrial water bodies (lakes, rivers and streams) and their evolution

case of high-level wastes (Easterling and Kunreuther, 1995).

4.3.3 Site Characterization

Site characterization, usually starts with the selection of a specific site during site selection. The level of activities during site characterization will therefore depend very much on the work done during site selection. Site characterization may therefore in some cases be restricted to a confirmation of the geohydrological conditions and geological sequence at the chosen site. However, it is essential that a comprehensive characterization of at least one site be performed, as this information is needed in the final assessment of the site (USDOE, 1988; IAEA, 1990).

Site characterization is usually performed in phases. The advantage of this approach is that it allows one to evaluate the process continuously and to determine whether the site remains suitable for further investigation. This not only enhances the inherent iterative nature of the site's safety assessment, but also facilitates modifications to or redesigning the investigation programme.

The main purpose of a site characterization is to determine what effect the presence of a repository will have on the environment and *vice versa*. It therefore mainly embodies the study of the near-field, geosphere and biosphere, as indicated in Figure 4-1.

4.3.4 Waste Characterization

Radioactive waste was qualitatively classified as low-level, intermediate-level and high-level waste in Section (2.4). However, the physical, chemical, biological and radiological properties of the waste can vary considerably within each level. It is consequently essential that a complete characterization of the waste be performed as part of a site assessment. Of particular importance in this regard is to establish the need for the further adjustment, treatment or conditioning of the waste, and its suitability for further handling, processing, storage or disposal (IAEA, 1993*a*). Other factors that are also important include: the waste type, its form (solid, liquid or gaseous), toxicity, level of radioactivity and radiation emitted and the quantity of long- and short-lived radionuclides in the waste (initial inventory) (IAEA, 1993*b*). A description of the initial inventory should include the decay chains, fission products and the half-lives of the daughter nuclides. In those cases where the reprocessing of the waste is considered, the heat output and volumes of waste will also be important. The information needed for the two radionuclides ^{60}Co and ^{90}Sr are presented as examples in Table 4-4.

Table 4–4 Information needed for a safety assessment analysis of ^{90}Sr and ^{60}Co (Seitz *et al.*, 1992).

Isotope	**^{90}Sr**
Half-life	28.78 years
Primary radiation and energies	β^-, 550 keV
Daughter Products	Yttrium (^{90}Y) Half-life: 2.671 d Emission: β^-, 2.27 MeV
Major Chemical forms	Sr^{++}, $SrCO_3$ (s)
Production methods	Fission Product
Waste stream	Nuclear power plants: Ion exchange resins, pre-coat filters, cartridge filters, concentrated liquid wastes, trash, dry active waste and waste from decommissioning
Major environmental pathway	Milk, Groundwater
Health hazards	Ingestion and internal exposure from beta emission ('bone seeker')

Isotope	**^{60}Co**
Half-life	5.271 3 years
Primary radiation and energies	gs: β^-, 2.7 MeV; m1: IT 59 keV (100%), β^-, 2.9 MeV (0.240%)
Daughter Products	Nickel (^{60}Ni) Stable
Major Chemical forms	Ni^{++}, $NiOH^-$, $NiCl_2$, $Ni(OH)_2$, NiS (s), $NiCO_3$
Production methods	Produced by the neutron activation of ^{59}Co
Waste stream	Nuclear power plants: ion exchange resins, concentrated liquids, filter sludge, cartridge filters, trash, dry active waste Waste from medical and academic institutions, teletherapy units, radiography cameras and geological and biological studies
Major environmental pathway	Groundwater (highly retarded by sorptive processes)
Health hazards	Direct exposure due to gamma emission, ingestion and internal exposure

4.3.5 The Engineered Barrier System

Engineered barriers, constructed from materials with favourable properties and predictable nuclide retention capacity, usually provide the primary containment of the waste in a modern disposal system. The possibility therefore exists that most radionuclides will decay to insignificant levels within the barriers, if the disposal concept and the construction of these barriers are implemented correctly (Smith *et al.*, 1994). However, nothing on earth is perfect. These barriers should therefore be constructed in such ways that they will still limit the release of nuclides to the geosphere in case they fail. This can only be achieved if the designer has sufficient information on the physical, chemical and biological properties of the proposed barrier materials, the surrounding near-field, geosphere and biosphere.

The first strategy conventionally employed to prevent the migration of the nuclides from the repository is to condition the waste, as discussed in Section 3.4. The ability to design a 'perfect' container for a specific disposal site will, however, be limited by economic and construction factors and the geometry of the waste form. The containers used in practice are usually cylindrical in form, mainly for two reasons: the cylinder has the largest volume to surface area ratio of all solid figures, except the sphere, and it is relatively easy to handle. However, this means that there will always be gaps between the containers and the repository wall, when the containers are put in the repository. This could expose large parts of the container to interactions with its immediate surroundings, thereby reducing its effectiveness. Repositories are consequently usually back-filled with suitable materials that will reduce the interaction of the containers with their immediate surroundings as much as possible. Considerable care must therefore be exercised in choosing the backfill material. Some of the more important interactions to consider in choosing the back-fill material are:

(a) the mobility of the nuclides from the repository and the corrosivity of the containers,

(b) the structural stability of the repository,

(c) heat flow in and around the repository, especially in the case of HLW,

(c) intrusion of the repository by humans and other animals.

It will be very difficult if not impossible to completely prevent the intrusion of humans and animals with the materials available today, but the influence of the other interactions can be readily reduced. One way to achieve this is to choose materials with low hydraulic conductivities but high sorptive capacities, and at the same time reduce the corrosivity of the groundwater.

A number of materials with these properties have already been studied intensively both in the laboratory and in field (AECL, 1994*a*). However, it may not always be possible to use the most suitable material for a specific site, because very little may be known of the material's long-term performance, or the material may not be available in sufficient quantities. (Several million cubic metres of material may be needed to fill a HLW repository, for example.) Bentonite, fullers earth, and various clays, zeolites and cements are considered as the best back-fill materials at the moment (IAEA, 1993*b*).

Infiltrating rainwater poses one of the biggest threats to the integrity of a near-surface repository. It has therefore become practice to place an engineered cap on top of such facilities to prevent rainwater from entering the system. Depending on the size of the facility, these caps are normally not more than a few meters thick, and constructed in such a way that most of the rainwater is diverted to a drain. Clay compacted to a very high density and with a low permeability is a very popular material to use in engineered caps. However, clays often tend to fracture, especially during dry periods, although the fractures may close again if the clay is wetted sufficiently. Nevertheless it will be difficult, if not impossible, to ensure the integrity of the cap indefinitely. However, near-surface repositories are mainly used for low and intermediate level wastes, which need only be shielded for a limited time from infiltrating rainwater. It may therefore be possible to design a suitable cap for such a repository by assessing the behaviour of caps, constructed from different clays, under different infiltration rates during the design phase of a repository.

It is rather unlikely that humans, burrowing animals or plant roots will never intrude a near-surface repository in the future. Future engineered caps for these facilities must therefore not only be infiltration resistant, but also intrusion resistant. The Intrusion Resistance Underground Structure (IRUS) at Chalk River in Canada is an example of such a facility (Dolinar *et al.*, 1996).

4.3.6 Human Behaviour

The fundamental principles of radioactive waste management in Table 1–1 have one common objective: to protect humans and their environment from adverse effects of the waste. It should therefore not be a surprise to learn that this is indeed the main concern of the safety assessment of any radioactive waste disposal site.

Although some influences that humans may have on the integrity of a waste disposal site, have been discussed above, such a safety assessment should also include identifying the group of people (known as the critical group), most likely to be affected by the site, and their local habits. The factors listed in Table 4-5 may serve as a guideline as to the type information that should be gathered in this connection. For the time scales of concern, it is virtually impossible to define the actual critical group. For this purpose, hypothetical critical groups are defined, based on past experiences and the factors listed in Table 4-5.

Table 4-5 Examples of human characteristics that should be included for the critical group in the assessment of a radioactive waste disposal site.

Time spent in the environment
Community and social characteristics
Normal activities of the group and their use of materials
Dietary patterns of the group and their variation with age
Leisure and other activities associated with the environment
Rural, urban, agricultural and industrial use of land and water
Information about the dwellings or other structures of the target group
Anatomical and physiological characteristics and their variability with age
Activities associated with the processing and preparation of food and water

4.3.7 The Near-Field

The near-field can be defined as the excavated area of the disposal system, including the repository, its engineered barrier and the part of the surrounding geology whose characteristics have been or can be altered by the repository or its contents (IAEA, 1993*a*). The near-field may therefore be considered as the source of all changes the repository will induce on the environment.

As shown in Figure 4-1, the waste characteristics and the engineered barrier system will mainly determine the strength of the source. However, there is another factor, not mentioned explicitly in Figure 4-1 that will influence the strength of the source—the location and nature of the repository. As discussed above, the repository can consist of either a vault, a trench in the earth's surface, or a borehole, which may be situated completely or partially in either the unsaturated or saturated zone of the earth's surface. A detailed study of the geohydrology of a disposal site's saturated zone, unsaturated zone, or both, must therefore form an integral part of the site's characterization.

4.3.8 The Geosphere

The term geosphere is generally used to denote the host medium that surrounds the near-field. This includes the soil and rock formations that over- and underlie the repository and their water content. The geosphere therefore forms a natural barrier between the biosphere and the near-field. This is the main reason why near-surface and geological disposal systems are such attractive options for the management of radioactive waste. Of particular importance in this regard is the ability of the geosphere to maintain conditions favourable for long-term waste isolation by

(a) Limiting the rate at which contaminants can migrate from the near-field to the biosphere, and

(b) Protect the repository from natural disruptions and human intrusion.

Special attention should therefore be paid to the geosphere during site selection and characterization. A partial list of factors that are especially important in this regard is given in Table 4-6.

The ability of the geosphere to act as a natural barrier is determined by a number of factors, of which the most important are its hydrological and geochemical properties, and the physical isolation and stability of the disposed waste.

Two of the most important parameters to include in the characterization of the geosphere are the groundwater fluxes and flow paths. The piezometric levels in the aquifer(s), the hydraulic conductivities, storativities and porosities of the different stratigraphic units and their structure, essentially determine these quantities. Geological environments whose geohydrological properties are particularly suitable for the disposal of radioactive wastes include (IAEA, 1993*b*):

- rocks with a very low water content and permeability (e.g. evaporates),
- fractured rock of low intrinsic permeability, in which radionuclide transport would be controlled by the fracture network (e.g. crystalline rock in low relief terrain),

Table 4-6 Partial list of factors that are important in characterizing the geosphere of a disposal site.

Topographic relief of the site
Stability of the geological formations
Thermal environment of the geosphere
Biology and biochemistry of the geosphere
Local topography and morphology of the site
The presence or absence of mineral resources
Geohydrology and geochemistry of the geosphere
Animal and plant intrusions that may disrupt the repository
Gas sources and production and their effect on the geosphere
Properties and characteristics of discontinuities in the geosphere
The stratigraphy of the rock formations, their physical and chemical properties
Mechanical deformation of the geosphere caused by the construction, excavation and presence of the repository

- rocks with a potentially very low content of mobile water and permeability (e.g. argillaceous rocks).
- unsaturated rocks and deposits.

The ability of the geosphere to act as a barrier between the repository and natural environment can be further enhanced by the geochemical properties of the geosphere. This applies in particular to the ability of the geochemistry to slow or prevent the degradation of engineered barriers, and to limit the rate of mobilisation and transport of radionuclides, when they are released from the near-field. The most favourable geochemical conditions for this purpose are sites with a redoxing environment, that are well buffered by minerals in the rock formations, have low concentrations of complexing ligands, and groundwater with a high pH-value. It must be remembered though that both the waste type and the repository design will influence the geochemistry at a particular waste disposal site. This is particularly important when engineered structures are expected to be stable for long periods, or interact dynamically with the geosphere.

The geosphere can only be considered an effective natural barrier if it can isolate the wastes over the time scale for which the repository was designed. Waste disposal sites, particularly the sites for radioactive wastes, should therefore not be sited near active volcanic centres or major fault zones, Attention should also be given to the possible neo-tectonic deformation and the associated erosion rates of the site.

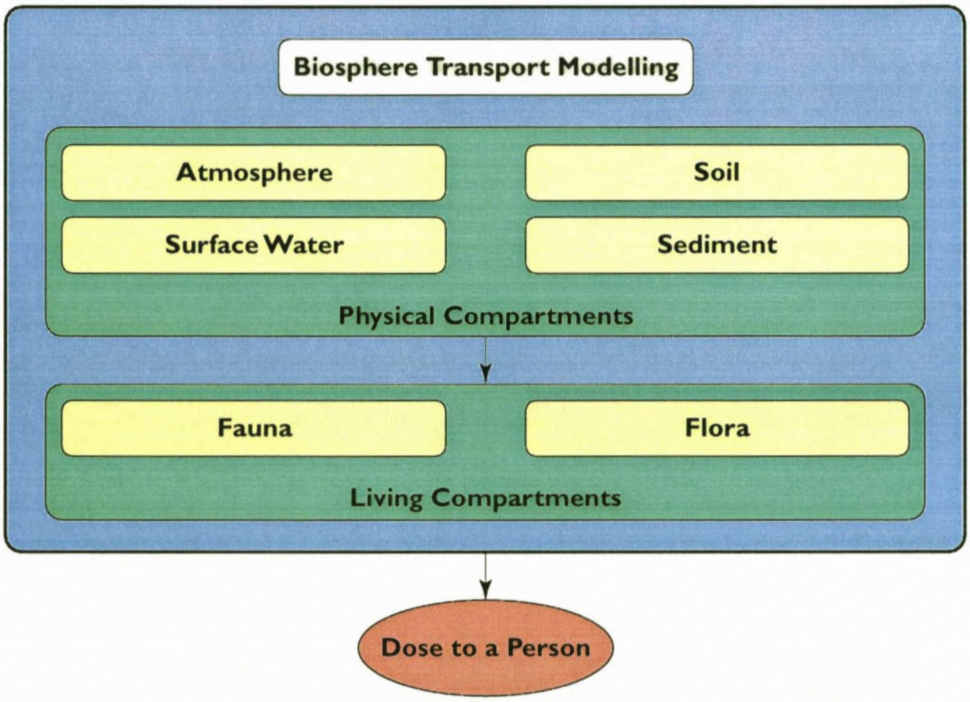
4.3.9 The Biosphere

The term biosphere is conventionally used to describe that part of the environment inhabited by all living organisms, no matter what their form or origin is. However, there is a tendency to restrict the term to those parts of the earth's atmosphere, hydrosphere and lithosphere associated with the activities of mankind, in the case of radioactive waste disposal (IAEA, 1993*b*).

The geosphere is obviously the domain mostly exposed to radionuclides that may leak from a repository. It is therefore unlikely that the radionuclides will affect most living organisms, especially people, before they reach the biosphere. The biosphere therefore represents the domain where living organisms will be exposed to radionuclides that have escaped from the near-field and geosphere. Since the major objective of radioactive waste disposal is to protect living organisms from the adverse effects of the waste, it is important to include the biosphere, be it only in the restricted form defined above, in the analysis of a disposal system.

The nuclides may follow numerous paths before one or other organism in the biosphere absorbs them. Torres and Simon (1997) devised the simple scheme of compartments in Figure 4-2 to describe this migration. Radionuclides are assumed to be instantaneously distributed within the compartments, but can be transported by different processes from one compartment to another. The behaviour of the people that may be potentially affected by the nuclides is particularly important in this regard.

Figure 4-2 Composition of the biosphere after Torres and Simon (1997).



There are situations where air is the major transport mechanism for radionuclides in the biosphere, for example, the movement of radon from tailings dams. However, water or the hydrological cycle in general, will be dominant transport mechanism at most waste disposal sites (Freeze and Cherry, 1979). Previous experience has shown, however, that the major contributions come from bodies of water, or *receptors* as Torres and Simon (1997) call them, such as boreholes, wells, watercourses (rivers, brooks, and streams), lakes and the oceans.

Boreholes and wells, although small in comparison with the other receptors, are particularly important for the transport of radioactive nuclides in the biosphere. The main reason for this is that they are the only receptors situated completely within the geosphere. The chances that the concentration of radioactive nuclides in their water will be diluted (as is almost always the case with the other receptors) are therefore zero, unless the aquifer on the site is multi-layered and not all the layers are equally contaminated. The travel time of contaminated groundwater from the repository to boreholes and wells is usually also much shorter than to the other receptors. These receptors may therefore become contaminated long before the other receptors.

Table 4-7 Summary of physical processes that may cause contamination of the terrestrial food chain by nuclides that leaked from a radioactive waste repository. [After Torres and Simon (1997).]

Uptake by plant roots
Interception and retention by vegetation
Retention by vegetables and soil surfaces
Depositions by either dry or wet processes, or both
Direct ingestion of surface soil by grazing animals
Transfer of the nuclides from groundwater to the terrestrial system
Transfer of the nuclides from surface water by droplets of water (e.g. mist)
Transfer of the nuclides from surface water to sediments and the aquatic biota
Processes responsible for the translocation of the nuclides from the deposition site
Transfer of the nuclides from the soil, air, water and vegetation into the milk and meat of grazing animals

There are a number of pathways along which radionuclides can move from the receptors to humans. For example, by eating fish and other seafood, swimming in contaminated waterways, lakes or the sea, and exposure to contaminated sediments on riverbanks, and the beaches of lakes and the sea. However, the most common pathways, particularly in areas where a disposal site is most likely to be situated, are through the domestic consumption of water and the food chain. A few examples of physical processes that are particularly important in the terrestrial food chain are summarized in Table 4-7.

CHAPTER 5

CONCEPTS FOR THE DISPOSAL OF RADIOACTIVE WASTE

5.1 INTRODUCTION

In this chapter, three examples of disposal concepts for radioactive waste will be discussed before presenting a broader perspective of the basic principles of a post-closure safety assessment.

As mentioned in Chapter 1 Vaalputs is the only one of the two sites, currently used for the disposal of radioactive waste in South Africa that underwent a detailed site selection and site characterization programme. The following discussion therefore begins with a description of the near-surface concept used for the disposal of low- and intermediate level waste at Vaalputs, in Section 5.2. This is followed by a discussion of the disposal concepts used at the other site, Thabana, in Section 5.3.

A permanent solution for the high-level waste generated by the Koeberg Nuclear Power Plant has not received much attention in South Africa. In fact, no conceptual plan of the disposal concept that will be used in South Africa has yet been formulated. The general idea now though is that the waste will be disposed at Vaalputs using a geological disposal concept. This is a very important problem for the present generation to solve, if we do not want to violate some of the fundamental principles of radioactive waste disposal, discussed in Chapter 1.

The geological characteristics of Vaalputs are very similar to the site proposed for geological disposal in Canada by Atomic Energy of Canada Ltd. (AECL). A geological disposal concept, based on the Environmental Impact Statement (EIS) prepared by AECL for the disposal of nuclear fuel waste in Canada (AECL, 1994*a*, 1994*b*), is therefore discussed in Section 5.4.

A large number of spent sources from the medical and other technical professions exist in many countries all over the world. Some of these countries, many of them from Africa, find it difficult to manage and dispose this waste. The AEC of South Africa therefore introduced the BOSS (Borehole disposal Of Spent Sources) disposal concept as part of an IAEA programme to strengthen the waste management infrastructure in African countries (AFRA I-14) (Van Blerk, *et al.*, 1999). This disposal concept is described in Section 5.5.

5.2 THE VAALPUTS NATIONAL RADIOACTIVE WASTE DISPOSAL FACILITY

5.2.1 General

In South Africa, the State accepts responsibility for disposing of radioactive waste, while the AEC was instituted as the agent of the state for this purpose by the Nuclear Safety Act of 1993. The National Radioactive Waste Disposal Facility at Vaalputs is almost exclusively being used for the disposal of low- and intermediate level waste from the Koeberg NPP. As discussed in Chapter 1, it went through a detailed screening, selection and characterization process, with the result that vast amounts of data and information are available. The screening phase commenced in 1979 and the following parameters were examined in detail (Moore and Hambleton-Jones, 1996).

Agriculture production	Climate and rainfall
Corrosion of groundwater	Ecologically sensitive areas
Industrial growth potential	Mineral occurrences
Population density	Proximity to international boundaries
Seismic hazards	Surface and groundwater hydrology

Vaalputs was identified as the most suitable site for the disposal of radioactive waste in 1983. This led to a more detailed site characterization programme that lasted from 1983 to 1986, after which the site came into operation. Characteristics that were identified as being particular favourable for the disposal of radioactive waste are: its arid and remote nature, very low population density, annual rainfall of only 80 mm year^{-1}, an unsaturated zone of 50 – 65 m, and its clayey soil properties. This led to the decision that near-surface earth trenches in the overlaying clayey soils will be sufficient for the disposal of low- and intermediate level waste and that concrete vaults are not necessary enhance the repository's safety.

5.2.2 Site Characteristics

The farm Vaalputs is situated on the edge of the western side of the escarpment that divide the inland and coastal plains in South Africa. As shown in Figure 5–1, the site is bordered by rugged, mountainous terrain on the west and the flat plains of the Bushmanland plateau, with an elevation of approximately 1 000 mamsl on the east. The repository is situated on this plateau.

Vaalputs is located in a transition zone between the winter and summer rainfall areas of South Africa with winter rainfall somewhat dominant. The annual average rainfall is 80 mm,

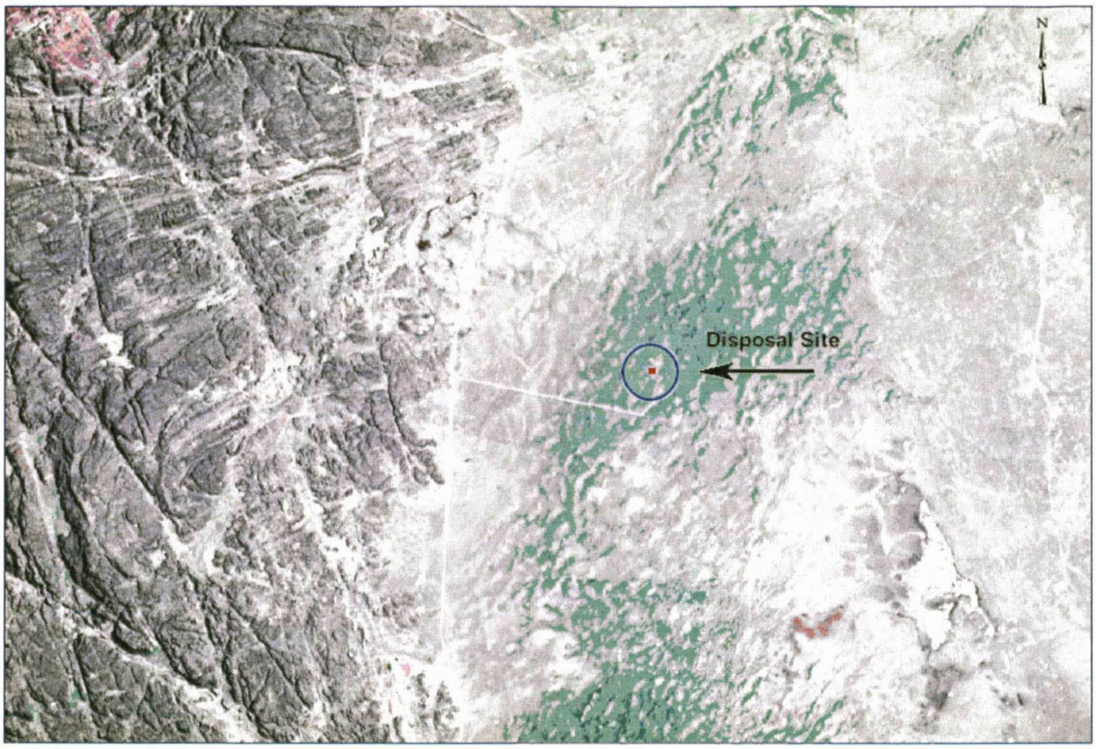


Figure 5–1 Space map (Bands 4,5,3) of the Vaalputs area. The dune fields are shown in light green, while the disposal site is shown in red.

while the potential evaporation is high with an annual average of 2 100 mm (Redding and Hutson, 1983). The local geology of Vaalputs, shown in Figure 5–2, indicates that Vaalputs is situated on the Precambrian Crystalline rocks of the Namaqualand Metamorphic Complex. The crystalline rocks form the basement are covered by younger sedimentary rocks. Metamorphism transformed the original sedimentary and volcanic rocks to granite-gneisses and metavolcanics.

The basement rocks belongs to the pre-tectonic Garies Subgroup of the O'Kiep Group and consists of light coloured biotite gneiss and quartzofeldspathic rocks (Brynard, 1988). The Vaalputs granite-gneiss is fine to medium crystalline with a uniform pinkish colour, with small amounts of garnet and biotite (Andersen, 1996). Large-scale tectonism folded, thrusted and fractured the bedrock extensively. The basement rocks were also intruded by granites, which caused extensive re-melting. The fractured basement rocks at Vaalputs are potentially important for geological disposal.

The near-surface disposal trenches are situated in the Vaalputs Formation. This formation consists of sediments - typical æolian sand, calcretized sandy, gritty clay and red to greyish fluvial gritty, sandy clay containing gravel and quartz pebbles - that accumulated in the Vaalputs Basin. Calcite veins cut through the whole Vaalputs sequence and are deeper than

Formation	Lithology	Age (Ma)	Geological Process	Geological Process
Kalahari Gordonia	Red sand	20–5	Aeolian dunes	
Vaalputs	Calcrete and silcrete Ferruginised sandy gritty clay Brown sandy gritty clay Grey sandy gritty clay with interbedded pebble bands Siliceous sandstone	35–20	Fluvial	Tertiary to Recent
Dasdap	Cross-bedded arkosic grit. Conglomerate	38–35	Alluvial fan	
Unconformity	Kaolinised and silicified surface			Late Cretaceous
Karoo Dwyka	Diamictite	300	Glacial	Jurassic-late Carboniferous
Namaqualand Metamorphic Complex	Basement granitoids	1 050	Tectonism	Pre-Cambrian
O'Kiep Garies	Metamorphic rocks		Intrusion	

Figure 5–2 Geological succession at Vaalputs. (Modified from Levin, 1988)

7 m (Levin, 1988). These veins resulted from infilled fractures. The Vaalputs Formation is overlain by north-east trending red Kalahari sand dunes of the Gordonia Formation that conform to the underlying geological structures.

5.2.3 Waste Characteristics

As mentioned above, Vaalputs is almost exclusively being used for the disposal of low- and intermediate level waste from the Koeberg nuclear power plant. It consists essentially of trash, equipment, filters, ion-exchange resins, evaporator concentrates and liquid chemical waste. The inventory for the low- and intermediate level waste disposed of at Vaalputs as at May 1998, is presented in Table 5-1.

The waste acceptance criteria for Vaalputs (AEC, 1997b) states that only solid waste will be accepted, while solidified or immobilized waste must contain no free-standing liquid. Waste containing transuranic nuclides may be accepted provided that the specific activity of such waste is less than 370 Bq g^{-1} and that they are evenly distributed within the waste form. The maximum permitted surface contact dose rate for any single waste package is 2 mSv h^{-1}. Physically, the mass of any individual package shall not exceed 6.3 ton, while the maximum dimension of any waste package in any direction, shall be 2 m. No explosives or pyrophoric materials shall be dispatched to the site.

5.2.4 Waste Packages

The only packages used for disposal at Vaalputs to date are the metal drums and concrete

Table 5-1 The low- and intermediate level waste inventory at Vaalputs in May 1998.

Isotope	Low Level Waste (GBq)		Intermediate Level Waste (GBq)	
	Received	Decayed	Received	Decayed
^{110}Ag	2.66E–04	1.31E–06	7.66E+02	4.46E–01
^{14}C	1.12E–02	1.12E–02	2.60E–01	2.60E–01
^{60}Co	1.19E+04	5.93E+03	6.33E+04	2.43E+04
^{134}Cs	3.91E–02	6.58E–03	2.46E+03	2.13E+02
^{137}Cs	3.56E+03	3.15E+03	1.13E+04	9.53E+03
^{134}Ce	0.00E+00	0.00E+00	8.94E+02	1.51E+00
^{3}H	3.56E+02	2.64E+02	4.07E+03	2.73E+03
^{129}I	9.62E–08	9.62E–08	3.71E+00	3.71E+00
^{54}Mn	1.47E–02	2.10E–04	1.43E+03	4.10E+00
^{90}Sr	2.33E–04	2.10E–04	1.43E+03	4.10E+00
^{99}Tc	4.49E–08	4.49E–08	1.34E–03	1.34E–03
Tru	7.68E–06	7.68E–06	4.13E+00	4.13E+00

containers shown in Figure 5–3. The concrete containers are all 1.3 m in height, with an outside diameter of 1.4 m. Depending on the waste characteristics the wall thickness will vary between 150 and 400 mm to ensure proper shielding against radiation. The metal drums are standard 200 l drums, with a height of 875 mm and an outside diameter of 585 mm.

The low-level waste is compacted in the metal drum, while cement is used to solidify the intermediate level waste in the concrete container. The concrete container is fitted with a steel liner and a concrete cap. A gap between the steel liner and the concrete wall allows for expansion of the core, while the cement immobilising the waste is setting. Separate trenches are used for the concrete containers and the metal drums. This arrangement simplifies inventory control, since most of the activity is confined to the concrete containers. It also adds to the stability of the intermediate level waste, as these containers tend to degrade and corrode less rapidly than the metal containers.

5.2.5 Physical Dimensions of the Waste Disposal Trenches.

The near surface trenches used for the disposal of low and intermediate level waste at Vaalputs are 100 m long, 7.7 m deep and 20 m wide at the bottom and with sides that slope upwards at an angle of 80^0, see Figure 5–4. The trenches are filled up from the bottom with a drainage layer of 0.2 m gavel, waste containers (5.2 m in height), a compacted clay cap of 2.0 m, and a top sand layer of 0.3 m. Screened clay is used as backfill between the waste packages.

The original idea was to fill each trench completely before it is sealed off. However, the

Figure 5–3 The waste packages used for the disposal of radioactive waste at Vaalputs.

volume of waste received at Vaalputs is considerably less than expected during the design and construction of the trenches. No trench has consequently been completely filled since the depository began to accept waste in 1986. Since the weather conditions at Vaalputs corroded the containers in the open more rapidly than originally anticipated, the trenches are now divided into sections by concrete walls, as shown in Figure 5–4, and each section backfilled and covered with sand when full. The trench will finally be provided with a compacted clay cap to serve as an engineered barrier to rainwater infiltration. The cap will be 1.5 m thick at the edge and 2.0 m at the trench centre, with a slope of 5% on its top surface. It will be compacted in layers approximately 150 mm thick to achieve optimal compaction. The cap will be covered with a layer of the original sand, at least 300 mm thick. The whole burial area will finally be restored to its original topographic contours.

5.3 THE THABANA RADIOACTIVE WASTE DISPOSAL FACILITY

5.3.1 General

As discussed in Chapter 1, very little is known about the criteria used to select Thabana as a disposal facility. The only information available is a report by Niebuhr *et al.* (1968). In

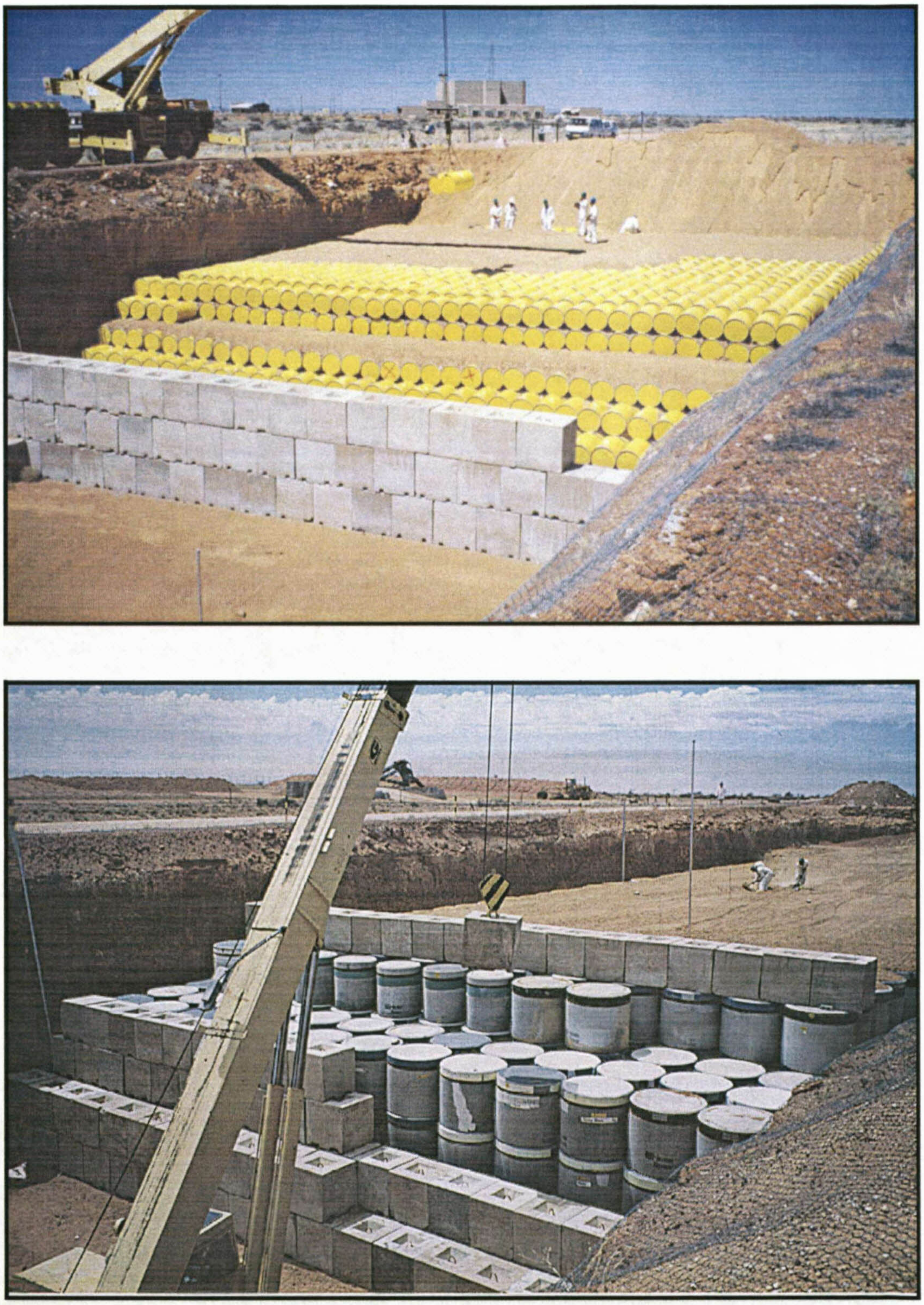


Figure 5–4 Near-surface earth trenches used for the disposal of low- and intermediate level waste at Vaalputs.

that report they motivate Thabana as a permanent storage area for active solid waste, given the geological and the ion-exchange properties of soils (Van der Westhuizen and Grobbelaar, 1967). Based on this information the types of waste that will be disposed are stated, as well as the method of disposal that is envisaged.

Although several studies have been performed in the vicinity of Thabana since 1968, the level of detail of the site and facilities are still very limiting compared to Vaalputs. This could possibly be attributed to the intention that the waste will be retrieved some time in future, i.e. Thabana was always considered as a *storage facility* (Van As and Basson, 1969). However, the viability of retrieving the waste from some of the repositories is questionable at this stage. Yet, the AEC has not obtained a license from the CNS to dispose radioactive waste at Thabana.

Key issues with regard to the Thabana facility is its present adequacy as a storage facility, the uncertainty regarding its total inventory and its future status, for example should it be upgraded to an approved disposal facility or rehabilitated and closed. In the meanwhile, the assumption will be made that the waste will not be retrieved in the near future.

5.3.2 Site Characteristics

As shown in Chapter 1, Thabana is situated at Pelindaba near Pretoria, the Headquarters of the AEC. The topography is rugged with a high relief and with slopes to the east and west. The mountainous terrain strikes east west with incised valleys that runs north south. Thabana itself is situated on a topographic high, with a potentiometric surface at 50 to 60 m below the collar height.

The general geology of Pelindaba consists of alternating shale and slate with interbedded ferrugineous quartzite layers of the Timeball Hill formation of the Pretoria Group. The slates are finely laminated with a heterogeneously developed cleavage (Courtnage *et al*, 1996). The quartzite layers have a variable grain size and are between a metre and two thick. These sediments dip to the north at an angle of between 23^0 and 34^0. The quartzite, which is more resistant to weathering forms prominent ridges and broken boulders. Two intrusive dykes—a dolerite dyke approximately 10 m wide and a syenite dyke 3 –5 m wide—is present just south of Thabana.

The average annual rainfall at Pelindaba is 621 mm with a minimum of 375 mm recorded in 1991/92 and a maximum of 1 196 mm recorded in 1995/96.

5.3.3 Waste Characteristics

The original specification for waste at Thabana made provision for the permanent storage of waste that contains low levels of α-, β-, and γ-rays that is sealed in plastic, or absorbed on vermiculite, (in the case of organic material) and compressed in drums. Provision were also made for solidified high-level waste (e.g. solidified concentrates from the evaporators), solid high-level waste (e.g. spent sources, irradiated samples) and 'absorbed high-level waste' (Niebuhr *et al.*, 1969).

As mentioned above, an inventory of what has actually been disposed at Thabana is one of the important uncertainties at this stage. The wastes include uranium (and its daughters) contaminated equipment, scrap, miscellaneous metal items and drums, plutonium mixed with cement and placed in metal drums, spent sources, medical waste, and baboon and pig carcasses used for medical purposes. These putrefiable carcasses, which contained only short-lived isotopes, such as ^{51}Cr, ^{22}Na, ^{57}Co, ^{59}Fe, ^{125}I, ^{95}Nb and ^{103}Ru, were mixed with lime and placed in drums.

5.3.4 Physical Description of The Repositories

Various kinds of repositories have been used for disposal at Thabana. The repositories used for the disposal of radioactive waste consist mainly of simple near-surface earth trenches and near-surface engineered boreholes. The level of information available on these repositories is very limiting compared to modern day facilities. It can only be assumed that at the time of implementation the level of information was thought to be sufficient. The trenches, with depths that vary between 3 and 8 m, contain a variety of wastes of which some were placed in drums, while others were simply packed loosely directly into the trench. Six of the trenches are filled and backfilled with excavated material and capped with imported clay. Two other trenches were in the process of being filled, when people purposefully intruded Trench 7 in search of a lost condenser containing 2 tonnes of depleted uranium in May 1996. No waste has since then been disposed at Thabana. As shown in Figure 5–5 the waste disposed in Trench 7 consists mainly of drums and contaminated scrap metal, which were then backfilled with soil.

The inventory of the trench is uncertain and the properties of the host medium, containment, backfill and cap material are largely unknown. The long-term strategy and performance of the trench is also unknown, as well as the rationale behind the physical design of the trench as a disposal concept. This trench is a typical example of an older disposal facility and an illustration of how radioactive waste should not be disposed.

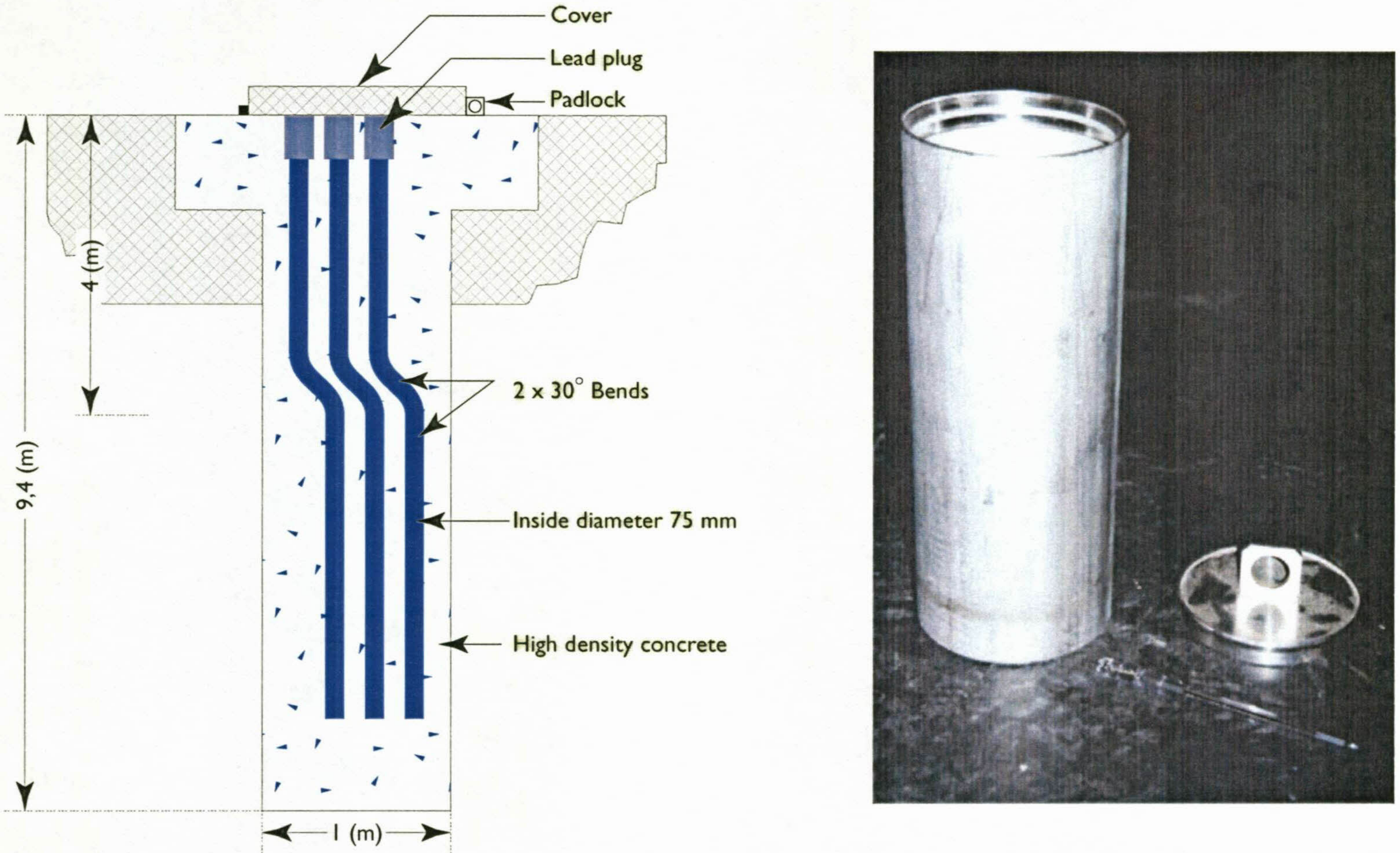
Figure 5–5 An excavated portion of Trench 7 at Thabana, which reflect a typical past practice near-surface disposal concept.

Another disposal system employed at Thabana is an engineered near-surface borehole facility, presented in Figure 5–6 (a). The facility, used for the disposal of highly active ^{60}Co pencil sources consists of a borehole approximately 1 m wide and 9.4 m deep. Three parallel stainless steel pipes, with inside diameters of 75 mm, were positioned in the borehole embedded in high-density concrete. There are two 30° bends in the pipes, approximately 4 m from the top to prevent a direct radiation pathway. The facility is capped with a concrete apron incorporating a hinged metal lid provided with a padlock. The sources (with an activity of 8.9×10^{13} Bq (in 1976) were conditioned in stainless steel containers, similar to the one shown in Figure 5–6 (b).

5.4 CONCEPT FOR THE DISPOSAL OF CANADA'S NUCLEAR FUEL WASTE

5.4.1 General

The selection of a site for the disposal of any high-level waste is a controversial subject with extensive financial implications. Although many countries have selection procedures in place, not one spent fuel element has been disposed in any facility in the world today. As mentioned before, some countries have developed underground research laboratories at potential sites, while others have developed 'engineered conceptual repositories'. These are repositories whose details, including their feasibility and potential environmental effects, have been planned and discussed in some detail, but not to the extent that construction can begin. One example of the latter facilities is the concept developed by Atomic Energy of Canada



(a) (b)

Figure 5–6 The engineered near-surface borehole facility (a) and stainless steel container used for the disposal of spent ^{60}Co sources at Thabana (b).

Ltd. (AECL) for the disposal of Canada's nuclear fuel waste (AECL, 1994*a*), the characteristics of which will now be discussed in more detail.

5.4.2 Waste Characteristics

In 1992, 15% of the electricity generated in Canada was produced using CANDU (CANada Deuterium Uranium) nuclear reactors, which are fuelled by uranium dioxide (UO_2). Three provincial electric utilities, Ontario Hydro, Hydro-Quebec and New Brunswick Power, own these reactors and the spent fuel removed from them. A limited amount of spent fuel, from prototype power reactors that have been shut down, is owned by AECL.

The UO_2 is formed into ceramic pellets that can withstand the high temperatures, pressures and radiation fields in a nuclear reactor. These pellets are placed inside a tube, called the fuel sheath, which is made of a zirconium alloy. Each end of the pellet-filled tube is sealed with a welded zirconium alloy plug to produce a fuel element. Up to 37 of these tubes is welded to zirconium alloy end plates to make a fuel bundle, which is about 100 mm in diameter and 500 mm long. It has a total mass of about 24 kg, of which about 19 kg is uranium. A schematic representation of the CANDU fuel bundle is presented in Figure 5–7. The ranges of radionuclides and activities present in the fuel bundle is typical of nuclear reactor waste.

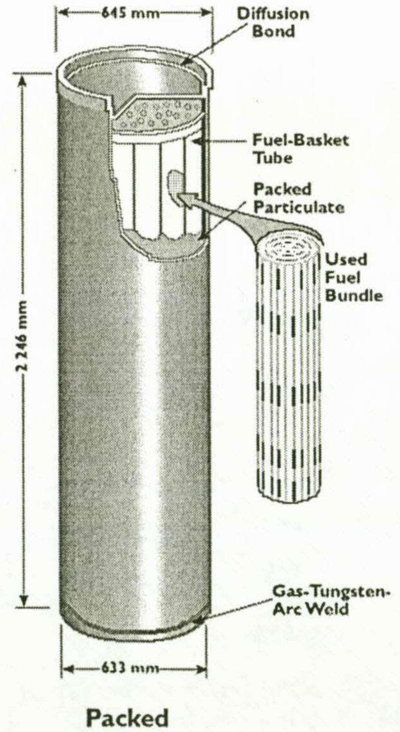


Figure 5–7 Schematic representation of the CANDU fuel bundle and container.

5.4.3 The Host Medium

A decision was taken in 1972 that geological disposal offers the best prospect for the disposal of Canada's nuclear fuel waste. However, in 1974 it was decided to direct the research mainly towards disposal in crystalline plutonic rock prevalent in the area of the Canadian Shield in Ontario, shown in Figure 5–8. AECL proposed that the waste be disposed in a vault, which would eventually be sealed, several hundred metres below the surface.

Plutonic rock is formed deep in the earth by crystallization of magma and/or by chemical alteration. It is often referred to as crystalline rock or intrusive igneous rock, of which the most common type is granite. This hard rock formation, which is similar to the basement rock of Vaalputs, will serve to protect the waste form, containers, and vault seals from natural disruptions and human intrusion. It should also be able to maintain conditions in the vault that will be favourable for the long-term isolation of the waste. However, the host medium will have to retard the movement of any radionuclides that will ultimately be released from the vault.

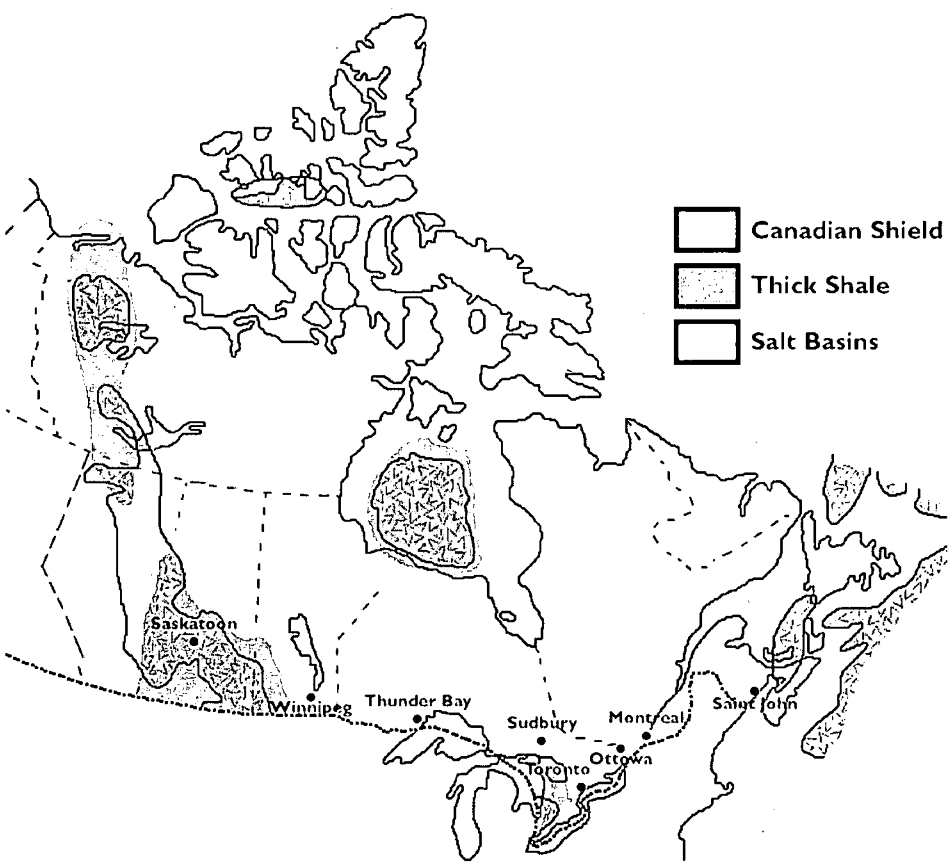


Figure 5–8 The plutonic rock in the Canadian Shield has characteristics considered to be technically favourable for the disposal of spent fuel in Canada (taken from AECL, 1994*b*).

5.4.4 Requirements for the Disposal Concept

The requirements of the disposal concept proposed by AECL are based on the results of a public involvement program. This program included consultation with special interest groups, discussions with focus groups, conducting public opinion surveys, and commissioning literature reviews and case studies on social issues. These requirements are clearly in line with the principles of radioactive waste management discussed in Chapter 1. That is, human health and the natural environment must be protected from radioactive waste and that the burden placed on future generations must be minimized, depending on the social and economic factors. The building of public confidence that started with the drafting of the requirements should continue in such a way that there is scope for public involvement during all stages of concept implementation. Finally, the disposal concept must be appropriate for Canada, that is, compatible with the geographical features and economic factors.

It was recognized from the beginning that these objectives could only be fulfilled if a suitable technology and expertise is developed and implemented, from the siting of the disposal site, to the closure of the disposal facility. This technology should

(a) not rely on long-term institutional controls as a necessary safety feature—that is, the disposal facility should be passively safe after closure.

(b) currently available or readily achievable.

(c) be adaptable to a wide range of physical conditions and social requirements and to potential changes in criteria, guidelines, and standards.

(d) include the ability to monitor the site from time to time.

(e) include the ability to retrieve the waste—the provisions needed to retrieve the waste, however, should not compromise the facility's passive safety.

5.4.5 Features of the Disposal Concept

A major feature of the proposed concept is that the spent nuclear fuel will be disposed in a geological setting, with multiple barriers to protect humans and the natural environment from both radioactive and chemically toxic contaminants in the waste. These barriers, shown in Figure 5–9, are the container, waste form, buffer, backfill, other vault seals and the geosphere. Institutional controls should therefore not be necessary to maintain safety in the long-term.

The expected waste form would be either spent CANDU fuel, or solidified HLW from reprocessing, if the spent fuel were reprocessed in the future. The form of the spent CANDU

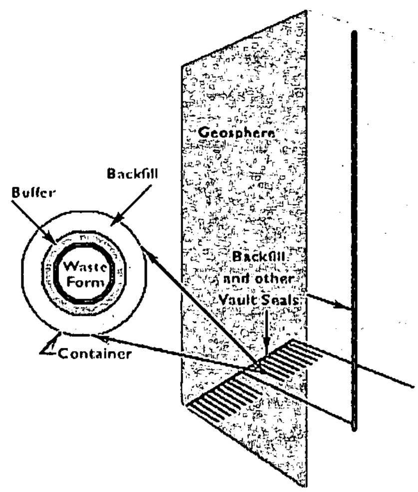


Figure 5–9 Schematic representation of the natural and engineered barriers that forms part of the geological disposal concept for the disposal of Canada's spent fuel (taken from AECL, 1994*b*).

fuel will reduce is solubility considerably and therefore contribute significantly to the retention of radioactive and chemically toxic contaminants under the expected disposal conditions. Nevertheless, it was decided that all waste, whether it is CANDU fuel in its current state, or liquid radioactive waste from reprocessing, would be sealed in a container to facilitate its handling and isolate it further from the surrounding environment.

The container, known as a packed-particulate container, is constructed from ASTM Grade 2 titanium (6.35 mm thick) to provide a design lifetime of 500 years, based on its corrosion resistance. The container, shown in Figure 5–7, is an enclosed cylindrical vessel of all-welded construction, with a maximum outside diameter of 645 mm and an overall height of 2 246 mm. Inside the container, a basket holds 72 spent CANDU fuel bundles in 4 vertical stacked arrays of 18 bundles. The basket is constructed of carbon steel tubes arranged in concentric circles. Glass beads are compacted around the fuel bundles inside the container to support the container wall and enable the container to withstand the external pressure that would be imposed on it in a disposal vault. This will ensure that the waste is completely isolated during the operation of the disposal facility and until there is a substantial decrease in the activity and heat output of the waste. (The activity of the radioactive waste will be more than 200 000 times less than when it came out of the reactor after 500 years.)

The waste containers will be emplaced in the disposal vault that will be excavated at a depth between 500 m to 1 000 m, below the surface, in the plutonic rock of the Canadian Shield. It consists of a network of horizontal tunnels and disposal rooms excavated deep in the rocks, with five vertical shafts extending from the surface to the tunnels, as shown in Figure

5–10. The shafts provide for ventilation and for transportation of container casks, materials, equipment, and personnel. The greater the depth, the greater the minimum possible transport distance from the disposal rooms to the surface, and lower the likelihood of any natural disruption or inadvertent human intrusion. However, the *in situ* temperature, stresses and cost of construction increase with depth. The vault itself consists of rooms or in boreholes drilled from the rooms.

To help achieve the overall objective of limiting the release of contaminants from the disposal vault, a number of different types of vault seals shown in Figure 5–11 would be needed. Each container in the disposal room would be surrounded by a buffer material, which would most likely contain clay. The buffer around the container has two purposes. The first is to limit the rate of corrosion of the container and the dissolution of the waste form (should groundwater seep into the container). The second is to retard the movement of any contaminants released from the waste form and the container. Each room would be sealed with backfill and other vault seals such as bulkheads, plugs, and grout made of clay-based or cement-based material. These seals would fill the space in the rooms, keep the buffers and containers securely in place, and retard the movement of any contaminants released from the container, waste form and the buffer. All tunnels shafts and exploration boreholes would ultimately be sealed in such a way that the disposal facility would be passively safe; that is, long-term safety would not depend on institutional controls.

5.5 THE BOSS CONCEPT FOR THE DISPOSAL OF SPENT SOURCES

5.5.1 General

As mentioned above, large numbers of spent sources from the medical and other technical professions exist in many countries, even countries that do not possess facilities related to the nuclear fuel cycle, that have to be disposed. This is particularly the case in Africa, South America and some members of the Russian Federation. Since these sources need to be handled separately from the other types of radioactive waste, mainly because of their activity to volume ratio, countries (even those with access to operational repositories) find it difficult to manage and dispose this waste. This has led to the use of boreholes as disposal units for these spent sources by some members of the Russian Federation (Van Blerk *et al.*, 1999) and in South Africa (See Section 5.3). However, the relatively shallow boreholes used by these countries are not suitable for the disposal of isotopes with long half-lives, such as ^{226}Ra and ^{241}Am—a defect that the BOSS disposal concept try to eliminate. Although the concept is still under development and evaluation, using the safety assessment approach discussed in this document, an initial post-closure safety assessment has shown

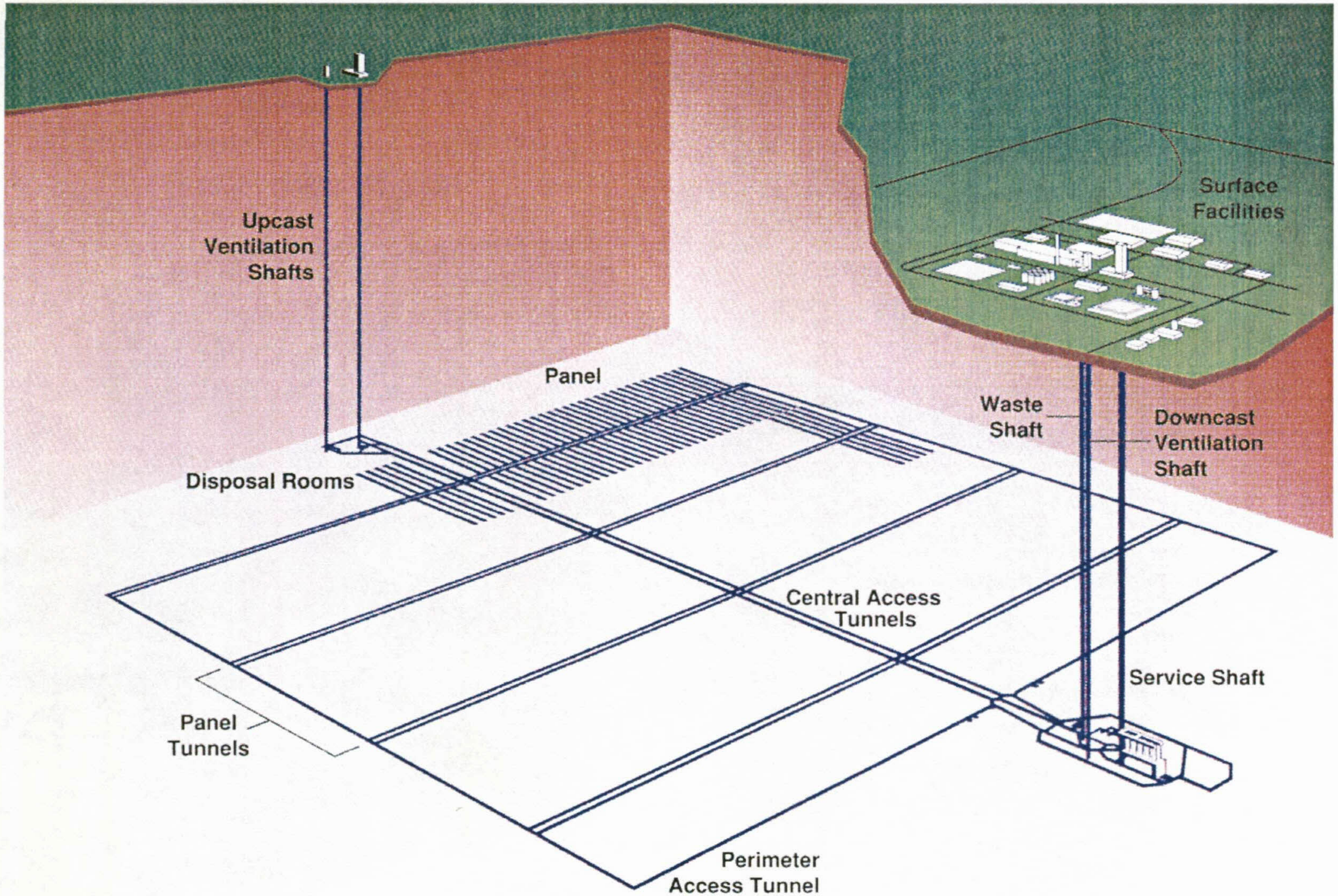


Figure 5–10 Cutaway view of the disposal vault for the geological disposal facility proposed for the disposal of Canada's spent fuel (AECL, 1994*b*).

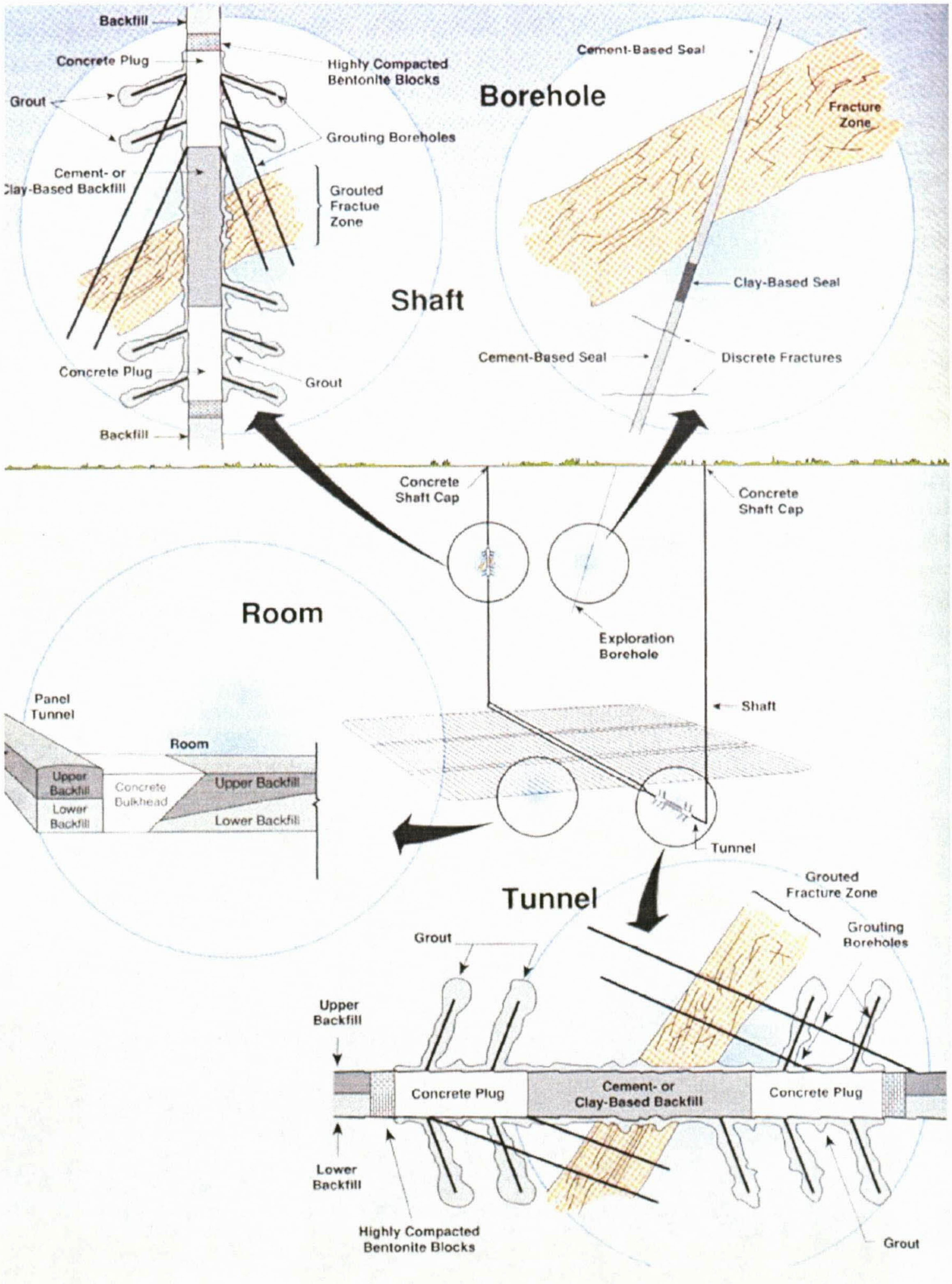


Figure 5–11 Schematic representation of the various vault seals used to limit the release of contaminants from the disposal vault of a geological disposal concept (AECL, 1994*b*).

that the concept is robust and viable (Kozak, *et al.*, 1999).

5.5.2 Requirements of The Disposal Concept

The BOSS disposal concept is intended to provide a solution specifically for the disposal of spent sources and therefore must take into consideration the size and number of the sources (i.e. the volume) that needs to be disposed. Several countries world-wide lack a nuclear infrastructure that can control sophisticated disposal facilities. What they need is a disposal concept that is technically feasible and economically viable to implement, taking the equipment at their disposal and the their financial situation into consideration. Nevertheless, it is important that the concept should comply with the safety objectives of radioactive waste management as discussed in Chapter 1. Particular attention should therefore be paid to the design of the repository and its operational requirements. For example, it would be advantageous if the waste could be emplaced routinely over the life-span of the repository, the need to actively maintain the site after closure could be minimized, and human intrusion (purposefully or inadvertently) could be difficult.

5.5.3 Waste Characteristics

The main aim of the BOSS disposal concept is to provide a solution for the disposal of both short- and long-life radionuclides, particularly isotopes such as ^{226}Ra and ^{241}Am. Table 5-2, lists a typical inventory of spent sources from a country for which the BOSS disposal concept is implied.

Table 5-2 A typical inventory of spent sources for which the BOSS disposal concept is implied. (All dimensions refer to the height and diameter of a cylinder.)

Isotope	No. of Sources	Activity per Source (GBq)	Total Activity (GBq)	Dimensions (mm)	Application
^{192}Ir	22	3.700E+03	8,140E+04	3x3	Gammagraphy
^{60}Co	2	3.700E+00	7,400E+00	3.2x3	Level Gauges
^{57}Co	4	1.850E–04	7,400E–04	–	Medicine
^{137}Cs	11	2.775E+00	3,053E+01	6x4	Gamma densitometers
	5	5.550E+01	2,775E+02	6x4	Well Logging
^{241}Am	560	1.850E–04	1,036E–01	60x5	Smoke Detectors
	9	5.550E–04	4,995E–03		
^{226}Ra	6	3.700E–03	2,220E–02	100x35	Calibration
	3	1.110E–04	3,330E-04	70x50	Teaching
^{238}Pu	1	3.700E+00	3,700E+00	20x50	Static Electricity Removal
$^{241}Am/Be$	5	1.850E+00	9,250E+00	10x10	Humidity gauge

5.5.4 Features of The Disposal Concept

The conceptual repository for the BOSS disposal concept consists of a standard 165 mm diameter borehole drilled to a depth of 100 m, as shown in Figure 5–12, with a 150 mm casing. However, wider and deeper (or shallower) boreholes can be drilled if required, depending on site-specific conditions and the sources to dispose. Van Blerk *et al.* (1999) discuss some guidelines for drilling the boreholes. The borehole and screen should be sealed off with a cement plug at the bottom to ensure that the disposal volume remains dry during the operational period. The disposal area can also be fenced in to limit access, and a temporary site office erected if necessary.

The disposal of the waste packages will be limited to the bottom 50 m of the borehole and the rest backfilled with concrete to close the repository and prevent human intrusion. Cement sludge will be poured over the waste package after its emplacement as backfill, illustrated in Figure 5–13.

Van Blerk *et al.* (1999) argued that a casing constructed from radiation resistant PVC might be the most suitable for the purpose, due to its small mass per unit length and potential shielding power (especially to neutrons). They also discuss the suitability of various drilling methods and their advantages and disadvantages under African conditions.

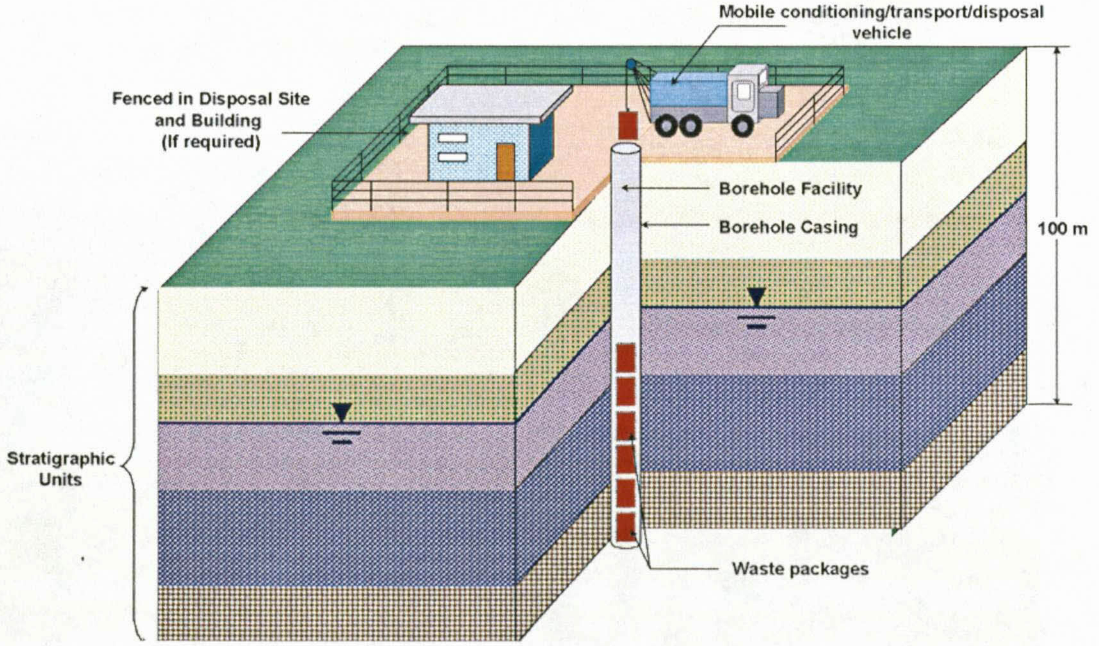


Figure 5–12 Schematic presentation of the repository area of the BOSS disposal concept.

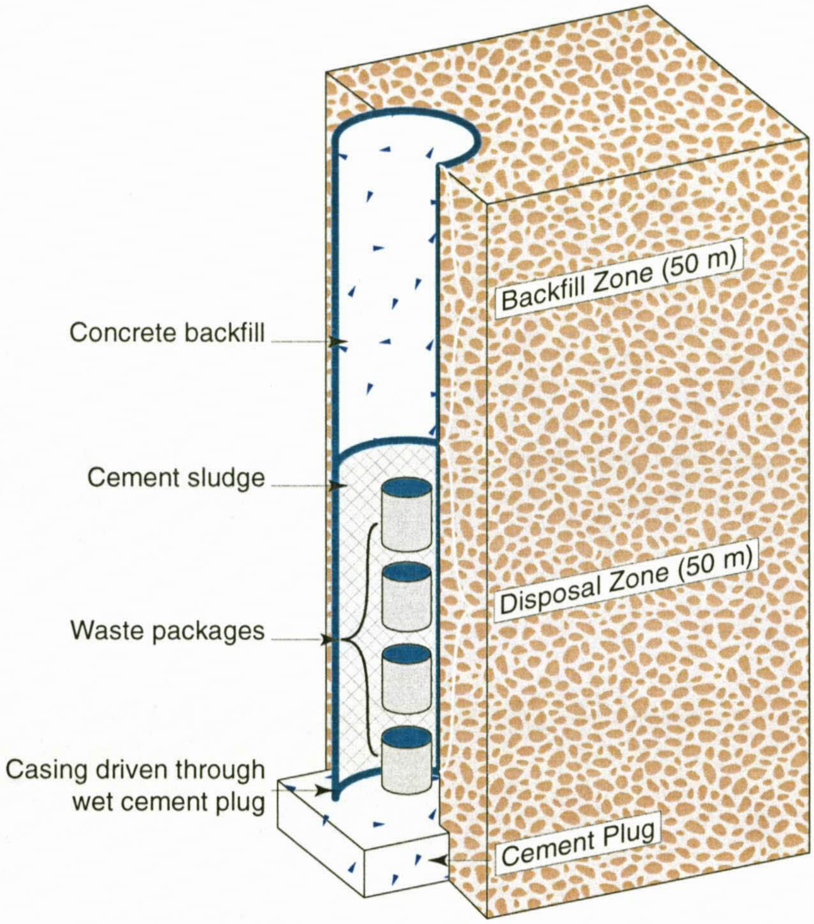


Figure 5–13 Graphical illustration of the *in situ* repository configuration for the BOSS disposal concept, showing the position of the waste packages and the use of different materials. (Not to scale.)

It follows from the preceding discussion that the disposal concept is, by definition, a near-surface disposal concept.

5.5.5 Design of the Disposal Waste Packages

The nature and construction of the waste packages can play a very significant role in the safety concept of the BOSS disposal concept, especially as an engineered barrier, but also to handle and transport the packages.

The approach followed in developing the BOSS disposal concept was to set a reference waste package design. There is little doubt that ^{226}Ra is the most demanding isotope to provide for in a disposal concept for spent sources. A waste package suitable for ^{226}Ra should therefore also be suitable for the other isotopes of importance, such as ^{60}Co, ^{137}Cs, and ^{241}Am.

5.5.6 Reference Waste Packages Design for ^{226}Ra

A large proportion of spent sources consists of radium needles. These needles are tubes of platinum, platinum-iridium, or (more rarely) gold, which contain soluble salts of radium (IAEA, 1996). Since the needles often tend to leak (Al-Mughrabi, 1998), the IAEA (1996) recommended that they be encapsulated, using the techniques described by the IAEA (1996) and Al-Mughrabi (1998). The capsules recommended for this purpose are tubes with an outside diameter of 21.23 mm, length of 110 mm and wall thickness of 2.77 mm, constructed from Type 304 stainless steel (Al-Mughrabi, 1998). Each of the capsules will be placed in a cementatory waste form within another Type 304 stainless steel container with outside diameter 114.3 mm, length 250 mm and wall thickness of 3.04 mm, as illustrated in Figure 5–14.

Once the waste package is closed and without any leaks, it is ready for disposal, as depicted in Figure 5–15. This means that each waste package will be separated form its neighbour by a cement layer 750 mm in the borehole. A single package and its backfill will therefore occupy a length of 1 m in the boreholes, so that 50 waste packages can be disposed of in a borehole with the standard dimensions proposed for the BOSS disposal concept. However, the exact number will be determined by the inventory and the site characteristics, as derived from the waste acceptance criteria for the disposal facility.

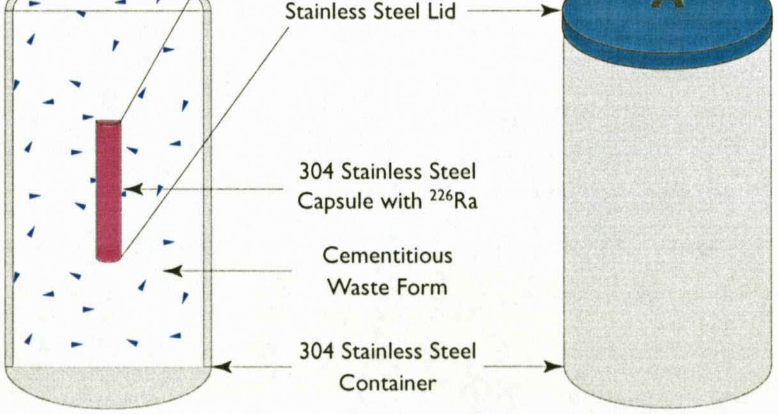


Figure 5–14 Various stages in the preparation of waste packages proposed for the disposal of spent sources in the BOSS disposal concept.

There are a number of reasons for proposing stainless steel as the container material and cement as waste form for the reference design in the BOSS disposal concept. Stainless steel is, for example more resistant to corrosion than carbon steel and passivated by high-pH conditions. The general corrosion and pitting corrosion rates are therefore expected to be low, so that the containers do not need a protective coating. Crack corrosion cannot be excluded, however, particularly in groundwater with a high chloride content. Agg *et al.* (1993) estimate that stainless steel will in general corrode at a rate of 0.3–1.0 µm year^{-1} under the alkalic and anaerobic conditions of a repository (Kozak *et al*, 1999).

The cement on the other hand will act as a barrier, first between the container and aggressive chemicals (primarily chloride) and then the capsule and chemicals, thereby reducing the initiation of corrosion on the capsule. The cement will also provide a physical barrier

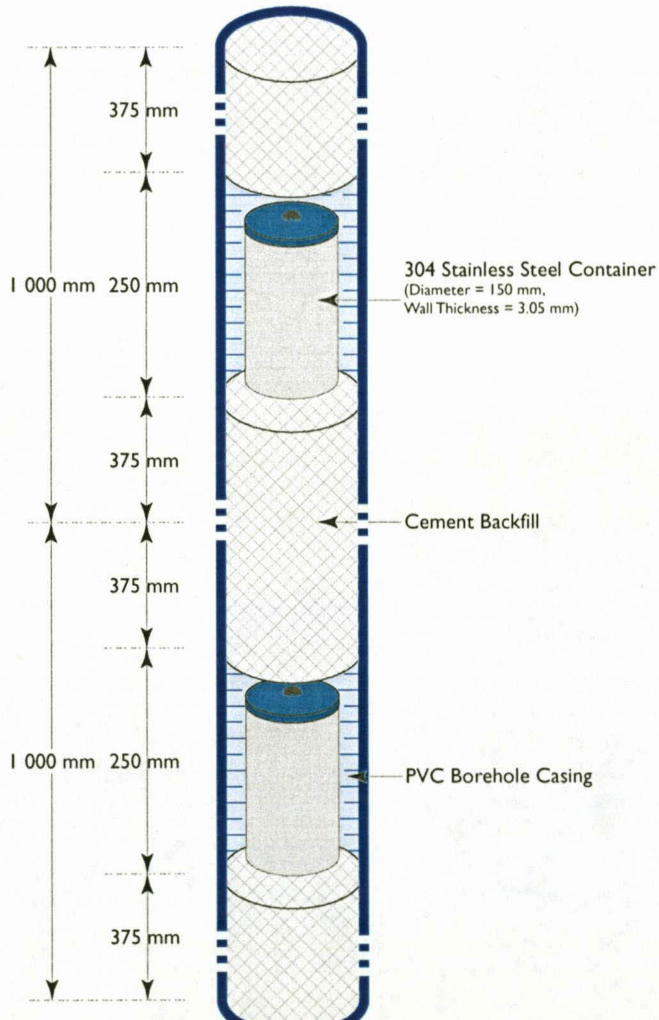


Figure 5–15 Schematic depiction of the *in situ* placement of waste packages with the BOSS disposal concept.

through which leached radium must pass before being released into the geosphere. At the same time, it may also act as a chemical buffer, thereby limiting the release of radium intrinsically (Kozak *et al*, 1999).

CHAPTER 6

BASIC PRINCIPLES OF A POST-CLOSURE SAFETY ASSESSMENT

6.1 INTRODUCTION

A post-closure safety assessment was defined in Section 1.2 as an attempt to quantify the exposure or risk posed by a radioactive waste disposal site to future generations of humanity and their environment. Intrinsically, to do this one can say that the observations needed for the safety assessment of a site should be carried out for the life span of the proposed disposal facility. However, this is neither physically possible nor desirable. The only viable approach to perform a complete radiological safety assessment is to try to obtain as much observational data as possible, on a limited time scale, and then simulate the future behaviour of the disposal system through what is known as a *model.* Although there are other methods, most models are developed on a computer, because of the complexity of the system.

The term model is, unfortunately, so often misused that it is not always clear what is meant by it. This applies in particular to the literature on groundwater models (De Marsily *et al.*, 1992; Konikow and Bredehoeft, 1992). There is reason to believe that the same can be said of the term model, as used in safety assessments. A more precise definition of the term model, as used in this thesis, is therefore given in Section 6.2 to avoid any confusion before a broader perspective of the basic principles of such an assessment is given.

The worldwide interest in the safety assessment of radioactive disposal systems has led to numerous proposals how such an assessment should be implemented. The approach proposed here, a flow chart of which is shown in Figure 6-1, is based on the approach of the Co-ordinated Research Programme (CRP) of the IAEA on the *'Improvement of Safety Assessment Methodologies for Near-Surface Radioactive Waste Disposal Facilities (ISAM)'* (IAEA, 1997*a*). Although this programme focuses on near-surface disposal facilities, the principles are also valid for geological disposal, as shown by the discussion of the basic framework of the approach in Section 6.6. However, it may be useful to discuss a few properties of a post-closure safety assessment first. The discussion therefore begins with the nature of a post-closure assessment in Section 6.3. This is followed by a discussion of the purpose of such an assessment in Section 6.4, and a few characteristics that are unique to a post-closure assessment in Section 6.5.

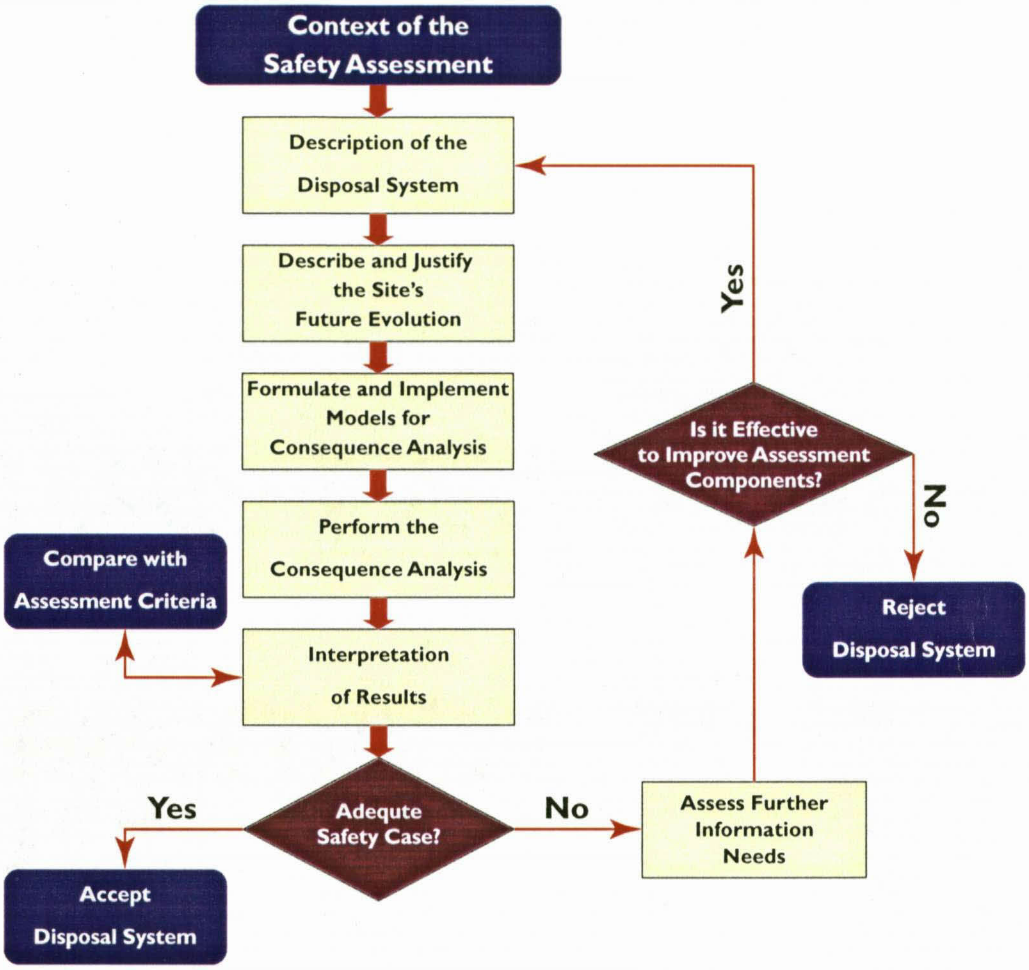


Figure 6–1 Flow diagram of the ISAM approach for the post-closure safety assessment of near-surface radioactive waste disposal systems.

6.2 MODELS IN SAFETY ASSESSMENTS

6.2.1 General

Botha (1994) ascribes the confusion in the interpretation of the word model in groundwater literature, to a desire to differentiate between what is known in the exact sciences as a *theory*, and the *application of the theory* to a real-world realization of the phenomenon for which the theory was developed. The simplest approach to try to eliminate the confusion of the word *model*, as used in the groundwater literature and safety assessments, is probably to look at how scientists in the exact sciences, particularly physics, perceive a theory. The theories of groundwater motion and mass transport that play a very important role in radioactive safety assessments will be used as examples in the discussion that follows. However, it must be remembered that they are not the only theories used in such an assessment.

6.2.2 Development of A Physical Theory

A physical theory is in essence nothing more than an attempt by scientists to understand and control the natural environment, using abstract, philosophical reasoning to explain a natural phenomenon, that is a phenomenon *observable through the human senses.* The first step in the development of a theory for a physical phenomenon is then also usually taken when the curiosity of someone is aroused by a particular *observation* of a phenomenon, who then tries to *explain* the behaviour of the phenomenon.

The ancient Mesopotamian, Egyptian and Greek civilizations already recognize that a physical phenomenon can be related to three basic measurable quantities—*space*, *time* and *matter.* However, it was left to the English scientist, Sir Isaac Newton, to show how a combination of these measured quantities, and the *ability of humankind to reason abstractly*, can be used to what Botha (1994) calls the *conceptualization of a natural phenomenon.* The result of this *conceptualization is what is* commonly known as a *theory* in the exact sciences.

Many attempts have been made in the past to try to base a theory just on the ability of humankind to reason abstractly, especially in subjects (e.g., groundwater) where it is difficult to make observations. Such an approach can be an interesting mathematical exercise, but is without real physical substance. A theory always consists of *two* inseparable components: *non-trivial observations on the phenomenon*, and the ability of humankind to reason abstractly.

It is quite common in developing a theory for a phenomenon, to come across quantities or *interactions* that play a basic role in the theory, but that cannot be measured directly. In such cases, the interaction is often related to more directly observable and measurable quantities, henceforth referred to as *observables*, through a number of so-called *relational or constitutive parameters.*

An important property of a theory, which is not generally recognized, is that a theory always contains at least one theoretical principle, or *postulate*, deduced from particular observational facts. This postulate, often referred to as a *law of nature*, or simply *law*, can be rejected or modified, but *never be verified experimentally or observationally.* A well-known example of such a theoretical principle in the study of groundwater motion is *Darcy's law* (Botha, 1994)

$$\mathbf{q}(\mathbf{x},t) = -\mathbf{K}\nabla\phi(\mathbf{x},t) \qquad (6.1)$$

where the hydraulic conductivity tensor, K, relates the gradient of the piezometric head, f, to the mass flux, **q**, per unit time over a unit area. Darcy (1856) derived this law from observations on the flow of water through a sand column.

Another example of a law that is particularly important in the assessment of a disposal system is the *law of mass conservation*. This law first enunciated by the father of modern chemistry, Antoine Laurent Lavoisier, holds that matter cannot be created nor destroyed. Lavoisier derived the law from numerous experiments with gasses and other elements.

6.2.3 The Mathematical Model

Although a law is crucial to a theory, it is often not suitable for a detailed study of the general behaviour of a phenomenon, because some of its observables cannot be measured easily. Darcy's law in Equation (6.1), for example, relates three quantities: the discharge rate **q**(**x**, *t*), the piezometric head gradient ∇f(**x**, *t*) and the hydraulic conductivity K(**x**, *t*) at a specific point (**x**, *t*) in space and time to one another. The law can therefore be used to determine the value of any one of the three quantities, by measuring the other two. However, it is not easy to measure the Darcy velocity in the field, with the result that it will be difficult to use this law for the study of the piezometric pressure head, as a function of space and time. What is commonly done in such circumstances, is to supplement the law with additional mathematical information to develop a set of equations, known as the *governing equation* of the observable.

The governing equation for groundwater motion is usually derived by applying the law of mass conservation. This yields the so-called continuity equation (Bear, 1972)

$$\mathrm{D}_t[\rho\theta(x,t)] + \nabla\cdot[\rho\mathbf{q}(x,t)] = 0 \tag{6.2}$$

where θ denotes the volumetric moisture content, ρ the density of the medium and D_t the partial derivative with respect to time. Substitution of Equation (6.1) into this equation yields

$$\rho S_0\mathrm{D}_t[\phi(\mathbf{x},t) = \nabla\cdot[\rho\mathbf{K}\nabla\phi(\mathbf{x},t)] + \rho f(\mathbf{x},t)$$

or if it is assumed that the density of the fluid remains constant

$$S_0\mathrm{D}_t[\phi(\mathbf{x},t)] = \nabla\cdot[\mathbf{K}\nabla\phi(\mathbf{x},t)] + f(\mathbf{x},t) \tag{6.3}$$

where S_0 is the storativity of the medium and *f*(**x**, *t*) the strength of any sources or sinks that may contribute to the flow. This differential equation can be solved for ϕ(**x**, *t*), once the three constitutive parameters S_0, K and *f*(**x**, *t*) has been determined. It is in this sense that Equation (6.3) is regarded as the governing equation of density indepedent groundwater motion.

The governing equation for density independent mass transport in groundwater can be derived using similar lines of reasoning. Application of the law of mass conservation, Fick's law for molecular diffusion and Darcy's law yield in this case the hydrodynamic dispersion equation (Botha, 1996)

$$[\theta D_t c + \rho_b D_t s] + \mathbf{q}\cdot\nabla c = \nabla\cdot[\theta \mathbf{D}\nabla c] - \lambda(\theta c + \rho_b s) + (c_0 - c)f(\mathbf{x},t) \qquad (6.4)$$

where c is the volumetric mass concentration of the solute, s the mass fraction of dissolved solids absorbed by the matrix, r_b the dry bulk density of the matrix (including the adsorbed solids), l the radioactive decay constant, D the *dispersion coefficient*, and c_0 the initial concentration of sources or sinks with strength $f(\mathbf{x}, t)$, as in Equation (6.3).

The dispersion coefficient, which is a second rank tensor similar to K, is usually expressed in the form

$$\mathbf{D} = \mathbf{D}_h + \mathbf{D}_m \qquad (6.5)$$

where $\mathbf{D}_m$ is known as the molecular diffusion tensor and $\mathbf{D}_h$ as the *coefficient of mechanical dispersion, or hydrodynamic dispersion tensor* (Botha, 1996). The *ij*–th component of this coefficient, which arises from the interaction between the dissolved solids and water, can be expressed as (Bear, 1979)

$$D^h_{ij} = \alpha_T \overline{v}(\mathbf{x},t)\delta_{ij} + (\alpha_L - \alpha_T)\frac{v_i v_j}{\overline{v}(\mathbf{x},t)} \qquad (6.6)$$

where α_L and α_T are two parameters, known as the *longitudinal* and *transverse dispersivities*, with dimensions [L], d_{ij} the Kronecker delta

$$\delta_{ij} = \begin{cases} 1 & (i = j) \\ 0 & (i \neq j) \end{cases}$$

and v_i the i-th component of the seepage velocity, $\mathbf{v}(\mathbf{x}, t)$, defined by the equation

$$\mathbf{v}(\mathbf{x},t) = \frac{\mathbf{q}(\mathbf{x},t)}{\theta(\mathbf{x},t)}$$

where $\mathbf{q}(\mathbf{x},t)$ and $\theta(\mathbf{x},t)$ are defined in Equations (6.1) and (6.2).

As shown by the examples above, governing equations arise from a combination of the laws that govern the behaviour of a phenomenon. Nowhere in the derivations of the governing equations were any particular realizations of groundwater flow or contamination used. The governing equation of a physical phenomenon therefore *does not depend on a specific realization of the phenomenon.* It therefore seems logical to refer to the governing equation as the *mathematical model* for the phenomenon (Botha, 1996).

6.2.4 The Conceptual Model

The governing equations of most physical phenomenon are expressed in terms of partial differential equations, such as Equations (6.3) and (6.4). As is well-known, a partial differential equation can only be solved if all *the constitutive parameters are known over the full domain* for which the differential equation is defined and appropriate boundary and initial conditions are prescribed along the boundary, and within the domain, respectively. To illustrate these concepts more concretely, consider the simple two-dimensional parabolic partial differential equation

$$D_t u(x,y,t) = D_x^2 u(x,y,t) + D_y^2 u(x,y,t) \tag{6.7}$$

defined over the rectangular domain, Ω, shown in Figure 6-2. The formal integration of this equation will yield a solution in the form of an equation with five unknown parameters, two from each of the double integration over x and y respectively, and one from the integration over t. In practical applications these parameters are usually determined by *prescribing* values of $u(\mathbf{x}, t)$ on the four sides of the boundary ∂Ω of Ω, in Figure 6-2, hence the name boundary conditions. The fifth parameter, which arises from the integration over time, can be similarly determined if one equates the solution with a *known value* of $u(x, y, t)$ at a time say $t = t_0$, hence the name initial condition.

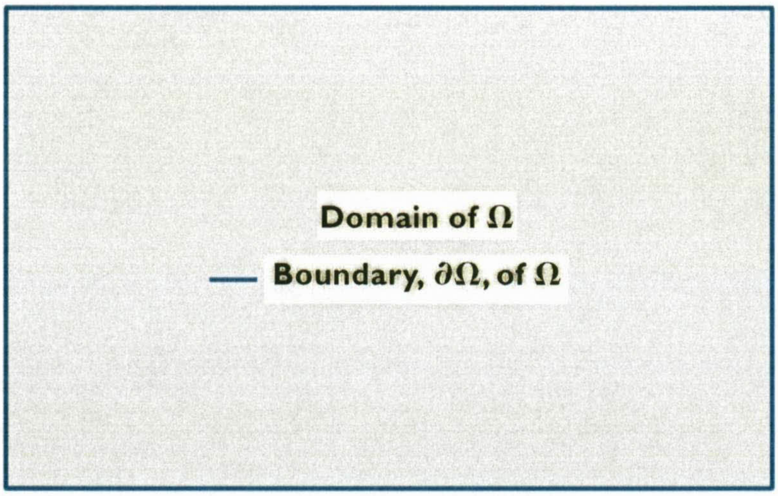


Figure 6–2 Schematic representation of the domain, Ω, and its boundary, ∂Ω, for the parabolic partial differential equation in Equation (6.7).

Boundary and initial conditions will clearly vary from one domain to another and can *only be derived once a domain is prescribed* for the governing equation. The same applies to the constitutive parameters—the hydraulic conductivity and storativity for an aquifer in fractured granite will clearly differ from that of a coastal aquifer in unconsolidated sands. Since it is not always possible to determine the boundaries of a phenomenon and its associated constitutive parameters uniquely in the environmental sciences, different investigators may assign different values to these parameters, as is often the case in practise. This observation led Botha (1996) to introduce the term *conceptual model* to describe the governing equation, or set of equations, with its associated constitutive parameters, boundary and initial (if required) conditions, for a specific realization of a phenomenon.

6.2.5 Analytical and Numerical Models

A conceptual model is not of much importance in practical applications, unless one can solve its governing equation explicitly, subject to the prescribed boundary and initial conditions. Although a large number of the governing equations of practical interest can be solved analytically, others can only be solved numerically. This led Botha (1994) to introduce the terms *analytical model* and *numerical model.* In his original formulation Botha uses the term analytical model to describe the solution of the governing equation in a conceptual model, when obtained with analytical methods, and numerical model, when the solution is obtained with numerical methods. However, recent developments in algebraic computer codes, such as Mathematica (Wolfram, 1996), blur this distinction somewhat.

Analytical models have one particular attractive property—they are always exact. Unfortunately, there are not many differential equations for which analytical solutions are known, particularly for phenomena with heterogeneous constitutive parameters and irregular boundaries. Moreover, many of the analytical models are expressed in terms of complex mathematical functions, which are difficult to compute. Since numerical models can handle such situations with ease, they soon replaced analytical models in many branches of science and engineering after the introduction of the electronic computer in 1945.

The advantage of numerical models, however, comes at a price in that they are not exact. Great care must therefore be exercised to ensure that the models are accurate. Numerical models must consequently often be restricted in space and time, unless one has access to large-scale computing facilities, which is not the case with analytical models. For example, Botha (1998) uses an analytical model to simulate the transport of radionuclides in the geosphere of tailings dams for periods up to 60 million years without any difficulties. Although this period is far beyond that required by the licensing authority, the simulation gave a considerably better insight into mass transport through the geospheres of the dams, than could have been derived from the available data, or a numerical model based on the data. It may therefore be worthwhile to give more attention to analytical models in the safety assessment of radioactive waste disposal systems, or even ordinary waste disposal systems, than is done at present.

6.3 THE NATURE OF A POST-CLOSURE SAFETY ASSESSMENT

A post-closure safety assessment is often viewed as a scientific exercise, carried out by scientists sitting in front of a computer, trying to predict the future, mainly because safety assessments are concerned with the long-term performance of the disposal system. Others see a post-closure safety assessment as a scientific exercise in which every physical aspect of the disposal system need to be analysed and included in the analysis to predict the *actual* behaviour of the disposal system over thousands of years. Another view is that it is simply a scientific exercise to demonstrate that a site meets regulatory safety objectives. These different opinions suggests that it may be worthwhile to study the carefully selected phrases in the definition of a safety assessment in Section 1.2 in greater detail, since they have specific implications for safety assessments.

The term *iterative process* in the definition simply means that one must expect that a safety assessment will have to be repeated two or more consecutive times. The advantage of such an approach is that it allows one to use information from the previous assessment to refine the design of the system and the collection of additional data. It also reduces the tendency that the assessment will focus on one component at the expense of others.

The term *site-specific prospective evaluations* emphasize the fact that the assessment should include data from the actual disposal system assessed. Not with the intent to predict its actual behaviour in the future, but rather to understand the behaviour of the system better and to reflect the importance of specific components with respect to the compliance criteria.

The safety assessment of a radioactive disposal system is not an exact procedure. The term *reasonable assurance* in the definition of a safety assessment emphasizes this inexact nature of the procedure. What one really wants to achieve in such an assessment is to reach defensible decisions on the extent to which the disposal system may comply with the regulatory criteria. A safety assessment is therefore more a decision tool to determine the conditions for which reasonable assurance of compliance with safety objectives can be provided than a method to predict the actual behaviour of a disposal system into the future. The results will therefore be largely a function of the data, design and assumptions used in the analysis. Changes in any one of these conditions can change the conclusions of the assessment (Kennedy, 1997).

6.4 PURPOSE OF A POST-CLOSURE SAFETY ASSESSMENT

The discussion in the previous section raises the question: *What is the purpose of a safety* *assessment if it cannot supply exact answers?* The best way to answer this question is to again look at the main objective of the management of radioactive waste in Chapter 1

> *'... to deal with radioactive waste in a manner that protects human health and the environment now and in the future without imposing undue burdens on future generations.'*

It is important to note that this objective does not say to *shield* the present and future generations *completely* from the effects of radioactive radiation, but to *protect* them from the effects of the radiation without imposing *undue burdens* on them. To achieve this, the IAEA (1989) introduced three other principles, closely related to the fundamental principles for the management of radioactive in Table 1–1. The first of these principles—*effects in the future*— states that:

> *'The degree of waste isolation provided by a disposal facility shall be such that there are no predictable future risks to human health or effects on the environment that would be unacceptable today.'*

A simple answer to the question above would thus simply be to quote this principle. However this raises another question: *how does one know that the risks acceptable today will not be detrimental to generations in the far future?* On the other hand, the possibility also exists that future generations may not be as susceptible to radioactive radiation as the present generation. Radioactivity is a natural phenomenon to which all people on earth are exposed daily and will be exposed to in the future. It is consequently simply impossible to shield future generations completely from the effects of radioactive radiation. The real problem today is therefore not radioactivity *per se*, but rather the concentration of radionuclides in hazardous sources by the present generation. The application of the previous principle will therefore not discriminate against future generations. Indeed, the applications of radioactivity is so beneficial to people that future generations may again turn to it, even if all radioactive wastes are removed from the earth today.

The main purpose of a disposal system is to protect future generations and the environment they are living in, at times, which cannot be controlled today. It is therefore necessary to determine how a disposal facility, which was designed, constructed and implemented today, will affect future generations. The last two of the three IAEA principles referred to above addresses this point.

The first of these principles—*independence of safety from institutional control*—says:

> *"The safety of a disposal facility in the post-closure period shall not rely on active*

monitoring, surveillance, or other institutional controls or remedial actions after the time when the control of the repository is relinquished."

This principle essentially means that it is the responsibility of the present generation to develop the necessary technology and infrastructure to ensure that future generations will not be exposed to a health risk if the site is not controlled institutionally. The site should therefore be safe enough to ensure that people will not be exposed to a health risk, even if the site is not monitored, or future generations are not aware of its existence.

The last of the three principles—*burden on future generation*s

"The burden on future generations shall be minimized by safely disposing of radioactive wastes at an appropriate time, technical, social and economic factors being taken into account"

is based on the ethical consideration that the generation who receives the benefits of a practice should also bear the responsibility to manage the resulting waste (IAEA, 1995*a*). In simple terms, it means that today's generation should clean up there own waste and not burdens future generations with it.

The implementation of a disposal system, unfortunately, does not depend only on the scientific and technical knowledge, but also on economic and social factors. One sociological factor—the unwillingness of people to have a radioactive disposal site in my backyard (Easterling and Kunreuther, 1995)—is, for example, responsible for the fact that the development of new disposal facilities has been postpone world-wide. It may therefore be very difficult for the present generation to fulfil the previous principle.

The application of the previous principles forces a constraint on the management of radioactive waste not imposed on that of other hazardous waste, although some of these wastes may pose a similar (perhaps even greater) risk to future generations. The management of radioactive waste therefore differs completely from any other scientific or technical activity (Kozak, 1998).

6.5 CHARACTERISTICS OF THE POST-CLOSURE SAFETY ASSESSMENT

It follows from the previous discussion that the post-closure safety assessment of a radioactive disposal system will have some unique characteristics, which will be briefly discussed, before describing the methodology itself.

As defined in Section 1.2, a post-closure safety assessment is concerned with the distant future, and not the present or even near present time. It may therefore have to cover a considerable period, as illustrated by the fact that high-level waste disposed today may affect the next 700 generations (Easterling and Kunreuther, 1995). Since the demographic features of future generations and the environment are unknown, a post-closure safety assessment can be interpreted as being more concerned with the protection of hypothetical persons and a hypothetical environment than real people and a real environment. There is therefore no guarantee that the principles valid today, such as the radiological protection standards of the ICRP, will still be valid at the times covered in a post-closure safety assessment. This poses a difficult problem for the post-closure safety assessment of radioactive waste disposal sites, which is commonly based on what may be called the *historical assumption—the past is an accurate reflection of the future.* However, the geological history of the earth, over the past few million of years, and the palæontologic record of *Homo sapiens* (modern humans), over the ±100 000 years that they exist, indicate that this is not such a dubious assumption as one may think at first. The real problem of a safety assessment therefore does not seem to be so much the historical assumption, than the approach used to perform the assessment.

Many interest groups view the post-closure safety assessment of a radioactive waste disposal site as a pure scientific problem that can be solved by the same scientific principles applicable to a pre-closure assessment. However, as pointed out in Section 1.2, this is not true. What is needed for a post-closure assessment, is what one can call a *compliance approach* rather than the *predictive approach* of a pre-closure assessment (Kozak, 1997). In other words, one tries to ensure that the disposal site will satisfy the regulatory criteria at all times, now and in the future, but not to predict its future. This can often be accomplished with less effort than that needed by the predictive approach, especially if a pre-closure assessment of the site has been conducted or is conducted before or during its operational life.

6.6 FRAMEWORK FOR THE POST-CLOSURE SAFETY ASSESSMENT

6.6.1 Safety Assessment Context

The main purpose of the assessment context is to define the scope and content of the post-closure safety assessment that is to be performed. Questions that the investigator could ask during this phase are for example: what are you trying to assess and why are you trying to assess it? The main effort of the exercise must, however, be to establish a set of higher level assumptions and constraints that will reflect the regulatory framework, purpose and focus of the safety assessment, what to include or exclude from the assessment and the justification for the choices.

As mentioned earlier, the main aim of a post-closure safety assessment is to ensure that waste management decisions made today do not affect the post-closure safety of the disposal system adversely. Special attention should therefore be paid to waste management aspects, such as the selection and design of a suitable disposal system, site selection and characterization, waste acceptance criteria, data collection and remedial actions.

Each regulatory authority will usually impose its own regulatory requirements and principles on a post-closure safety assessment that must be taken into account in the assessment context. (See Environmental Agency *et al.* (1997), for an example.) These may include aspects such as the principles that should apply to protect the public, regulatory requirements (e.g., protection criteria) and more technical requirements such as institutional control and time frames. In addition, a summarized description of the site, waste and repository characteristics can be included, because it contains information that may influence the assumptions of the assessment and that should be documented in the assessment context.

Institutional control can be described as the control of a disposal site by an authority or institution designated under the laws of a country or state (IAEA, 1993*a*). The control (which will be an important factor in the design of the disposal facility) may be exercised either actively (monitoring, surveillance, remedial work) or passively (land use and access control). Special attention should be paid to the period(s) of institutional control; because it defines the extent to which humanity can ensure that the waste is safely confined and prevent intrusion into the repository. The institutional control period is normally culture specific, and may depend on the history of the country. Control periods typically vary from 100 – 300 years (IAEA, 1997*a*).

The regulatory authority usually sets the time frame that should be used in the assessment. Since this frame can have a major influence on the assessment results, it may be advantageous to consider other options in the assessment context. One approach would be to extend the assessment over the full time that the activity of the waste is above the regulatory criteria. Although the activities of all radionuclides decline exponentially with time, the half-lives of some of them are so long (~ 10^6 years) that the assessment will have to be continued almost indefinitely, if this criteria is used to establish a time frame. Another approach is to simply define a cut-off time of say hundreds or thousands of years. The problem with this approach is that the peak activities at large distances from the near-field will occur much later than at the repository. Regulators are consequently reluctant to specify a cut-off time. The most used approach is therefore to perform a post-closure safety assessment until one can say with reasonable confidence that the peak activities have occurred at all points of the site's geosphere.

The radiological protection criteria—the criteria used to decide whether the proposed disposal system is acceptable or not—is usually set by the regulatory authority. There are essentially five indicators that can be used for this purpose. These indicators and their associated safety criteria are illustrated in Figure 6-3. Which indicator to use depends on the purpose of the assessment, which also indicates the end entity of the assessment (e.g., a flux, concentration, and dose). For this reason, they are also known as *assessment endpoints* or *calculational endpoints.*

As pointed out in Section 6.2 a model, more specifically a conceptual model is the only means with which the behaviour of a disposal system can be studied in some detail over the time frames of a post-closure safety assessment. Since none of these models is able to predict the future behaviour of the disposal system with certainty, the uncertainty in the post-closure safety criteria will increase as one moves from the radioactive waste to its impact on people's health in Figure 6-3. Despite this uncertainty, the individual dose and risk indicators remain the most widely used assessment endpoints in post-closure safety assessments, although the waste activity or concentration is also used sometimes. This is especially the case when the assessment is aimed at the definition of site-specific waste acceptance criteria (IAEA, 1999).

The individual dose and risk indicators are measures of the extent that the disposal system will affect the health of individuals near the facility. To determine these indicators all path-

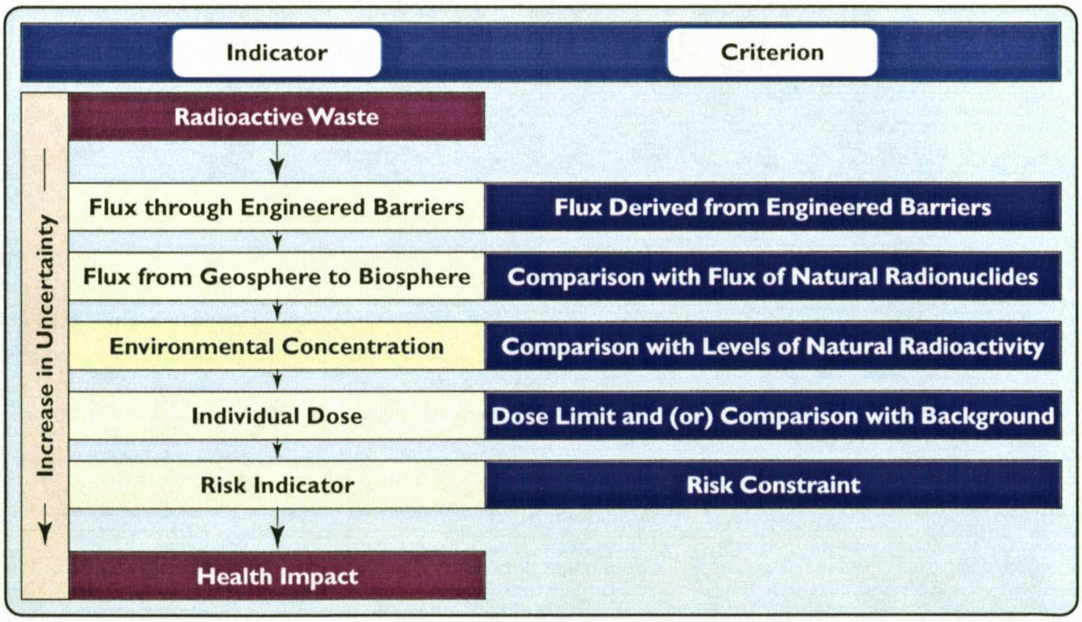


Figure 6–3 The safety indicators used in the post-closure safety assessment of a radioactive waste disposal site and their associated safety criteria (IAEA, 1997*b*).

ways of radioactive material or radiation from the disposal system to people must be considered. To determine a suitable criterion is another matter though. The criterion in use today (a dose limit of 1 $mSv \cdot a^{-1}$) was established by the ICRP (1990) for *real* doses to *real* people coming from *real* practices and not for future generations. However, if one accepts the view that the fourth fundamental principle for the management of radioactive waste does not discriminate against future generations, a similar criterion can be used here. It is consequently assumed in post-closure safety assessments that the projected dose to future members of the public, arising from events likely to occur at the disposal facility, shall not exceed an appropriate fraction of the present dose limit of 1 $mSv\ a^{-1}$. The appropriate fraction, called the dose constraint, is usually a fraction (0.1 – 0.3) of the dose limit (IAEA, 1999*a*).

The term risk is usually defined in safety assessments as the product of the probability that an individual will be exposed to the radiation and the probability that the exposure will give rise to a detrimental health problem in the individual. The safety criterion commonly used for the risk indicator states that the predicted radiological risk to individuals, arising from the waste disposal facility, should not exceed $5 \cdot 10^{-5}$ fatal cancers or other serious generic effects in a year (IAEA, 1999*a*).

6.6.2 Description of the Disposal System

The components of the disposal system, the flow of information between them, and the information that is required to characterize the different components, have already been discussed in Chapter 4. During the first iteration of a post-closure safety assessment, the disposal system description is mainly limited to a description of the available information in the different elements of Figure 4–1. The emphasis will therefore be placed more on the existing data, rather than the collection of new data. A detailed system characterization is therefore not required at this stage. However, the emphasis will shift towards a more detailed site characterization and the collection of new information and data in subsequent iterations. This may also provide additional information for assumptions and constraints imposed on the assessment, which may require an update of the assessment context.

As mentioned before, a summarised description of the disposal system can be included logically in the assessment context, because it contains information that could influence assumptions to be documented in the assessment context. Together with the assessment context, the system description will therefore form the basis for the description and justifications of scenarios in the next step of the safety assessment process, and to a great extent on the success of the assessment.

A considerable effort was often directed towards the collection and interpretation of site specific data in the past that often led to an over-characterization of the site. Since this is a time consuming and expensive exercise, it is very important that new data and information accumulated in subsequent iterations should complement the safety assessment, particularly the regulatory compliance decision. This aspect will be discussed further in Chapter 9.

6.6.3 Description and Justification of the Site's Future Evolution

Radioactive waste has the advantage above chemical and toxic waste that its activity decreases with time. However, the long half-lives of many isotopes makes it virtually indestructible as far as human life is concerned. The major effort in a post-closure safety assessment is therefore to determine what effect the radioactive waste will have on future generations and under future conditions, something that is obviously not known.

An approach commonly used to circumvent this problem is to collate and screen all currently available information on the characteristics of the disposal system (e.g., behaviour of people, climate change, geohydrological conditions), and any other natural or human induced condition. Personal judgement is then used to develop a number of scientifically sound descriptions, commonly referred to as *scenarios*, of the future conditions at the site.

According to the Oxford English Dictionary the word scenario (like model) has so many meanings and is so overused that it is not always possible to determine its precise meaning. However, in this thesis the word will be used exclusively in the following sense.

> *A scenario is one member in a set of hypothetical features, processes and events devised to rationalise the future behaviour of a repository system in a safety assessment.*

This definition is very similar to the one given by Chapman *et al.* (1995), based on their experience in the SKI SITE-94 study, in which special attention was given to the development of scenarios through formal techniques. Other definitions for the word can be found in IAEA (1999*b*).

The scenario approach has the advantage that it allows for a mixture of quantitative analysis and qualitative judgements of the future conditions. In principle, a number of scenarios can be generated. In practice, however, a base case or normal scenario is selected, followed by a selection of alternative and disruptive scenarios that are most likely to have the greatest impact on system performance. Care should, nevertheless, be taken to ensure that the selected scenarios provide an appropriately comprehensive picture of the system, its possible evolutionary pathways, critical events and system robustness, based on the assessment context and system description. The transparent generation of the scenarios and their associated justification methodology can be very important in building public confidence for the post-closure safety assessment. The scenarios and the methods used to generate them are also often the main focus points of independent reviewers of the safety assessment (IAEA, 1997*a*). More details of the procedures that can be used to develop and justify scenarios can be found in Chapter 8.

6.6.4 Formulation and Implementation of Models

Once the scenarios have been developed, their consequences must be analysed in terms of the assessment context. Although a qualitative analysis may be sometimes sufficient, there are situations where a quantitative analysis will be necessary. The most direct approach to achieve this is to develop a conceptual model for each of the scenarios, as discussed in Section 6.2. The mathematical models, on which the conceptual models are based, can be derived from either sound physical principles or empirical observations, depending upon the level of understanding of the site and the available information

It will seldom be necessary to develop new mathematical models for the scenarios encountered in safety assessment analyses. These scenarios are usually closely related to subjects such as: groundwater flow, heat flow and air dispersion, for which there exist a large number of mathematical models. Only if interactions are identified that are not included in the established models, will it be necessary to develop new mathematical models. The only situation where this may be necessary is when observations indicate that unknown or less understood interactions influence a decision adversely.

As explained in Section 6.2, a conceptual model can be expressed either in the form of an analytical model or a numerical model. Numerical models and in some cases analytical models are today often implemented in commercial or private computer packages which are widely available. Unfortunately, many of these packages do not describe the underlying mathematical model in detail. The possibility therefore exists that a safety analyst may use a computer package that does not satisfy his or her conceptual model for a scenario. *It is therefore very important that the safety analyst should ensure that the mathematical model used in the chosen computer package is consistent with the conceptual model of the scenario that has to be investigated.* One must also keep in mind that in some cases it will not be desirable, nor feasible, to model a scenario with a single computer package. More details of the procedures that can be followed to develop conceptual models for each scenario can be found in Chapter 8.

The level of detail needed in a conceptual model and its associated mathematical model will depend, not only the assessment context, but also the stage of iteration of the safety assessment. It may therefore be possible to begin an assessment with simple analytical models and then proceed to more complex numerical models as the assessment proceeds.

An important question to ask in choosing a conceptual model for the assessment of a radioactive disposal system is, how will the uncertainties be handled? For example, will a deterministic model (perhaps adjusted for statistically distributed constitutive parameters and boundary conditions) be sufficient, or does one need a full probabilistic model? This will have a significant implication on the way the rest of the assessment will be done and should therefore be defined in the assessment context. Although some regulatory authorities may provide guidance in this connection, it may not be a bad idea to use both types of models during the first one or two iterations of the investigation. Attention should also be given to the type and quantity of data needed for the model. The data collected during the initial characterization of the disposal system, and generic data may be sufficient for the first few iterations steps, but latter iterations may require more or other types of site-specific data. It therefore makes sense to make sure that the computer package(s) used in the modelling can accommodate these additional data.

6.6.5 Consequence Analysis

Once the necessary models are formulated, the major purpose of the modelling exercise is to determine what consequence will a particular scenario have on the future development of the disposal site—hence the name consequence analysis. For this purpose, the definition in Section 6.3 is very important. This emphasizes that a safety assessment analysis is a prospective evaluation. Not with the intent to predict its actual behaviour in the future, but rather to understand the behaviour of the system better and to reflect the importance of specific components with respect to the compliance criteria.

It is important to ensure that the consequence analysis results are consistent with the assessment context, for example, that the correct endpoints are calculated over the time frame of concern. Depending on the assumptions made in the assessment context and the models developed during the previous stage, the analysis will be done in a deterministic or probabilistic mode or both.

Incorporated into the consequence analysis, is uncertainty and sensitivity analysis. *Uncertainty analysis* can be defined as an estimation of uncertainties and error bounds of the quantities involved in and the results from the solution of a problem (IAEA, 1993*a*). This requires the application of statistical techniques and definition of the input data information in probabilistic form. Uncertainty analysis will be discussed in more detail in Chapter 8.

Sensitivity analysis is a quantitative examination of how the behaviour of the system varies with changes in the values of the initial conditions, boundary conditions, parameters built into the code and parameters that must be provided as input data. Attention should in addition also be paid to how sensitive the results are to the general assumptions on the system behaviour and whether all the considered processes needs to be included in the model development or not (IAEA, 1993*b*). Two common approaches often used in sensitivity analysis (IAEA, 1993*a*) are:

(a) Parameter variation, in which the variation of the results is investigated for changes in one or more input parameter values within a reasonable range around a selected reference or mean value, and

(b) Perturbation analysis, in which the variations of the results with respect to changes in all the input parameter values are obtained by applying differential or integral analysis.

It is clear from Figure 4–1 that a consequence analysis will be required for one or more of the near-field (engineered barriers), the geosphere (natural barriers) and the biosphere. Which component(s) to include in the analysis will dependent very much on the type of assessment, that is whether it is a full safety assessment, or just to investigate the performance of one component. Consequence analysis forms therefore an integral part of any post-closure safety assessment, as shown by the more detailed discussion in Chapter 7.

6.6.6 Interpretation of Results

The first opportunity a safety analyst will have to interpret the results is after the modelling of at least one of the different scenarios has been completed. However, a meaningful interpretation of the assessment in terms of the assessment context can only be done after a first iteration of all scenarios. At this stage he or she will have quantitative data available to compare with the regulatory and other criteria, outlined in the assessment context, to assess reasonable assurance of compliance.

The interpretation of data is a multi-faceted process in that several varied and sometimes competing factors must be brought together and reconciled to reach a decision as to whether the system and post-closure safety assessment are adequate. A major activity will therefore be to screen and condition data from the modelling results (even eliminate unnecessary data) before comparing it with the relevant criteria. Once this is done, the analyst will be able to make a decision.

Since the post-closure safety assessment is an iterative process, the process will usually follow the branch to the second decision process in the flow chart of Figure 6-1, if compliance cannot be demonstrated after the first iteration step. In this case, a second iteration of the safety assessment is appropriate, which—according to Figure 6-1—requires an assessment for further information needs. The information that is required will be determined by the reason for the inadequacy, although it can be expected that it will be those input parameters and assumptions that have the greatest impact on system performance. Once the information needs have been identified, the next step is to assess the worth of the data associated with collection of data to resolve the input parameter and model assumption inadequacy. The assessment of data worth will again be discussed in Chapter 9. If subsequent iterations show that it will be inefficient to further try to improve system components and the system still does not comply with the regulatory criteria, the system must be rejected.

6.6.7 Confidence Building

As mentioned in Section 4.3.2, it will most likely be impossible to develop a disposal a site for the disposal of radioactive waste without the confidence, co-operation and participation of the local population and other interested parties. Scientists and engineers working on the technical aspects of radioactive waste disposal have developed an international consensus that the waste can be permanently managed in a manner that protects the environment and public health. However, this view is not necessarily shared by the public (IAEA, 1992). A post-closure safety assessment must therefore be conducted in such a way that this confidence building is always enhanced. This may be achieved by open discussions with all interested parties (stakeholders) in the early stages of the assessment, supplemented later with discussions of the site-specific results as the safety assessment proceeds. Apart from the general public, other stakeholders include national and provincial governments, regulatory agencies, local communities, the media, environmental groups, and employees in the nuclear industry and opinion leaders.

It may also be helpful to demonstrate to the interested parties that the assessment is based on sound scientific and engineering principles. For this purpose, efforts are directed towards three areas, namely model verification and validation (natural analogues), quality assurance, and critical peer review.

Verification is the process of showing that a mathematical model, or the corresponding computer code, behaves as intended, i.e. that it is a proper mathematical representation of the phenomena it represents and that the equations are correctly encoded and solved. *Validation*, on the other hand, is a process carried out by comparison of model predictions with field observations and experimental measurements (IAEA, 1993*a*). A model is considered validated when it is able to simulate a sufficient number of observations on the phenomenon for which it was developed. Natural analogues (Apted and Engel, 1991) may be especially useful in this regard. These are features similar to that investigated in the safety assessment, but occur naturally in nature for very long times. The confidence in a site assessment model will certainly increase considerably if it is able to simulate its natural analogue counterpart accurately (IAEA, 1993*a*). Most of the known natural analogues are associated with physical and chemical processes in the geosphere, such as the migration of radioactive materials and other deposits and matrix diffusion. However, there are others relating to the near-field and biosphere. These include, for example, corrosion and penetration of waste containers, waste form degradation, stability of backfill materials, elemental solubility, sorption processes, microbial activity and speciation (IAEA, 1992).

Quality assurance comprises all those planned and systematic actions necessary to provide adequate confidence that a structure, system, process or component will perform satisfactory (IAEA, 1993*a*). Quality assurance is normally applied to repository design and site characterization. However, the need to generate confidence in safety assessments dictates that appropriate quality assurance methods is also applied to aspects such as data collection, model development, computer code development and integrated assessments. These measures will contribute significantly to reducing uncertainty and increasing confidence in safety assessment results (IAEA, 1993*b*). The treatment of uncertainties in safety assessment will be discussed in detail in Chapter 8.

Critical peer review forms an integral part of the scientific process to gain acceptance of a new concept. This allows a scientist to publish his results in the open literature where it can be reviewed and criticized by other experts in the field. It is therefore logical to assume that a safety assessment can also benefit from such a critical peer review. Such a peer review should preferably be done by a group of people who are experts in the field, but not associated with the assessment to be reviewed. The major task of such a group will be to review the assessment critically and make suitable recommendations (IAEA, 1993*b*).

CHAPTER 7

PROSPECTIVE EVALUATION OF A DISPOSAL SYSTEM

7.1 INTRODUCTION

The assessment of a radioactive waste disposal site was historically often seen as an attempt to predict the behaviour of the site far into the future, using deterministic or predictive models. The models, which are discussed in Section 6.2, are based on the physical principles that underlie a specific phenomenon and should therefore be able to predict the future behaviour of the site. The models, unfortunately, frequently require information on parameters whose behaviour cannot be determined with certainty far into the future. It is therefore impossible to apply the models in the historical sense of a safety assessment. However, as pointed out in Section 6.3, this is neither necessary nor desirable. What one really wants to do in a safety assessment is to be able to give a reasonable assurance that the site will comply with the regulatory criteria for safety. This led to the introduction of the term compliance calculations (Kozak, 1997).

Compliance calculations in a safety assessment essentially involve the prospective evaluation of radionuclide transport through the near-field, geosphere and biosphere, where people will be exposed to the detrimental effects of the radiation. As shown by the discussion of the near-field and geosphere in Sections 7.2 and 7.3 respectively, this can often be achieved by using predictive models, but applied in a heuristic and not predictive sense.

Heuristic predictive models can also be used to evaluate physical structures in the biosphere, such as wind erosion of land-fills, but not the biodynamical components of the biosphere (food chains, for example). These components are best evaluated with the phenomenological models that are described in Section 7.4.

7.2 THE NEAR-FIELD

7.2.1 General

The definition of the near-field in Section 4.3.4 indicates that it is the primary source of radiation to the environment at a waste disposal site, and at the same time the first defence against the leakage of radioactive nuclei from the site. It must therefore be considered as the most important component in the safety assessment of a radioactive waste disposal system.

The evaluation of the near-field in a safety assessment context involves the characterization of the groundwater and vapour motion, especially if the repository is situated in the unsaturated zone, and the associated transport of radionuclides. The mass flux of radionuclides across the boundary of the near-field is thus the primary source of radiation at a waste disposal site.

Radionuclides can escape from the near-field in either one of two phases—a gas phase and a solid phase. The gas phase will ultimately escape to the atmosphere where its transport can be studied with an atmospheric transport model (Ellis, 1999). This discussion, however, will concentrate exclusively on the groundwater pathway in the geosphere, although it is recognized that the atmospheric pathway may be important, especially for near-surface disposal systems.

The near-field characteristics of a near-surface disposal system will differ considerably from that of a geological disposal system, because of differences in inventory, engineered barriers and design of the repository. The two systems will therefore be discussed separately.

7.2.2 Near-Surface Disposal Systems

The rate at which radionuclides will leak from the near-field will be determined essentially by the degradation of the containers and engineered barriers, and the subsequent distribution of the nuclides in the near-field. Although there are other factors that may cause the containers and engineered barriers to fail, the metal and concrete used in the conditioning of the waste and engineered barriers are especially susceptible to infiltrating water. Some of the processes responsible for the degradation of these structures by infiltrating water are listed in Table 7-1. Phillip and Clifton (1989) also provide a good overview of the processes, including estimates of the service life of different concrete structures.

The strength of near-field source, caused by the infiltration of water, will be determined by a number of factors, of which the following four are the most important (Sullivan 1997):

(a) the inventory by waste form and container,

(b) container performance,

(c) waste form performance, and

(d) transport of the contaminants through the disposal facility.

Table 7-1 Degradation processes in engineered barriers associated with the infiltration of water, taken from Andrade (1997).

Process	Mechanism	Agent
Biological	Expansion, leaching	Bacteria
Electrochemical	Corrosion	Electrochemical potentials
Mechanical	Cracking	Shrinkage, deformations and loading
Physical	Expansion, erosion	Freezing and thawing, salt crystallization, delamination
Physico-chemical	Leaching, hydrolysis, expansion	Acids, sea water, ammonium salts, sulphates, alkali-aggregate reactions

The inventory, taken as the quantity of radioactive material in the waste, is obviously the source of all the radiation that will be released in the geosphere. There are essentially two approaches that can be used to determine this quantity. The first is on a container or package basis and the second as a site average. The site-average approach is considerably simpler than the per package approach and is the one commonly used in site assessments.

Although the inventory is the major source of the radionuclides, the source strength will be determined by the performance of the container and waste form, that is the abilities of the container and waste form to limit the migration of the nuclides from the source. An intact container will usually be able to shield the waste completely from the infiltrating water, with the result that one has only to take degraded containers into account when estimating the source term. This can be a very time consuming and expensive exercise, judging from the time-dependent nature of the processes responsible for the degradation of containers and engineered barriers in Table 7-1. Such a computation is, however, rarely necessary or desirable in safety assessments, unless a more simplified approach has a negative impact on the regulatory decision.

The ability of the waste form to retain radionuclides will be determined mainly by the types of wastes in the repository and the distribution of the radioactive nuclides in the waste form. Based on the knowledge of the inventory, the analyst should attempt to define the leaching mechanisms appropriate for each waste type. Sullivan (1997) defined four models for the release of radionuclides from the waste form: rinse release with partitioning, diffusion release, uniform degradation release and solubility-limited release.

(a) **Rinse Release with Partitioning** – The radionuclides are assumed to be situated on the surface of the waste form and will be released as soon as the waste form is infiltrated by water, subject to equilibrium partitioning and solubility limits. Rinse release without partitioning therefore means that the entire inventory will be released instantaneously, once a container fails.

(b) **Diffusion Release** – The waste is assumed to be uniformly distributed throughout the waste form and can only escape through molecular diffusion.

(c) **Uniform Degradation Release** – The waste form releases the radioactivity at a uniform rate. Uniform degradation release is therefore similar to rinse release without partitioning, except that the radionuclides are released at a uniform rate and not instantaneously. An example of a situation where this type of release can occur, is when the container has corroded completely and the waste form itself decays.

(d) **Solubility-limited Release** – Radionuclides are instantaneously released to solution until the solubility limit is reached. Further releases are controlled by the migration of radionuclides away from the waste form.

Radionuclides are mainly transported from the waste in near-surface facilities by the motion of groundwater through advection and dispersion. However, there are also other mechanisms, such as diffusion, solubility, sorption and radioactive production and decay that will contribute to the transport. The contributions of the different mechanisms, however, may vary with site and waste type. For example, diffusion will be the dominant mechanism in a site with intact containers.

It will clearly not be easy to establish a suitable conceptual model for the transport of radionuclides within and out of the near-field, unless there is sufficient experimental data, although the situation may vary with site and waste form. Solid concrete, for example, is a porous material. Its moisture retention and transport properties can therefore be characterized with parameters similar to that used in groundwater flow models (Shuman and Rogers, 1992). However, such a model may have to be replaced by a fracture flow model, once the concrete begins to crack (Walton *et al.*, 1990; Walton and Seitz, 1991). The transport of radioactive nuclides in the near-field is consequently often modelled with less general, but more conservative models in safety assessments of near-surface disposal facilities. The time-delay model, which assumes that all the nuclides, will be released simultaneously but only after a certain time, is an example of such a model.

7.2.3 Geological Disposal Systems

As mentioned in Section 3.2.3, geological disposal systems are mainly aimed at the disposal of high-level radioactive wastes in deep geological formations, although there is now a tendency to dispose low- and intermediate-level waste in similar structures. The exact construction of the repositories used in these systems will therefore differ considerably, depending on the design criteria, and whether the repository is situated near, or far below the surface. A schematic illustration of such a facility is shown in Figure 7–1.

Geological disposal systems are usually constructed with multiple engineered barriers, as shown in Figure 7–1. Since the main aim with the barriers is to prevent the release of radionuclides as far as possible, the number included may vary from one to another.

A geological repository will be drained and ventilated during its operational life. It is therefore unlikely that radionuclides will be transported from it during this time. However, the conditions in the near-field will change gradually with time once the repository is closed. The ensuing 'normal conditions', can be divided in a number of distinct stages (Chapman and McKinley, 1987).

(a) Water will first infiltrate into the zones of the host rock, drained during the operational phase, and then the backfill material, causing swelling and plastic flows in some materials containing swelling clays, thereby sealing constructional gaps.

(b) Radiogenic heat, generated especially by high-level waste, will cause a thermal gradient to develop within a few years after the closure of the facility that may last for

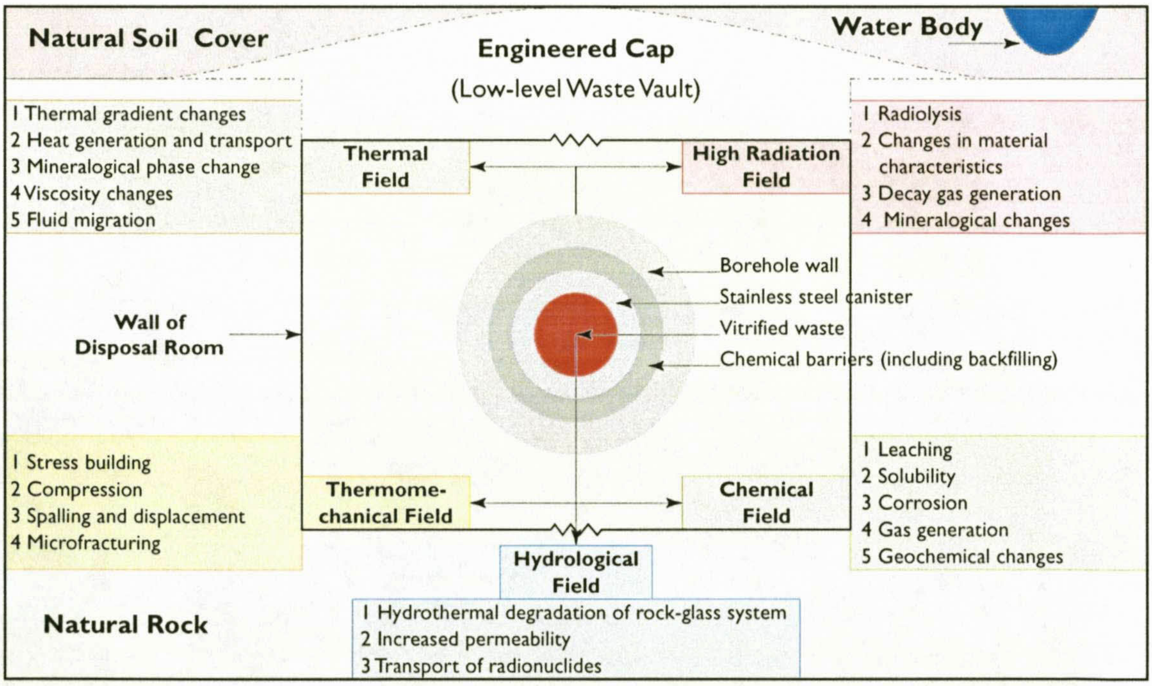


Figure 7–1 Schematic partial cross-section through the wall of a chamber in a hybrid geological disposal system, with the different processes and components that could influence the performance of the system's near-field. [Redrawn from IAEA (1992).]

thousands of years. This will heat the engineered barriers and the surrounding host rocks, thereby disrupting the normal groundwater flow pattern in the geosphere of the facility.

(c) As groundwater infiltrates the backfill, its chemistry tends to change due to a complex series of reactions, such as ion-exchange, dissolution and precipitation of particular minerals. The degree of change will depend on the type of material used as backfill.

(d) The infiltrating water will ultimately reach the canister and begin to corrode it. The low seepage velocity of the infiltrating water will be not only limit the corrosion of the containers, but also restrict the mass transport at this stage to diffusion. The containers, nevertheless, will fail sometime, either because of corrosion or mechanical forces. However, this does not mean that radionuclides will be released immediately to the water, since the remnant canister material and corrosion products may still retard the release of the radionuclides.

(e) The waste form begins to disintegrate when it is exposed to the water, release the radionuclides, either in solution, as particulates or colloids, and begin to migrate through the near-field. The rate at which the radionuclides are released into solution is determined by the solubility of the radionuclides and the rate at which the matrix disintegrates. The chemical environment in the contact zone is very complex and difficult to evaluate.

(f) Once released from the near-field, the rate of migration of radionuclides though the geosphere is determined by the geohydrological conditions along the flow path and the extent of retardation processes.

7.3 EVALUATION OF THE GEOSPHERE

7.3.1 General

The radionuclides released by the near-field to the geosphere pose no risk to humans until they are released in the biosphere. Special attention should therefore be given to the evaluation of the geosphere as a natural barrier for the transport of radionuclides from the near-field to the biosphere. The approach commonly used for this purpose is to use the information gathered during the site's characterization and the mathematical models for groundwater flow and mass transport in Section 6.2, to develop a suitable conceptual model for the geosphere of the site. Of particular importance in this regard are the physical principles embodied in the mathematical models, that are discussed below.

Although there already exists a large volume of literature on the subject, the time scales involved and the amount of data available often introduce considerable uncertainties into the approach. Several countries have consequently developed underground research laboratories to obtain more site-specific data and improve the various mathematical and conceptual models needed for the approach (Kickmaier and McKinley, 1996). Conservative assumptions and analytical models are also sometimes used to compensate for the uncertainties, especially in preliminary safety assessments. However, site-specific data and models will be needed to license a particular disposal system (IAEA, 1993*b*).

7.3.2 Advection

Radionuclides in the geosphere are predominantly transported with the water present in the pores or fractures of the geosphere matrix—a process known as advection. The advective transport of the nuclides, represented by the term $\mathbf{q}\cdot\nabla c$, in Equation (6.4), is mainly controlled by the concentration gradient of the dissolved radionuclides and the Darcy velocity of the water. Since the Darcy velocity differs considerably from an unsaturated to a saturated medium, the degree to which the geosphere matrix is saturated with water can play a significant role in the transport of the radionuclides by the water.

The flow of groundwater in consolidated rocks is often completely controlled by fractures (Botha *et al.*, 1998). One therefore has to replace the simple porous flow models of Section 4.4 with more complex mathematical models that takes the aperture, direction and frequency of the fractures into account, when studying these formations. It is consequently not an easy task to predict travel times, flow rates and contaminant discharge points in these formations.

7.3.3 Mechanical Dispersion

The term mechanical dispersion is a collective term used to describe microscopic processes that cause dissolved solids to mix and spread in groundwater. It is generally assumed that the phenomenon is caused by interactions between the dissolved solids, pore structure and the microscopic velocity of water in the pores (Freeze and Cherry, 1979), although very little is known of the exact mechanisms responsible for the phenomenon.

The main effect of mechanical dispersion in groundwater is to spread the contaminants over a larger volume than would have been the case if dispersion were absent, and in the form of a plume. This tends to reduce the peak concentrations of contaminants at the front of the plume, but at the same time increase the concentrations on its sides.

There do not exist methods with which the generalized microscopic Darcy velocity can be measured routinely (Botha, 1996). Mechanical dispersion is therefore usually expressed in terms of the macroscopic quantities, such as the seepage or Darcy velocity of the fluid, through the second rank tensor, $\mathbf{D}_h$, in Equation (6.6). This representation has an important consequence for groundwater pollution studies, as it implies that the main axis of a contamination plume may not coincide with the direction of the macroscopic Darcy or seepage velocities. The behaviour, which is particularly noticeable in heterogeneous aquifers, unfortunately, is often disregarded in studies of groundwater pollution.

7.3.4 Molecular Diffusion

It is well-known that a dissolved solid will tend to spread evenly though the fluid, even if the fluid is at rest. The random thermal motion of molecules in the solution causes this mechanism, known as molecular diffusion, and represented by the molecular diffusion tensor in Equation (6.5). This mechanism is therefore always present in all fluids and solids, except at the zero-point temperature (^{0} K).

The major effect of molecular diffusion is to transport solids from areas with high concentrations of a substance to areas with low concentrations of the same substance. Molecular diffusion depends only on the diffusing substance, the medium in which it is embedded and its concentration gradient within that medium. It can therefore occur even in stationary groundwater or between groundwater and the surrounding rock matrix.

The first impression of molecular diffusion is that it will retard the motion of contaminants through the subsurface of the earth. Because of this and the fact that the molecular diffusion coefficient for ordinary substances in water is not very large, molecular diffusion is often neglected in groundwater pollution studies (Botha, 1996). However, such an approach can have dire consequences, not only for radioactive waste disposal systems, but also for all waste disposal systems, for a number of reasons. The first is that molecular diffusion is not restricted to fluids alone, but can occur in any form of matter. This means that no engineered barrier, even those that may last for thousands or millions of years, will ever be able to prevent radionuclides to escape from the near-field. A second reason is that molecular diffusion will tend to broaden the advection plume further, even contaminate adjacent aquifers separated by an impermeable flow boundary from the contaminated one.

To illustrate the latter behaviour, consider the two-layered heterogeneous aquifer system in Figure 7–2. The higher hydraulic conductivity of Layer 2 will obviously cause this layer to form a preferential flow path in the aquifer, along which advection of pollutants will dominate. However, this will create a concentration gradient between the contaminants in Layer 1

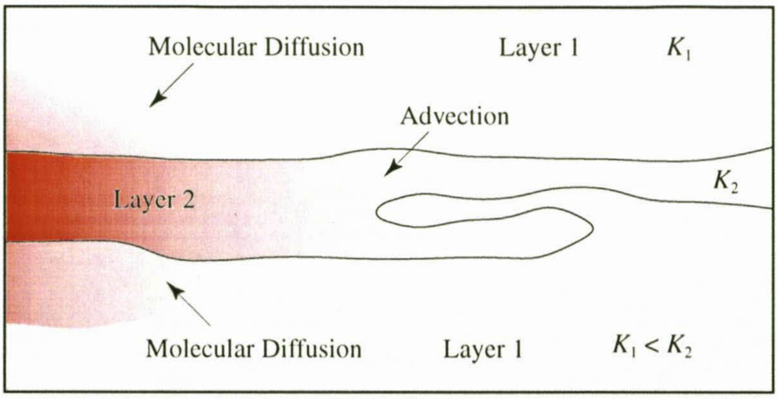


Figure 7–2 Schematic illustration of advection and molecular diffusion in a two-layered heterogeneous aquifer, after Botha (1996).

and those in Layer 2. This gradient will not only cause the pollutant to diffuse into Layer 1, but also decrease the concentration of the pollutant in Layer 2, thereby retarding the pollutant's motion and at the same time smooth out any variations in the concentrations of the two layers. Studies of contaminant transport, over the last couple of decades, have shown that molecular diffusion can often dominate advection in some aquifers, because of this behaviour. This is especially the case in heterogeneous and fractured rock aquifers (Sudicky, 1983).

7.3.5 Retardation Processes

Molecular diffusion is not the only mechanism able to retard the transport of a dissolved solid in a porous or fractured medium. Another very important example is sorption. Although sometimes regarded as a single phenomenon, there are quite a number of different mechanisms that can contribute to sorption. Some of the most important of these are:

(a) absorption of the dissolved molecules by the rock matrix,

(b) precipitation of the dissolved molecules caused by changes in the pH-value of the groundwater,

(c) ion exchange and the isomorphic substitution of ions between the solute and the rock matrix,

Although it is possible to account for all these mechanisms in the conceptual model of a waste disposal site's geosphere, the existing geochemical databases are often not sufficient for this purpose. Moreover, there are not many computer resources in the world that can be used to evaluate such a conceptual model. Almost all safety assessments of radioactive waste disposal systems are consequently limited to conceptual models in which the sorption mechanisms are presented by a single parameter, K_d, known as the *distribution coefficient* or simply K_d*-value*, defined by the equation

$$s(\mathbf{x},t) = K_d c(\mathbf{x},t)$$

(also known as the Freundlich isotherm) where $s(\mathbf{x}, t)$ is the mass fraction of the solute that partakes in sorption, and $c(\mathbf{x}, t)$ the volumetric concentration of the solute. Substitution of this value for $s(\mathbf{x}, t)$ into Equation (6.4) yields

$$R_d \mathrm{D}_t c + \mathbf{q}\cdot\nabla c = \nabla\cdot(\theta \mathbf{D}_h \nabla c) - \lambda R_d c + (c_0 - c) f(\mathbf{x},t)$$

where

$$R_d = \theta + \rho_b K_d$$

is known as the retardation coefficient.

The distribution coefficient represents a very efficient method with which to study sorption in the framework of a safety assessment, despite its simplistic nature. It is very important though that the K_d-values used, should be representative of the site. Too high K_d-values will tend to concentrate the radionuclides in the vicinity of the near-field, while too low K_d-values will tend to rinse the near-field of radionuclides, with adverse effects for other components of the assessment. Too high K_d-values can, for example, lead to an overestimate of the dose humans will receive on intrusion of the site, while members of a radioactive chain will not have sufficient time to produce daughter elements in the second case. The latter situation should especially be avoided in the case of elements, such as ^{238}U, whose daughters (^{226}Ra, ^{222}Rn and ^{210}Pb) are often more active than the parent (Sullivan and Kozak, 1996).

7.3.6 Radioactive Decay

Radioactive decay itself will reduce the activities of the radionuclides in the geosphere with time. Unfortunately, the half-lives of some of the radionuclides in radioactive wastes are too long for their activities to be reduced noticeably during the time they are transported through the geosphere. One cannot therefore rely on radioactive decay alone to reduce the activities of all radioactive waste to acceptable limits in the far future.

A second difficulty with radionuclides is that the daughters of elements in decay chains are completely different elements, in general, with completely different physical and chemical properties, and therefore transport properties. Although possible, it is not always easy to provide for these daughters in the conceptual models used in the safety assessments of radioactive waste disposal systems. One approach to avoid this is to assume that the parents and daughters are always in radioactive equilibrium. Since the rate at which a parent produces a daughter is known, one only needs to compute the activity of the parent and used this to compute the activities of the daughters. This approximation yields excellent results when the half-lives of all daughters are considerably less than that of the original parent, and the transport rate is slow enough to ensure equilibrium.

Another approach that is also used is to assume that the parent and daughters are transported in the same manner and at the same rate. One then only need to compute the activity (c_j) of the parent in the geosphere and use the rate at which the i-th daughter is produced by the j-th parent and the generalized Bateman equations (Skrable *et al.*, 1974)

$$\frac{\partial c_i}{\partial t} = -\lambda_i c_i + \sum_{j=1}^{N} f_{ij} c_i$$

to compute the activities of the daughters (c_i) as a function of time.

7.4 EVALUATION OF THE BIOSPHERE

The transport of radionuclides through the biosphere are better understood than through the natural and engineered barriers (IAEA, 1993*b*). The major uncertainties occur in the evolution of the biosphere in the future and the changing patterns of human behaviour.

It follows from Figure 7–1 that the results from the evaluation of the geosphere and the characteristics of human behaviour form the basis for the evaluation of the biosphere. Two kinds of models are conventionally used to evaluate the behaviour of the biosphere—*deterministic* and *phenomenological* models.

Deterministic models are conventionally used to describe mechanistic phenomena, such as wind erosion of land-fill and soil surfaces, while the phenomenological models are used for the biological pathways, such as the food chain. The phenomenological models used in radioactive safety assessments are usually based on transfer coefficients. Although these models are not able to explain a phenomenon, as is the case with deterministic models, they are simple to use and based on phenomenon specific data (Torres and Simon, 1997).

The main purpose with models of the biosphere is to estimate the radiation doses that people will receive when exposed to the environment. The potential pathways that may cause such an exposure are usually divided into two groups: internal and external. The internal pathways consist of the ingestion of contaminated food; the drinking of contaminated water and the inhalation of contaminated air. The external pathways, which are more diffused include exposure to contaminated ground, houses built with contaminated materials and swimming or bathing in contaminated water. These pathways are usually divided into 'com-

Table 7-2 Parameters and equations often used to compute the radiation dose received by humans through pathways in the biosphere of a radioactive waste disposal site.

General parameters		
DFing	= Dose conversion factor for ingestion	$Sv\ Bq^{-1}$
Cs	= Concentration in soil	$Bq\ kg^{-1}$
Cw	= Concentration in water	$Bq\ L^{-1}$
CMMxxx	= Annual individual consumption	L, kg
Drinking water dose Dw		
Dw	= DFing · Cw · CMMwater	$Sv\ a^{-1}$
Dose from milk consumption Dmi		
Dmi	= DFing · CMMmilk · DFmilk · A	$Sv\ a^{-1}$
DFmilk	= Transfer factor for milk	$d\ L^{-1}$
A	= CMCgrass · Cs + CMCwater · Cw	–
CMCgrass	= Daily grass consumption by cows	$Kg\ d^{-1}$
CRgrass	= Concentration ratio from soil to grass	–
CMCwater	= Daily water consumption by cows	$L\ d^{-1}$
Dose from meat consumption Dme		
Dme	= DFing · CMMmeat · DFmeat · A	$Sv\ a^{-1}$
DFmeat	= Transfer factor for meat	$d\ kg^{-1}$
A	= As under 'dose from milk consumption'	–
Dose from consumption of leafy vegetables Dbl		
CRleaf	= Concentration factor from soil to leafy vegetables	–
Dbl	= DFing · CMMleaf · CRleaf · Cs	$Sv\ a^{-1}$
Dose from cereal consumption Dget		
Dget	= DFing · CMMcereal · CRcereal · Cs	$Sv\ a^{-1}$
CRcereal	= Concentration factor from soil to cereal	–
Dose from consumption of root vegetables Dwu		
Dwu	= DFing · CMMroot · CRroot · Cs	$Sv\ a^{-1}$
CRroot	= Concentration factor from soil to root vegetables	–
Dose from fruit consumption Dft		
Dft	= DFing · CMMfruit · CRfruit · Cs	$Sv\ a^{-1}$
CRfruit	= Concentration factor from soil to fruit	–
Dose from poultry consumption Dpt		
Dpt	= DFing · CMMpoultry · DFpoultry · B	$Sv\ a^{-1}$
DFpoultry	= Transfer factor for poultry	$d\ kg^{-1}$
B	= CMHcereal · CRcereal · Cs + CMHwater · Cw	–
CRcereal	= See 'dose from cereal'	–
CMHcereal	= Daily cereal consumption by hens	$Kg\ d^{-1}$
CMHwater	= Daily water consumption by hens	$L\ d^{-1}$
Dose from egg consumption Dei		
Dei	= DFing · CMMegg · DFegg · B	$Sv\ a^{-1}$
DFegg	= Transfer factor for eggs	$d\ kg^{-1}$
B	= See 'dose from poultry consumption'	$d\ kg^{-1}$
Dose from fish consumption Dfi		
Dfi	= DFing · CMMfish · CRfish · Cw	$Sv\ a^{-1}$
CRfish	= Concentration factor from water to fish	–

partments' in which the radionuclides are distributed instantaneously and homogeneously in biospheric models. The models also assume that the rate at which the radionuclides are transported from one compartment to another is proportional to their concentration in that compartment. The transport of the radionuclides between the various compartments (e.g. water to grass, grass to cow, cow to milk or beef, milk or beef to man) is then modelled with two simple parameters the transfer coefficient and concentration ratios (CR). The dose received by a person is finally computed from dose conversion factors, as illustrated in Table 7-2. These factors are based on the experience with radioactive materials over the last century and make provision for variations of dose per unit intake, the age of the individual and the chemical properties of the radionuclide (IAEA, 1995*b*).

CHAPTER 8

UNCERTAINTIES IN SAFETY ASSESSMENTS

8.1 INTRODUCTION

The discussion of the principles and the evaluation of safety assessments in Chapters 6 and 7 may create the impression that this is a 'squishy' problem. That is, the analysis appears to be rigorous on the surface, but are underlain by large and irreducible uncertainties (Kozak, 1994). Safety assessments must extrapolate over large spans of time and space, into conditions that cannot be observed empirically. Moreover, the currently prevailing conditions at a site are often so complex that it confounds all attempts to *predict the actual behaviour* of the system accurately. However, as mentioned in Section 1.2, the main aim of a safety assessment is to determine the extent to which a waste disposal site will comply with the regulatory constraints and not to predict its actual future behaviour. The uncertainties inherent in quantitative post-closure safety assessments thus mean that results are to be considered as representative indicators of safety, rather than definitive predictions of the future radiological impact (Watts *et al.*, 1999). A safety assessor is therefore not so much concerned with numerical uncertainties in the final results of the different models used in the assessment, as is customarily the case when using models to describe physical phenomena. What is important here is whether *changes in the assumptions, models and parameters used in the assessment can affect the decision on the regulatory compliance of the disposal system adversely* (Kozak, 1997). There is consequently a vast difference between a numerical uncertainty analysis as practised in many branches of science and engineering, and uncertainty analysis in the assessment of radioactive waste disposal sites. The Scientific Committee 87-3 of the NRCP therefore introduced the term *Importance Analysis* in an attempt to clarify the difference between the two approaches, and to stress the fact the processes associated with a safety assessment are themselves uncertain. However, this does not change the fact that uncertainties do exist in safety assessments, and that they must be addressed, before anyone will accept an assessment with confidence. In fact, the treatment of uncertainty is central to the establishment of the post-closure safety case for a radioactive waste disposal system in the United Kingdom regulatory guidelines (Environment Agency *et al.*, 1997).

The uncertainties associated with safety assessments can be conveniently divided into three categories:

(a) uncertainties related to the unknown future state of the disposal system,

(b) data and parameter uncertainties, and

(c) model uncertainties.

The major sources of the first type uncertainties are discussed in Section 8.2. This is followed by a discussion of the data and parameter uncertainties in Section 8.3 and model uncertainties in Section 8.4. Uncertainties associated with the future evolution of the disposal system and justification step of the assessment, introduced in Section 6.6.3, are discussed in Section 8.5. Various methods that can be used to reduce the second and third types of uncertainties in safety assessments, including deterministic and probabilistic approaches to reduce data and parameter uncertainties, are discussed in Section 8.6, and verification and validation to reduce model uncertainties in Section 8.7.

8.2 UNCERTAINTY IN THE UNKNOWN FUTURE STATE OF THE DISPOSAL SYSTEM

The major sources of uncertainty in the assessment of a radioactive waste disposal system are without a doubt the future evolution of the system. One of the major difficulties experienced in this regard is that a post-closure assessment must often be performed even before the disposal system is in place. The assessment could therefore be influenced adversely by factors such as potential differences between the original design of the repository and the eventual facility, as well as the intended and actual waste emplacement schedule and configuration. Factors such as these can, to a certain extent, be controlled on a regulatory basis. However, there are others, such as the habits and behaviour of people in the future and natural and human induced disruptive events and processes, which cannot be controlled in this way, and must be addressed explicitly in the assessment.

The approach commonly followed today to address the future evolution of a disposal system is scenario generation, introduced in Section 6.6.3. In scenario generation, personal judgement is used to assign individual probabilities to each of the postulated scenarios that can be compared with a prescribed risk standard, without the restriction that the probabilities of the selected scenarios should sum to unity. This is not a trivial task. A structured approach is therefore necessary to ensure that the process is systematic, well documented, and that the uncertainties have been addressed adequately to arrive at a credible decision. To assign probabilities to each of the individual postulated scenarios may not be mathematically rigorous, but it is more useful than other approaches (Kozak *et al.*, 1999). The key point here is to use all the available information, and information that can be derived from the analysis itself, to justify the inclusion of a specific scenario in the assessment.

8.3 DATA AND PARAMETER UNCERTAINTIES

The term datum (plural data) is defined in the Oxford English Dictionary as: *A thing given or granted; something known or assumed as fact, and made the basis of reasoning or calculation; an assumption or premise from which inferences are drawn*. However, the term will be used here exclusively to denote *directly measurable* and not *derived* quantities, for which the term *parameter* will be used. For example, the term data could refer to measurements of water levels in boreholes, while the hydraulic conductivity and storativity are parameters that can be derived from the measurements.

Uncertainties in the data can only arise from measurement errors, according to the previous definition, for which there are essentially two sources: instrument and human errors. It is usually not too difficult for an observer to limit the influence of instrument errors, which are mainly caused by the imprecision and malfunctioning of the available measuring devices, but human errors are often difficult to detect. A few of the major human sources include, for example, incorrect or misapplied measuring techniques, systematic errors and blunders. A common source of systematic errors in safety assessments is where measurements are taken on disturbed samples, because *in situ* measurements are impossible.

Measurement errors will obviously carry over to parameter errors and therefore uncertainties. However, there are additional sources of parameter uncertainties. One source often encountered in safety assessments is the large spatial and temporal variation in many of the data sets. It is therefore not always possible to obtain sufficient data, or ensure that the methods used in deriving the parameters are above suspicion. A common approach to circumvent this, so-called insufficient data uncertainty is to say that the person responsible for the analysis of the data should apply his or her personal judgement. However, nature behaves in very subtle ways. The approach can therefore add even more uncertainty to the data, unless the analyst can demonstrate explicitly that his judgement is supported by the observed behaviour of the phenomenon.

Another major source of parameter uncertainty is that the data used to derive a parameter may not be representative of the parameter. Two of the most common effects responsible for this are: *scale* and *geometric* effects. Scale effects can often be observed in situations where parameters derived from laboratory measurements are used to describe a phenomenon in the field. Geometric effects can arise on various occasions, but the one most commonly observed in practice is associated with the geometry of the aquifer. Most of the mathematical models employed in the study of groundwater flow, including those discussed in Section 6.2, are based on the conventional geometry of a porous medium. Considerable errors could therefore arise if such a model is applied blindly to an aquifer in fractured rocks, or the multi-porous fractured aquifers of the Karoo formations in South Africa.

8.4 MODEL UNCERTAINTY

8.4.1 General

Although data and parameter uncertainties will certainly affect model uncertainty, there are a number of additional uncertainties that are particularly important for the development of models in the analysis of scenarios. These uncertainties arise in all the model types discussed in Section 6.2—mathematical, conceptual, analytical and numerical models. Numerical and many of the analytical models are usually implemented on a computer through computer codes that are often quite complex. It is consequently not always possible to prove that the codes implement the conceptual models correctly and do not contain programming errors. This adds an additional source of uncertainty referred to here as *code uncertainty*. These uncertainties will now be described in more detail.

8.4.2 Mathematical Model Uncertainty

There are basically three sources of uncertainties associated with the formulation of a mathematical model for a physical phenomenon, according to its definition in Section 6.2.3:

(a) An incomplete knowledge of the features, events and processes associated with the phenomena.

(b) The inability to express many of the features, events and processes in exact mathematical terms.

(c) Errors made in the development of the mathematical model.

Although these uncertainties can occur in all models used in safety assessments, they are especially important for the phenomenological models introduced in Section 7.4. As mentioned there, the intention with these models is not to *explain* the behaviour of a phenomenon, but only to *describe* it. Since it is impossible to predict the behaviour of a phenomenon if one cannot explain it, a completely different phenomenological model (or coefficients, in the case of transfer coefficient models) may be needed to describe the phenomenon in the future. Such variations can cause large uncertainties in the computed potential dose or the risk to human health in safety assessments.

It is well-known that many of the constitutive parameters used in deterministic models, such as the hydraulic conductivity and specific storativity of an aquifer, vary in space and therefore subject to parameter and insufficient data uncertainty. What is quite often neglected though is that the parameters are also intrinsic functions of time. Although this dependence may be weak, there are indications that some of them can vary considerably over periods of a few decades. This applies in particular to the hydraulic parameters of the highly elastic Karoo aquifers of South Africa (Botha *et al.*, 1998). Since this dependence cannot be derived from the underlying theory, special attention need to be given to the time-dependence of these parameters, either during the site characterization or pre-closure assessment of a radioactive waste disposal system.

8.4.3 Conceptual Model Uncertainty

The term conceptual model was defined in Section 6.2.4 as the governing equation, or set of equations, with its associated constitutive parameters, boundary and initial (if required) conditions, to describe a specific realization of a given phenomenon. This may not always be an easy task. For example, it is very laborious and expensive exercise to determine the specific storativity and the three-dimensional hydraulic conductivity tensor of an aquifer (Hsieh *et al.*, 1983). It is therefore often tempting to describe a given phenomenon with a less sophisticated conceptual model. An approach that is frequently applied in such cases, is to replace the three-dimensional mathematical model with a reduced two-dimensional mathematical model in the conceptual model for such phenomena (Botha, 1996). This approach may be acceptable when the conceptual model will only be used for a short period (a few decades), but can yield completely wrong results if the model is to be used for a hundred (in some cases even less) years.

A similar situation arises in those cases where the conceptual model includes a differential equation and one has to prescribe suitable boundary and initial conditions. Although it is usually not too difficult to prescribe suitable initial conditions, the exact boundaries of many phenomena, especially those related to the geosphere, are often not known. A common practice in such situations is to use observations of a phenomenon, such as water levels, to prescribe suitable boundaries and boundary conditions for the conceptual model. Many boundaries used in conceptual models therefore do not reflect the actual extent of the phenomena. The possibility therefore exists that an assessor may exclude regions from his or her model that will be influenced by the phenomenon over the time scales relevant to a safety assessment of radioactive waste disposal sites. The combination of choosing a wrong conceptual model and wrong boundaries will certainly introduce considerable uncertainties in the final results of the safety assessment of a waste disposal site—even lead to meaningless results. Special care should therefore be exercised in choosing appropriate conceptual models in safety assessments of waste disposal sites.

8.4.4 Numerical and Approximation Uncertainty

As mentioned in Section 6.2.5 the governing equations of a conceptual model must be solved either analytically or numerically for the model to be useful. In practice, this means that one must be able to assign specific numbers to the conceptual model. The equations that appear in many of the conceptual models cannot be expressed in terms of simple arithmetic expressions, such as polynomials, with the result that they must be first approximated with simple arithmetic expressions. This will introduce a degree of what will be called *approximation uncertainty* into the final result. Approximation uncertainties can arise from a number of sources such as the truncation of an infinite series. However, the largest source of approximation uncertainty is probably that associated with the discretization of a domain when solving a differential equation through a numerical method such as the finite difference or finite element method.

Approximation uncertainties can be limited considerably by studying the convergence properties of the approximation. Unfortunately, this is often considered as too time consuming and expensive, and therefore simply neglected in many modelling exercises—a luxury that a safety assessor cannot afford.

Computer arithmetic has advanced considerably over the past half century, to such an extent that it is now possible to represent a number with thousands of digits if necessary (Wolfram, 1996). However, it is doubtful if there is enough computer resources in the world to carry out the computations in safety assessments to this accuracy. It will therefore be unwise to completely neglect numerical uncertainty in the safety assessment of radioactive waste disposal systems.

8.4.5 Computer Code Uncertainty

Coding errors and user errors mainly cause uncertainties in computer codes. Coding errors can vary from straightforward blunders to more subtle errors, such as typing I for 1, and the inefficient or deficient implementation of convergence and stability criteria. However, the majority of models used in safety assessments have usually been tested thoroughly for coding errors. User errors are therefore often the major sources of computer code uncertainties, because of the complexities of the models and processes involved in the safety assessment.

8.5 TREATMENT OF FUTURE UNCERTAINTIES

8.5.1 General

It follows from the discussion of the framework for a post-closure safety assessment in Section 6.5 that the formulation and the analysis of well-justified scenarios will form an integral part of such an assessment. According to OECD/NEA (1998), Herman Kahn introduced this method of analysing futuristic events in the 1950s. The method was initially used to study the various options civil defence authorities could take in response to a thermal nuclear war. The method has since become a standard tool in decision and statistical analyses of future events.

The term scenario as defined in Section 6.6.3 and used in this thesis, is associated solely with the anticipated *future* evolution of the disposal system, as opposed to its *normal* evolution. The main purpose of scenario generation in the post-closure safety assessment of a radioactive waste disposal system is thus essentially to use scientifically-informed expert judgement to guide the development of descriptions of the disposal system and its future behaviour. However, it does not try to predict the future; rather, the aim is to identify salient changes, based on analysis of trends, within which variants are explored to investigate the importance of particular sources of uncertainty. The emphasis is therefore on providing meaningful illustrations to assist the decision process (Watts *et al.*, 1999).

The first step in the generation of scenarios for such a system is to identify a comprehensive list of Features, Events and Processes (FEP) associated with the components of the disposal system, described in Chapter 4. Since not all the FEPs in the list will be equally important for a specific site, this list is systematically screened and only those that are significant for the underlying assessment context and system description are retained. The relationships between these FEPs are then investigated in detail and used to establish a list of appropriate scenarios for the site.

New approaches to scenario generation and the uncertainties associated with the future state of a disposal system are developed continuously. Despite differences in approaches and ordering of the steps, the concepts behind the steps are the same for all scenario generation procedures. Before a systematic approach to generate scenario are discussed, a summary will be presented of the work that has been done on the development of a comprehensive FEP list, methods to screen the FEP list, and methods to order the screened FEPs in a structured manner to generate scenarios.

8.5.2 Developing a Comprehensive List of FEPs

The first step in scenario generation is to compile a list of all FEPs related to the specific waste disposal site. Since the compilation of such a list is a creative, multi-disciplinary task, that may depend exclusively on scientific reasoning and expert judgement (Bonano *et al.*, 1990), various attempts have been made to establish an independent, comprehensive list, which can be used to establish site-specific lists. The latest of these lists is the one developed for geological disposal systems by the Working Group on Scenario Generation of the OECD Nuclear Energy Agency (OECD/NEA, 1998). This document also provides a good summary of the experience gained in developing such a FEP list with its associated databases and scenarios.

The Scenario Generation and Justification Working Group of ISAM (see Section 6.1) are currently addressing a similar investigation of FEPs related to near-surface disposal systems. This Working Group adopted the International FEP list of the NEA Working Group and expand it to include FEPs relevant to near-surface disposal systems (IAEA, 1998*b*). The current version of the list, known as the *ISAM International FEP List,* is given in Appendix A. Other sources of information for these systems are the report of Sumerling *et al.* (1989) on scenario generation for the Drigg facility in the UK, the preliminary safety analysis for the Canadian Intrusion Resistant Underground Structure (AECL, 1997), and the assessment of the Greater Confinement Disposal Facility at the Nevada Test Site (Guzowski and Newman, 1993).

8.5.3 Screening of the FEP List

It is simply impossible to consider all FEPs in a comprehensive FEP list, from a practical, technical and economical point of view. Fortunately, some of the FEPs may not be applicable in terms of the assessment context, existing legislation, regulatory requirements, characteristics of the site or disposal concept, or are unlikely to be present at a specific site. The second step in scenario generation is thus to screen the list and discard irrelevant FEPs with the necessary justification. The methods used in such a screening are usually based on (a) expert judgement, (b) probability of occurrence and (c) consequences of the FEPs (Kozak *et al*, 1999). In the so-called *Sandia Scenario Approach,* which stems from the work of Cranwell *et al.* (1990), fault trees are used to represent key aspects of a scenario. The branches of a tree are then assigned probability values that may be propagated to produce an overall probability of occurrence of the scenario. In later versions of the approach (Guzowski, 1990), FEPs are screened based on expert judgement of either the likelihood that they may occur or their consequences. The full scenario of remaining FEPs are then screened, based on either the propagated probabilities of the FEPs or expert judgements of their consequences.

The Sandia Approach will not be discussed further here, because of a number of disadvantages such as: the tendency to generate a large number of scenarios that must be evaluated simultaneously, and its poor ability to address time-dependent variables. The method was, moreover, developed for particular regulations in the US high-level waste programme (Chapman *et al.*, 1995). The probabilities assigned by the method are also primarily derived from expert judgement. Significant resources for elicitation of judgements is, therefore, required when applying the method. There is thus a risk that the results will be biased by these judgements and that they have more mathematical significance than is actually the case (Kozak *et al.*, 1999).

An alternative method for screening FEPs, which will be used in this thesis, was introduced by Eng *et al.* (1994). This method differentiates between FEPs acting *within* the temporal and spatial boundaries of what they call the *Process System* and those that act from *external* the temporal and spatial boundaries onto the Process System—the so-called scenario-generating FEPs (EFEPs). The Process System itself is defined as *the organized assembly of all FEPs required for description of barrier performance and radionuclide behaviour in a repository and its environment, and that can be predicted with at least some degree of determinism from a given set of external conditions* (Eng *et al.*, 1994). This division corresponds very much with the conceptual representation of a disposal system in Figure 4–1. The terms *Process System* and *disposal system* as used in this thesis can therefore be considered as synonyms. A scenario in the Process System approach is then defined by a specific set of external conditions, which will influence the FEPs in the Process System (Eng *et al.*, 1994).

8.5.4 Ordering of FEPs

The next step in scenario generation is to order the screened FEPs into a coherent structure suitable for a consequence analysis. The increasing scrutiny of safety assessments analysis by regulatory bodies, and the public, have led to increasing requirements for scenario generation and justification of the associated models. Three methods have been described in the literature that can be used for the ordering of FEPs: lists and tables, influence diagrams and the Rock Engineering System (RES), matrix approach.

The ordering of FEPs for near-surface disposal systems have usually been limited to lists and tables (Sumerling *et al.*, 1989; AECL, 1997), while the influence diagram and RES matrix approaches have received considerable attention for geological disposal systems (Eng *et al.*, 1995; Skagius and Wiborgh 1994; Skagius *et al.*, 1995; Smith *et al.*, 1996;

BIOMOVS II, 1996, Kozak and Wei Zhou, 1998). Skagius and Wiborgh (1994) who evaluated the use of influence diagrams concluded that the method is very useful in the development of FEP interactions, but that it requires a considerable effort to develop the influence diagrams. They also claim that influence diagrams do not significantly improve the transparency of the FEP process, because it is difficult to use and read the diagrams. The RES matrix method, on the other hand, increases the transparency of the ordering considerably (Kozak and Wei Zhou, 1998), and makes it possible to trace judgements made during model development (Skagius *et al.*, 1995). The method also provides a good basis for interaction between regulators and site developers. This method will consequently be used in this thesis.

8.5.5 A Systematic Approach to Scenario Generation

The integrated approach to scenario generation presented in this thesis considers the ISAM International FEPs list as the primary reference point to ensure that all issues relevant to the assessment are covered. The subsequent screening and categorisation of such a list generates an audit trail, allowing the analyst and regulators to scrutinize the logic of the arguments supporting the definition of models and the calculations effectively.

The first step in this approach is a clear definition of the extent, nature and content of the Process System to be analysed; thereby enabling all potentially relevant FEPs to be categorised either as Process System FEPs or EFEPs for the purpose of the assessment. The choice of EFEPs or scenario-generating FEPs is based on expert judgement, but in a qualitative sense (Kozak *et al.*, 1999), thereby avoiding the difficulties associated with the fault tree analysis of the Sandia Approach. These scenarios are also not intended to be mutually exclusive or comprehensive, since there is no intent to apply probability theory to them. Figure 8-1 gives a broad outline of the Process System FEPs and EFEPs relevant to the disposal system. As can be seen from the figure, the assessment context defines the framework for the identification of all relevant FEPs, while the EFEPs represent more 'global' phenomena, which act as a boundary condition for the disposal system domain. Fundamental to the use of the Process System is the identification of the boundary of the Process System, the important components on either side of that boundary and their interrelationships (Watts *et al.*, 1999).

The Process System itself and its associated FEPs defines the anticipated normal evolution of the disposal system and its environment and provides a necessary central theme in reporting on the radiological performance of the disposal facility. This requires a conceptualisation of the Process System, based on its component features and characteristics, and an appraisal of potential pathways for the released waste to the environment. The Process System FEPs are therefore included implicitly in scenario generation, and does not represent a scenario in itself. With this configuration, the safety performance of the Process System can be assessed independently of the influence of EFEPs. This may be unrealistic as changes in the environment, for example, are neglected. However, it constitutes a useful benchmark against which the significance of other results can be compared and it represents a practical basis for exploring sensitivities to parameters and modelling uncertainties Watts *et al.* (1999).

From a scenario-generation point of view, it is necessary to perform a structured screening and review of all EFEPs, taking into account prescribed screening arguments and the overall assessment context. As an example, the screening argument currently anticipated for the Drigg safety assessment include the following (Watts *et al.*, 1999):

(a) physically implausible given the site context or time scales of concern,

(b) rate or probability small relative to other EFEPs,

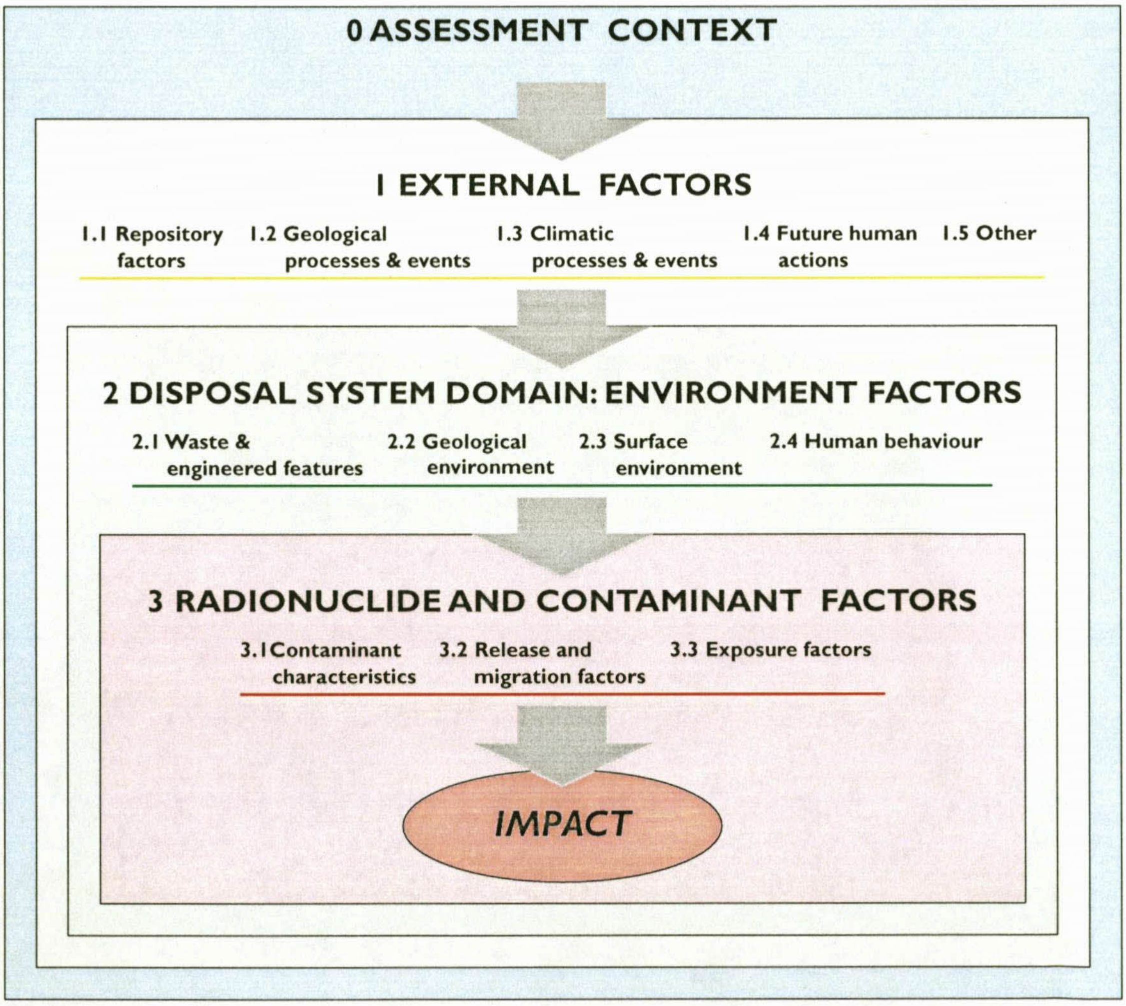


Figure 8-1 Outline of the ISAM international list of features, events and processes relevant for near-surface disposal systems (IAEA, 1998*b*).

(c) global disaster,

(d) included elsewhere, and

(e) excluded by assessment context or regulatory guidance.

This screening will serve as a basis to distinguish those external factors that represent a significant influence on the long-term evolution of the disposal system. The EFEPs are therefore included explicitly in scenario generation.

The Process System FEPs and screened EFEPs are next used to develop and justify alternative conceptualisations of the Process System, including appraisals of the potential pathways for the released waste to the environment. For each of the conceptualisations, outline scenario descriptions (alternative futures) are also derived from a consideration of the interactions between the Process System FEPs and the EFEPs system. A structured representation of the interactions will provide an audit trail and facilitate conceptual model development. As discussed in Section 8.5.4, this is best achieved using the RES matrix approach.

The basic device used in the RES matrix approach is the interaction matrix, in which the main variables or parameters are identified and listed along the leading diagonal of a square matrix. The interactions between the parameters occur in the off-diagonal terms. A simple example of a 2x2 matrix is illustrated in Figure 8-2 with the concrete container and backfill

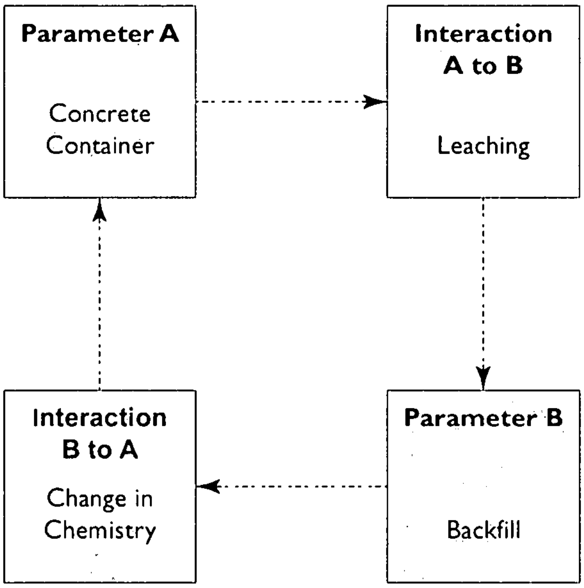


Figure 8-2 A simple 2x2 interaction matrix illustrating the application of the RES matrix method to interactions between features, events and processes in a radioactive safety assessment.

as diagonal elements. Leaching represents an interaction between the container and backfill, which in turn will introduce a change in the chemistry of the backfill that could influence the concrete container again. This defines a clockwise convention for the influence direction. The matrix can of course be easily extended to an nxn matrix, for a system with n main parameters.

In the Process System approach, the components of the disposal system can also be included in the interaction matrix, and analysed in greater detail by creating one ore more sub-matrices, as shown in Figure 8-3. This suggests that the elements on the main diagonal can be represented by a specific theme, such as the migration pathway of radionuclides from the waste to humans. The off-diagonal elements represent the interactions of FEPs

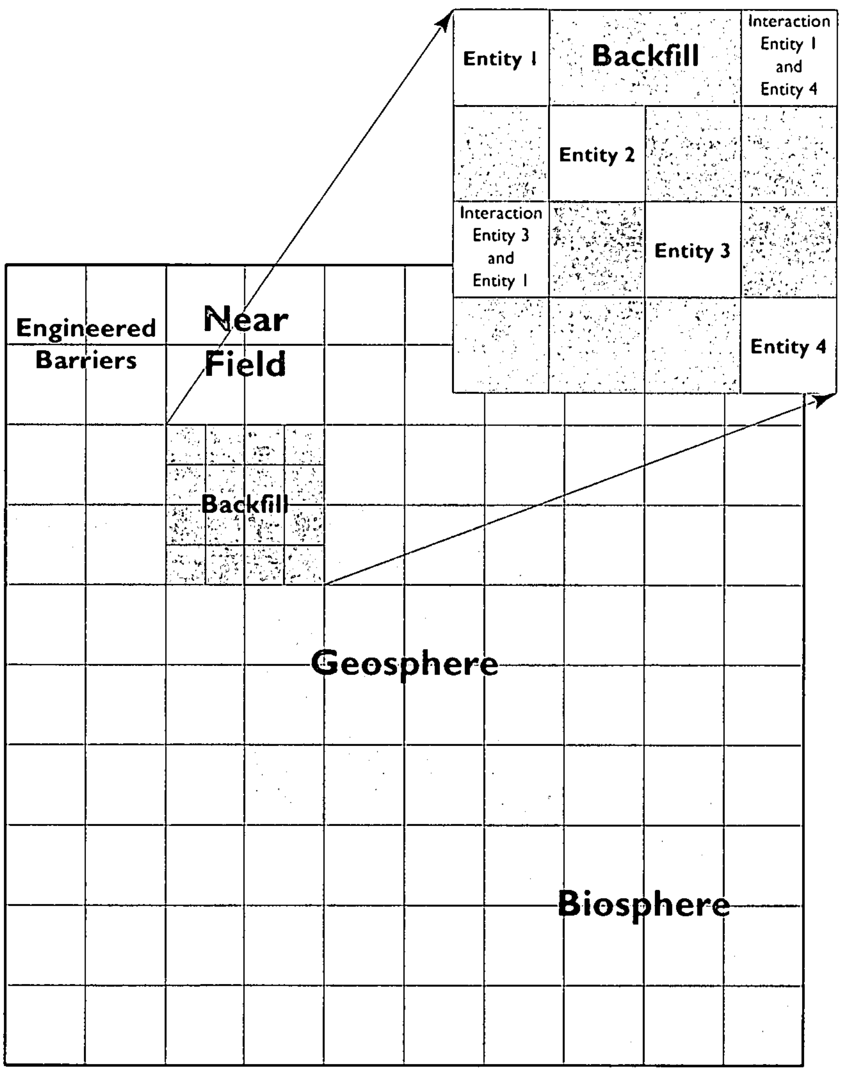


Figure 8-3 A conceptual interaction and secondary interaction matrix for the Process System approach with the components of the disposal system included explicitly.

that cause or influence the migration of the radionuclides from one diagonal element to another along the identified pathway. Those above the diagonal represent the influence on forward movement, while those below influence the backward movement. This is illustrated in Figure 8-4, which represents a 5x5 matrix and the potential migration pathway of radionuclides from element D, through various interactions between diagonal and off-diagonal elements, to element E.

It is important to note that there is no rigorous way of using the RES matrix. It is merely an auditing tool to order FEPs in a structured and traceable manner to facilitate scenario generation and conceptual model development.

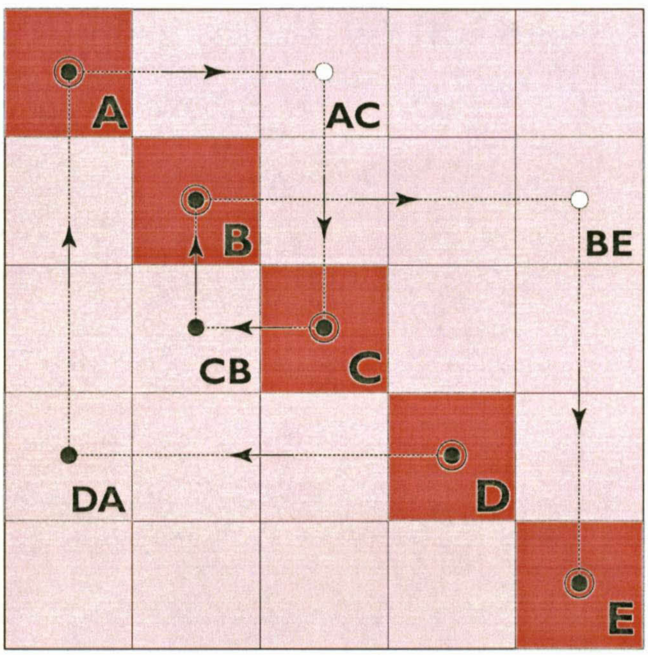


Figure 8-4 Principle of a migration pathway of radionuclides through the interaction matrix.

8.5.6 Model Development for Consequence Analysis

By following the migration pathway of radionuclides in Figure 8-4, one can identify all the processes, interactions and phenomena that may influence the migration and must be accounted for with heuristic or phenomenological mathematical models. Some of the processes can be coupled and included in one mathematical model, such as the moisture movement through the concrete barriers, back-fill of the near-field, and the unsaturated zone. However, the degradation of the concrete structure will require a different mathematical model.

Although it will probably be rarely necessary, new mathematical models have sometimes to be developed for features that could influence the decision adversely, but is not described adequately by existing models, or if no model exists for the feature.

Once all mathematical models needed to represent the migration of radionuclides along a chosen pathway are identified, the information from the conceptualisation of the Process System can be used to develop a conceptual model for the migration. This include, for example, the domain over which the mathematical model is to be applied, initial and boundary conditions and any constitutive parameters required to solve the mathematical model. This defines the conceptual model for the scenario.

8.6 TREATMENT OF DATA AND PARAMETER UNCERTAINTY

Uncertainties in the quality and accuracy of data and parameters can often be considerably reduced in safety assessments, by applying suitable quality assurance procedures during the collection and manipulation of the data. However, many of the data required in a post-closure safety assessment are not known, or will ever be known, by the assessor. It is therefore impossible to eliminate data and parameter uncertainties completely from safety assessments.

There are essentially two types of methods that can be used to study the influence of data and parameter uncertainties in safety assessments—*deterministic* methods and *probabilistic* methods. In deterministic methods a set of estimated values of a parameter is used to evaluate the impact that the parameter may have on the final results, particularly the regulatory decision.

Probabilistic methods, such as the Monte Carlo method, represent a parameter through a probability density function (pdf), or the mean and variance of such a distribution. These values are then used in a series of analyses to arrive at a probability that the regulatory decision will be achieved or not. It stands to reason that these pdfs must be carefully constructed if the uncertainty estimates for the model predictions are to be meaningful. In classical statistics the pdfs are usually derived from known values of the parameter or data. In practice this usually means that a large set of parameter or data values must be known, before one can obtain estimates of the mean and variance of the data set and confirm the pdf by the testing it with various hypotheses. A common approach in classical statistics is therefore to use whatever data is available at a specific time and recalculate the pdf whenever additional data becomes available, thereby increasing confidence in the pdf (Freeze *et al.*, 1990). Such an approach may not always be acceptable in the safety assessment of a waste disposal.

The pdfs used in safety assessments are consequently often based on the Bayesian method.

To describe this method, let **A** and **B** be two sets of random variables each with its own so-called prior probability functions, say P(**A**) and P(**B**). In multivariate analysis it is sometimes necessary to know what the distribution of **A** will be given, a set of observations on **B**. Let P(**A**|**B**) denote this so-called *conditional* or *posterior* probability function of **A**. By using Bayes's theorem (Groebner and Shannon, 1989), this posterior probability function can be expressed, in the case where **A** and **B** are discrete variables, as

$$P(\mathbf{A}_i \mid \mathbf{B}) = \frac{P(\mathbf{B} \mid \mathbf{A}_i)P(\mathbf{A}_i)}{\sum_j P(\mathbf{B} \mid \mathbf{A}_j)P(\mathbf{A}_j)}$$

where it is assumed that both P(**A**) and P(**B**|**A**) are known. The importance of this theorem is that it allows one to derive a probability distribution for the variable **A**, even when very little information is available on **A**, provided that P(**B**|**A**) is known. It is therefore sometimes possible to derive a pdf for a given variable, even before any measurements are taken of the variable through Bayes's theorem (Freeze *et al.*, 1990). Such a pdf can consequently differ considerably from the classical estimate when the data are sparse.

Confidence in a pdf derived with the Bayesian method can of course also be increased by recalculating it every time more data become available, relying less on the conditional information. The pdf should under these conditions converge towards the classical estimate. However, there may not always be enough time to establish that the pdf will represent the behaviour of the parameter far into the future. Nevertheless, there are situations where the conditional information may be sufficient to derive a pdf with confidence. This is, for example, the case of groundwater levels in open boreholes that tend to follow the topography of an area quite closely, as long as the aquifer is not pumped excessively (Van Sandwyk *et al.*, 1992).

The deterministic and probabilistic methods each have certain advantages and disadvantages, as far as the implementation of safety assessment methodologies is concerned. The biggest advantage of the deterministic approach is its simplicity of implementation and transparency of interpretation. However, deterministic variations in the parameters will not necessarily yield bounds on the final result. The approach therefore does not lend itself to the development of a clear decision criterion that can be used to determine whether the facility complies with the regulatory decision or not (Kozak, 1997).

Although the probabilistic method can provide a criterion for the regulatory decision, the criterion need not necessarily yield a statistically rigorous estimate of site's performance, because of the uncertainties involved in deriving the necessary pdfs. All that the method really can do is to provide the analyst with quantitative information about the relative likelihood of various estimates of the performance. Kozak *et al.* (1993) suggests that this indeterminacy can be reduced in the case of the Monte Carlo method, by coupling the method with a stratified parameter sampling strategy, such as the Latin Hypercube Sampling method (Iman and Shortencarier, 1984). Since it is already difficult to interpret probabilistic methods and to communicate the results to non-expert audiences (Kozak 1997), this extension may not always be acceptable. Another argument often raised against probabilistic methods is that they require a considerable computational effort and resources, although this argument is loosing ground with the rapid increase in computing power in recent years.

In summary it can be said that neither the deterministic nor the probabilistic method has special advantages as far as the reduction of parameter uncertainty in the safety assessment of radioactive waste disposal sites is concerned. A more pragmatic approach—use the method that suites a particular site or pathway the best—would therefore seems to be more practical.

8.7 TREATMENT OF MODEL UNCERTAINTY

There do not exist specific guidelines to study model uncertainties, similar to those for parameter and data uncertainties, except for the *verification* and *validation* of the conceptual model. Otherwise one has mainly to rely on strict quality control over the calculations and the application of various assurance procedures.

The first uncertainty that may arise in the application of models is to what extent does the mathematical model, on which the conceptual model for the phenomenon to be modelled is based, represent the physical processes responsible for the phenomenon. The mathematical models normally used in safety assessments have usually been tested thoroughly for this. One can therefore safely neglect mathematical model uncertainties in safety assessments, when comparing it with the other uncertainties in the assessment.

The main contribution to model uncertainty comes from the ability of the chosen conceptual model to represent the phenomenon. As discussed in Section 6.2.4, the conceptual model represents the assumptions made to describe a specific realization of a natural phenomenon. These assumptions represent, amongst others, the domain of interest, the dimensionality of the problem and the initial and boundary conditions that may be needed to solve any differential equations that appear in the conceptual model. Very little work has been done to treat conceptual model uncertainty (Kozak *et al.*, 1993). One approach is to use a set of conceptual models that represents different types of boundary conditions, the spatial and temporal extent of the domain, may be even different processes, and compare that with site-specific data. This approach is commonly referred to as the *verification* and *validation* of the conceptual model.

Model verification can be described as the process of showing that the chosen conceptual model and the analytical or numerical method used to solve it behave as intended. In other words, the model must be able to reproduce observations of the phenomenon for which it was developed. Model validation, on the other hand, is a process to show that the model is able to describe observations of the phenomenon not used in its verification. Unfortunately, it is not always easy to determine what component of the conceptual model is incompatible with the phenomenon if the process fails.

Uncertainties associated with the numerical methods, used in the implementation of the conceptual model, can be studied by comparing the results with an analytical solution of a similar model, or by studying the convergence properties of the model.

CHAPTER 9

A DECISION ANALYSIS FRAMEWORK FOR POST-CLOSURE SAFETY ASSESSMENTS

9.1 INTRODUCTION

The post-closure safety assessment of a radioactive waste repository was described in Chapter 6 as a decision tool to determine the conditions under which compliance with safety objectives can be reasonably assured. Such a safety assessment therefore very much resembles system analysis, as developed in the operational research literature (Kozak, 1994), at least in principle. However, as shown by the discussion in Chapters 7 and 8, a post-closure safety assessment differs in one important aspect from a classical system analysis—the members of the system are subject to large uncertainties. No attempt has therefore been made to date to formally include system analysis theory into the post-closure safety assessment of a radioactive waste disposal site, nor is it the purpose of this thesis to do so. What will be done here is to present a decision analysis framework that will aid in making feasible decisions in such a safety assessment.

The decision analysis approach that will be followed here is based on a revision of the hydrogeological decision analysis framework proposed by Freeze *et al.* (1990), which is particularly suited for systems with large uncertainties and high risks. The basic philosophy behind the methodology is that the risk that an engineering design will fail to meet design objectives depends on uncertainties in the technical analysis. However, decision-makers often base their decisions more on economic than technical factors in such analyses. It therefore makes sense to also introduce the economic factor into the decision framework. Indeed, this was one of the reasons why Freeze *et al.* (1990), introduced their decision analysis framework.

The importance of the economic factor from an integrated radioactive waste management perspective was introduced in Section 3.6.3. As used there, nuclear liabilities management refers to both the *reduction* and *discharge* of nuclear liabilities. It is impossible to include a full description of the framework here. However, the interested reader can find full details in the references cited in Section 9.2, where the development and application of the methodology is reviewed briefly. This is followed by a discussion of the basic principles of decision analysis in Section 9.3. The framework proposed for the post-closure assessment of a radioactive waste repository is discussed in Section 9.4.

9.2 REVIEW OF THE DECISION ANALYSIS FRAMEWORK

Massmann and Freeze (1987*a*, 1987*b*) introduced the basic philosophy behind a decision analysis framework in two papers on hydrogeological projects. The first paper describes the methodology for the risk-cost-benefit analysis in the design of new waste management facilities. The second paper applies the methodology to assess alternative design strategies for such a facility, and how a regulatory agency can use the analysis to assess alternative regulatory policies. This was followed by a comprehensive series of four papers that give excellent discussions of risk-cost-benefit analysis. These pioneering papers laid the foundation for the application of a decision analysis framework for hydrogeological projects.

The series of papers begin with a detailed description of the framework as a tool in hydrogeology (Freeze *et al.*, 1990) and the application of the framework to two case studies in groundwater contamination (Massmann *et al.*, 1991). The first case study involves the selection of an optimal pumping rate for an extraction well to capture an existing contamination plume, whereas the second evaluates the design of a leachate collection system for a soil remediation facility. In the third paper (Sperling *et al.*, 1992), the framework was used to demonstrate its usefulness in solving geo-technical problems associated with the design of a groundwater control system at an open pit mine. The fourth paper (Freeze *et al.*, 1992) provides an introduction to the concepts of data worth analysis in reducing the uncertainty, and the use of Bayesian statistics to provide a criterion when to stop the collection of data in iterative site investigation programs. One of the key advantages of the methodology is its suitability to aid in the design of site investigation programs and monitoring networks and to assess the potential financial worth of additional data.

There are essentially two questions that are very important in many hydrogeological investigations that can be answered by such a decision analysis (Freeze *et al.*, 1992). The first is:

> *"Given a set of data on the hydrogeological parameters, which design alternative from an available suite of design alternatives is the best one?"*

and the second

> *"Is it worthwhile trying to improve the data set by taking additional measurements, before deciding which is the best design alternative?"*

The decision analysis framework has been applied to a large number of hydrogeological investigations since the publication of the four papers. For example, James *et al.* (1996*a*) use the framework to determine what is the best remedial action for a contaminated aquifer, what are the most important parameters to sample in such an investigation, and to what extent will the collection of additional data be cost effective. James *et al.* (1996*b*) further show that such an analysis can be a useful tool to help site managers to effectively allocate the limited remediation funds and resources in cleaning the large number of contaminated sites throughout the United States.

Jardine *et al.* (1996) apply the method to the design of a monitoring network at a waste management facility, overlying fractured bedrock, with the objective to detect contaminants before they reach a regulatory compliance boundary. The same approach was also used by Bugai *et al.* (1996) to evaluate alternative management strategies for the water wells at Pripyat Town, where the groundwater is contaminated with ^{90}Sr from the Chernobyl accident.

Janse van Rensburg (1992) introduced an interesting variation of the framework in his analysis of the design of a well field, for the supply of water to a diamond mine in the Northern Province of South Africa. His analysis not only takes uncertainties in the hydrological parameters into account, but also uncertainties associated with the financing of the scheme.

9.3 BASIC PRINCIPLES OF A DECISION ANALYSIS

The one essential element of a *decision*, as defined by Kirkwood (1997) is the existence of *alternatives*. That is, one must have at least two different options of which only one can be selected. There is a problem if only one option exists, but this is not a decision problem. Most significant decisions involve situations where the various options lead to different *consequences* or *outcomes*.

The models in safety assessment methodologies usually use only localised information to describe the response of the environment to a specific option. They may therefore not be able to identify an optimal management strategy if the same objective can be achieved with more that one option. This can only be achieved through a thorough study of all alternatives, requirements and constraints of the management strategy. This is the analysis referred to as system analysis above. The analysis, which requires a clearly defined objective, can be carried out in either an *optimisation* framework or a *decision analysis* framework.

In the optimisation framework the main aim of the analysis is to determine optimal values for a given set of decision variables in a system, subject to a specified objective function and a set of constraints. It is a general approach that will determine the best alternative from a given set of alternatives, subject to the given objective function. Unfortunately, it is not always easy to determine the global extreme of a non-linear function. The approach is consequently mainly restricted to problems with a linear objective function and constraints, in practice. The approach, moreover, does not lend itself readily to a statistical interpretation, with the result that it is not suitable for the analysis of risks or statistical variations in the data, constraints and objectives (Freeze *et al.*, 1990).

Decision analysis, described by Massmann and Freeze (1987*a*) as a formalisation of common sense for decision problems that are too complex to solve with informal common sense, does not suffer from such limitations. Although the approach is less general than optimisation, and therefore cannot provide an optimal solution across all the system variables, it is quite useful in eliminating alternatives that do not satisfy prescribed constraints from a system.

9.4 DECISION ANALYSIS FRAMEWORKS

9.4.1 The Framework of Freeze

The original decision framework of Freeze *et al.* (1990) has six components, shown in the blue boxes of Figure 9-1, to which Janse van Rensburg (1992) added the hydrological and financial components, shown in the orange boxes.

The purpose of the parameter, financial and geological uncertainty models is to study uncertainties in these entities, as revealed by data obtained in the Field and Data Acquisition Program. The uncertainties in the hydrological and hydrogeological parameters will usually be based on actual field observations, while the financial uncertainties will depend on variations in the financial markets. The analyses of these uncertainties should not present any profound difficulties in practice *per se*, since there exists a number of well-known models that can be used for this purpose (Freeze *et al.*, 1990; Janse van Rensburg, 1992). A similar situation also exists in the case of the engineering reliability model, where the properties of most engineering materials are well-known from laboratory studies and practical experience. The same can, unfortunately, not be said of the geological uncertainties.

As used here, the term geological uncertainties refers to all quantities related to what may be termed the *subsurface geometry*. This includes, for example quantities such as, discontinuities in the geology, the presence (absence) of a fluid conductive fracture networks and planes, variations in the lithology and the tectonic and structural stability of the site. These quantities are usually neglected in the majority of hydrogeological investigations, with the result that their influence on subsurface flow is not well documented in the literature.

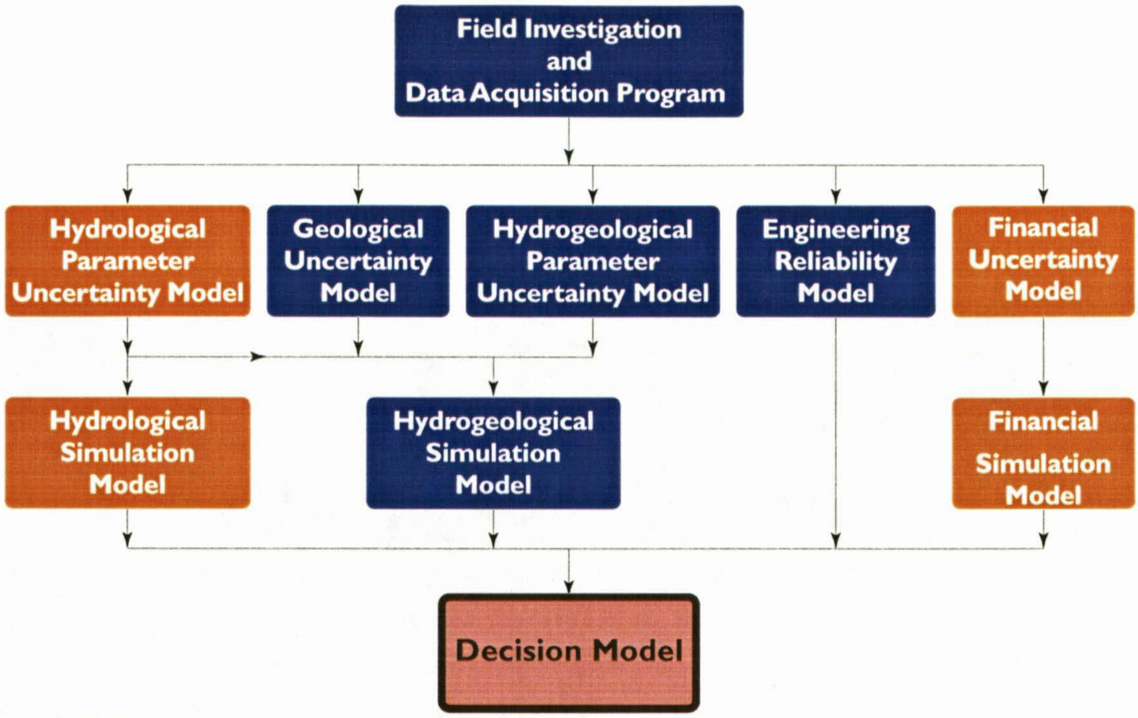


Figure 9-1 Framework for hydrogeological decision analysis as proposed by Freeze *et al.* (1990) and modified by Janse van Rensburg (1992).

The establishment of a suitable geological uncertainty model may, therefore, present the analyst with his or her biggest challenge in developing a decision analysis framework, particularly one devised for the post-closure assessment of a radioactive waste disposal site.

The main purpose of the various simulation models in the framework is to provide an estimate of the behaviour of the particular component at a time t, so that the analyst can decide whether the system still behaves within the prescribed constraints. The hydrological and financial simulations are usually based on an existing stochastic model in these fields, while the geohydrological simulations are based on either a numerical or analytical solution of the flow and mass transport equations in Chapter 6. However, the model must then be applied stochastically and not deterministically (Janse van Rensburg 1992), as is usually the case in groundwater modelling.

9.4.2 The Disposal System Framework

A comparison of the framework in Figure 9-1 and the discussion of the uncertainties associated with the post-closure safety assessment of a radioactive waste disposal site as discussed in Chapter 8, shows that the framework contains many of the uncertainties. This suggests that it may be advantageous to use a similar framework in the post-closure assessment of a radioactive waste disposal site.

The ideal approach to design such a framework would have been to use existing data on the post-closure safety assessment of a radioactive waste disposal site. Unfortunately, the idea of a post-closure safety assessment has only been introduced very recently, as can be seen from the discussion in Chapter 6. Indeed, the methodology itself is still in the development phase. There is therefore not enough data (at least in South Africa) to use in designing such a framework. However, a detailed review of the discussion in Chapters 7 and 8 suggest that the one presented in Figure 9-2 may be quite suitable for this purpose.

As can be seen from Figure 9-2 all the components of the framework in Figure 9-1 are also present in the new framework, albeit with different or additional functions. The engineering reliability model, for example, now combines with the near-field uncertainties to determine uncertainties in the source term, while hydrological uncertainties now affect the source term, geosphere and biosphere. The geosphere uncertainty and simulation models will probably be similar to the geohydrological uncertainty and simulation models in Figure 9-1, except that the geosphere models will always have to include mass transport.

The new components introduced in Figure 9-2 include the near field uncertainty and simu-

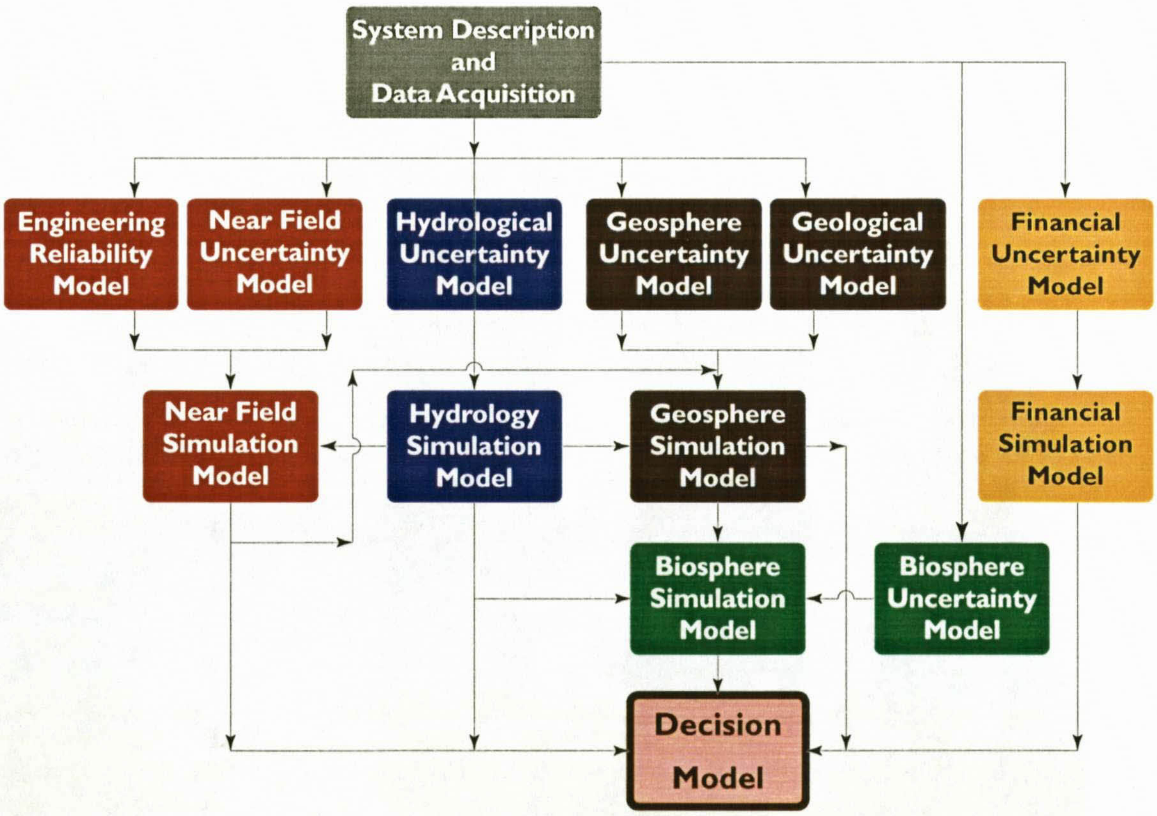


Figure 9-2 A possible decision analysis framework for the post-closure safety assessment of a radioactive waste disposal site.

lation models, the source term simulation model, and the biosphere uncertainty and simulation models. The near field can be handled with models very similar to that used for the geosphere, adjusted appropriately, but special models will have to be used for the source term and biosphere models.

An important prerequisite for the application of a decision analysis framework is that the various uncertainties must be known with a sufficient accuracy, so as not to skew the final results. The common approach in classical statistics is to assume that the only pertinent information about the uncertainty of a given sample of data is contained in the sample. However, this is rarely the case in the application of a decision analysis framework where the number of elements in the sample is often very small. Moreover, decision-makers feel that the sample is not the only source of information on the data, and that subjective forms of information, such as an expert opinion or personal experience, should be included in the decision process. As pointed out in Section 8.6, this can be achieved by using Bayesian statistics instead of the classical statistics. A decision analysis based on Bayesian statistics is known as Bayesian decision analysis (Groebner and Shannon, 1989). A good summary of previous applications of Bayesian decision analysis in geohydrology and hydrology can be found in James and Gorelick (1994).

9.4.3 The Decision Model

The basic aim of the decision model is to give the analyst an indication whether a given system will satisfy the objectives of a person who wants to implement the system. It usually consists of a scalar function that measures the relative effectiveness of different alternatives in achieving the specified objectives of the system. It is therefore comparable with the objectives of a post-closure safety assessment, discussed in Chapter 6.

It follows from the previous brief description that the decision model selected for a specific study will very much depend on the purpose of the study and whether it should be applied in an optimization sense or a decision sense. The type of constraint(s) placed on the system will also be very important. For example, the operator of a water supply scheme may decide to change some of the constraints, even though it may affect his objective adversely—an approach that will not be possible if the objective is prescribed by a regulatory body.

A common approach used in the case where the system consists of a facility devised for economic production purposes is to use an economic objective. In other words the facility must be technically designed and operated in such a way that it maximizes the profit (or minimizes the loss) of the owner. Let j be a member of sequence, **J**, of processes from which the owner or operator of the facility may benefit. Let $B_j(t)$, $C_j(t)$ and $R_j(t)$ denote this benefit and its associated cost and risk, respectively, at a time t during the lifespan, or other time horizon, of the facility. The objective function for j can then be taken as the net present value (NPV) of j taken over the chosen time horizon, discounted at the market interest rate (Crouch and Wilson, 1982), and expressed mathematically as

$$\Phi_j = \sum_0^T \frac{1}{(1+I)^t}[B_j(t) - C_j(t) - R_j(t)] \tag{9.1}$$

where T is the time horizon (typically expressed in years) and I the market discount rate (a decimal fraction). Notice that the three parameters in the square brackets, on the right hand side of Equation (9.1) have to be expressed in monetary values. If the decision model is applied to a radioactive waste management problem, then it is clear from Section 3.6.2 that $C_j(t)$ refers to the cost associated with the management strategy and by implication, contribute to nuclear liabilities. The risk, $R_j(t)$, in Equation (9.1) is often related to the expected cost, $C_j^f(t)$, associated with the probability, $P_j^f(t)$, that the j-th member may fail in the year t, and expressed as :

$$R_j(t) = \gamma_j P_j^f(t) C_j^f(t) \tag{9.2}$$

where γ_j is a suitable weighting factor, introduced to account for the so-called risk-averseness of many decision-makers. Owners and operators of facilities which do not have a strong financial backing, may decide to choose $\gamma_j > 1$, while large companies more likely will take a neutral approach and choose $\gamma_j = 1$. Some owners and operators may even want to use a value of γ_j that depends on $C_j^f(t)$. Notice that all the terms in Equations (9.1) and (9.2) are derived from economical considerations, except for P_j^f, which will depend on the actual behaviour of the j-th process.

It follows from the definition of the risk term in Equation (9.2) that this term will only occur if the process j fails. However, notice that the term failure is used here to denote a situation where the process fails to comply with certain criteria laid down for the decision analysis, and not in its more common sense. For example, a failure may occur in a groundwater-monitoring network if the concentration of a pollutant rises above an expected or regulatory limit in one of the observation boreholes, the position of which may also be subject to some form of control.

The cost, $C_j^f(t)$, above will only arise in the event that the process j fails. This may typically include fines, taxes, and charges levied by a regulatory agency for failure to meet regulatory standards, the cost of litigation should any arise, and the revenue foregone if the operations must be curtailed or stopped, because of the failure. The cost incurred in any remedial action that may be necessary after such a failure should also be included in $C_j^f(t)$ From a radioactive waste management perspective, it is clear that these costs amount to another source of nuclear liabilities that needs to be accounted for, as described in Section 3.6.2. It is therefore often important to know exactly what caused a failure. In the case of a technical facility this can often be achieved by studying a model of the process. This is the reason for the inclusion of the uncertainty and simulation models in the decision analysis frameworks, presented in Figure 9-1 and Figure 9-2.

The time or planning horizon defines the period over which the analysis has to be performed, in other words, over which time period the system poses a risk to future generations. The pertinent discount rate is the market interest rate on borrowed money.

It is not difficult to show that the NPV value of future money approaches zero not very far into the future (approximately 50 years in the case of a discount rate of 10%). The discount rate and the time horizon are therefore not really independent in an NPV analysis. This has obviously serious implications for the application of decision analysis in the post-closure safety assessment of a radioactive waste disposal site, where the time frame of interest may involve hundreds to thousands, even millions, of years. In this case it makes more sense to compound the NPV at the inflation rate, rather than discount it at the interest rate. This yields an objective function of the form (Gitman, 1988)

$$\Phi_j = \sum_0^T [B_j(t) - C_j(t) - R_j(t)](1+I)^t \tag{9.3}$$

where I now represents the inflation rate.

To conclude this discussion of the decision model, it must be pointed out that the processes or decision variables, as they are also called, in a decision analysis will vary from one analysis to another. For example, in siting a new waste disposal facility where there is more than one site to choose from, the sites themselves will be selected as decision variables, while different designs will be the obvious candidates in designing a disposal facility for the containment of waste. In designing a monitoring scheme, the monitoring positions, sampling protocol and sampling frequency will all be suitable candidates for decision variables, or in a remediation scheme the position and discharge rates of the pumping and injection wells.

There is little doubt that a decision analysis approach can make a significant contribution to the management of nuclear liabilities, in particular to the reduction of nuclear liabilities, if applied correctly. This statement is perhaps best illustrated through the following hypothetical example.

During the planning of a new radioactive waste repository for a specific type of waste, a liability study with Equation (9.1) or (9.3), based on economic constraints alone, has shown that a certain type of container and certain construction materials will yield the minimum liability. However, a post-closure assessment of the repository's near-field revealed that the interaction between the construction materials of the repository and the immediate geosphere may create an environment that will corrode the selected container faster than its design specifications. The possibility therefore exists that the repository could have been constructed and even commissioned, if these two studies were performed separately and not related to each other, with disastrous consequences.

9.4.4 Data Worth Analysis and Nuclear Liabilities

If the flow diagram of the safety assessment approach shown in Figure 6-1 are followed for at least one iteration, the safety analyst will come to a point where he or she have to decide whether the safety case was adequately demonstrated or not. In the latter case, a further iteration of the safety assessment may be needed, which means that more information must be generated, according to Figure 6–1. Such an exercise will clearly add to the nuclear liabilities, even if the investigation is restricted to those parameters and assumptions that have the greatest impact on the system performance. However, there is no assurance that the additional data will reduce the uncertainty in the dose to acceptable levels. It would therefore be advantageous if the safety analyst can determine before hand if the additional data will reduce the uncertainty in the dose sufficiently to justify the additional cost. One approach to solve the previous problem is to use a so-called data worth analysis as described in Freeze *et al.* (1992), James and Freeze (1993), James *et al.* (1996*a*) and James *et al.* (1996*b*).

There are two ways in which the worth of additional information can be estimated. The first is to derive an expected value that perfect information will have on the safety assessment—the so-called EVPI approach. This approach gives an estimate of the maximum impovement that could be expected if the site conditions were known. The approach is consequently usually used in conjunction with the second appoach, known as the expected value of sample information (EVSI) (James and Gorelick, 1994). This approach is based on the assumption that the uncertainty in a sample of numbers decreases as the sample increases.

CHAPTER 10

CONCLUSIONS AND RECOMMENDATIONS

10.1 INTRODUCTION

The main objective of radioactive waste management and its underlying principles is to ensure that human health and the environment are protected now and in future, without imposing an undue burden on future generations (IAEA, 1995*a*). This implies that the impact the disposed waste will have on individuals and their environment must be determined as a function of time, before any long-term management strategy of the disposed radioactive waste can be implemented. This procedure is generally referred to as a post-closure safety assessment. This term is a relatively new concept in waste disposal, introduced by the IAEA around 1981 (IAEA, 1981) in an attempt to provide guidelines for the solution of one of the most vexing problems in the world—the disposal of radioactive waste.

South Africa produces radioactive waste through the full nuclear fuel cycle and the application of radioactive materials in industry, research and medicine. There are two sites where the waste from these activities are disposed at the moment—Vaalputs in the Northern Cape Province and Thabana at Pelindaba near Pretoria. Vaalputs was selected after a detailed site selection and evaluation program, but very little attention was paid to the post-closure safety of the facility. Although it is known that Thabana was selected for 'the permanent storage of radioactive waste', there is no indication that the site was ever subjected to a detailed site selection or evaluation program. Indeed, all indications are that the site was merely selected because of its proximity to Pelindaba, where most of the South African research activities related to the nuclear industry were concentrated shortly after 1960. The result is that huge uncertainties exist about the inventory of the waste disposed at the site and its future impact on people and the environment.

One reason for the present situation in South Africa is that not much thought was given to the impact that the disposal of radioactive waste may have on future generations. However, there is a growing need to find a long-term solution for the disposal of spent fuel produced by the AEC and ESKOM, in a manner that will ensure the safety of the public, and in accordance with modern, internationally accepted, safety practices. Although some of the other African countries are involved in activities related to the nuclear fuel cycle (e.g. neutron activation or TRIGA type reactors), many of them posess spent nuclear sources. This thesis describes a structured methodology that should contribute significantly to the solution of both problems, if implemented judiciously and conscientiously, and lay a sound foundation for the long-term disposal of radioactive waste in the countries.

The proposed methodology—an integrated and structured approach to the post-closure assessment of a radioactive disposal system—is to:

(a) ensure the safety of the present public and future generations,

(b) enhance the public acceptance of the methodology,

(c) keep the expenditure associated with the implementation of the methodology at a minimum,

The methodology recognises the interdependence between operational phase activities and the post-closure behaviour of the disposal system. It is an iterative process that considers site-specific, prospective evaluations of the post-closure phase.

The implementation of a post-closure safety assessment is a highly technical exercise, but also contains significant social and political elements. Many of the elements, especially the technical elements, will not be known at the beginning of the assessment. Moreover, it is highly unlikely that a disposal concept can be introduced without the active participation of the population. It is therefore of the utmost importance that the assessment should be performed in conjunction with a research and development program aimed at reducing the unknowns and promoting public awareness of the assessment.

Safety assessments of radioactive waste disposal sites have historically often been interpreted as an attempt to *predict* the future transport of radionuclides through the geosphere and biosphere of the site, until ingested by a person. Many countries have consequently built and developed surface and subsurface research laboratories to investigate this path in detail (Kickmaier and Mckinley, 1996). However, the view expressed in this thesis is that it will be difficult (if not impossible) to achieve such an objective. The proposed methodology is therefore not aimed at satisfying the historical objective, but rather to convince all interested parties that the disposal system will satisfy all regulatory and other constraints within reasonable limits. This objective can be achieved with considerable less effort and a corresponding reduction in research and development costs and time.

10.2 CONCLUSIONS

The development of any, and not only radioactive waste disposal, systems will always cost money. Some of the costs will be clearly identifiable, such as those associated with the design and development of the site, installation of the necessary infrastructure and the operational costs, but other are hidden. This is for example the case with liabilities that may arise should the system fail in future, thereby causing damages to other people's properties and lives. Although the origins of the two types of costs differ, they are still costs that have to be met at some point in time. Moreover, these costs are not necessarily independent from each other. For example, the failure of the system could probably have been prevented if more attention was paid to its design and construction. The view taken here is therefore not to consider the costs separately, but simultaneously in what is known as nuclear liabilities. This has the advantage that one can use a decision analysis framework to manage all the costs effectively. Since the costs associated with the planning and development of a radioactive waste disposal system can be huge, it is essential that no post-closure assessment of such a system should be undertaken without a decision analysis framework for the nuclear liabilities. The advantage of the approach is that it allows one to link today's actions with tomorrow's consequences within the limits of the prescribed regulatory and possibly public constraints.

The first step to take in a post-closure safety analysis is to identify the type and nature of the waste that must be disposed and then to select a suitable site. For example, the geology at some of the prospective sites may support the disposal of low and intermediate level radioactive waste in a near-surface disposal system, but not the disposal of high-level waste in a geological disposal system.

There will probably not be very many sites that are technically suitable and acceptable to the public for disposal of radioactive waste. Nevertheless, one of the sites has to be selected as the disposal site at some point in time. One approach to solve this problem is to carry out a site selection, as described in Section 4.3.2 and use this information in the decision analysis framework, discussed in Chapter 9, with the sites themselves as decision variables. Although the costs of these investigations will add to the nuclear liabilities associated with the future acquisition and development of the chosen site, this will be more than accounted for in the later stages of the post-closure assessment analysis. Moreover, a further analysis of the results may already indicate that there is a possibility the site may fail in the future and to what extent the failure may add to the nuclear liabilities of the site. In such cases, including an estimate of these liabilities in the decision analysis framework can further enhance the site selection procedure.

Although the geological properties will play a very prominent role in site selection, one should never neglect the type of waste. For example, it is commonly accepted that clayey formations are ideal for the near-surface disposal of low- and intermediate level wastes, because of their ability to retard the motion of the nuclides. While this may be true for most of the types of radioactive materials, this is certainly not true in the case of radioactive waste that consists mainly of carbon compounds. It is known that carbon compounds can increase the hydraulic conductivity of clays by a factor of 10^4 and more (Quigley and Fernandez, 1989), thereby almost nullifying their retention properties. Since this property of carbon compounds is well-known, it makes sense to include it already in the site selection. However, other compounds may show a similar behaviour, but on a lesser scale and may not be detected before a comprehensive site characterization, described in Section 4.3.3, is completed. This suggests that the waste acceptance criteria, discussed in Section 3.3.3, be included in the site characterization. This will allow one to determine the inventory and concentration limits of the waste that can be safely disposed in the system. An IAEA initiative is currently underway to address this problem in existing and proposed near-surface disposal facilities with the following objectives (IAEA, 1999).

(a) To test the feasibility of a proposed disposal concept as a means to manage the waste.

(b) Provide some guidance (be it only in order of magnitude) on the likely impact that the disposal system may have on people and the environment.

(c) Assist with the 'screening process' in the organization and development of a near-surface disposal system.

The derived activity values can also be used as a starting point by countries and organisations that are developing plans for a near-surface disposal, but do not have site-specific activity values.

The next step after a potential site has been selected is to design and characterize a disposal concept that will comply with the regulatory and other constraints. The information on the site's geosphere and biosphere acquired during the preceding phases will provide valuable information for the design, particularly the information on features that may act as natural barriers. However, other factors, such as the characteristics of the waste, and the design of the waste packages, engineered barriers and repository will all influence the design of the disposal concept. This will require vast amounts of information on the system's near-field, geosphere and biosphere, even if the more liberal approach advanced in this thesis is followed. However, this can be reduced if the safety analyst's attention is focussed on the primary objective of the investigation—to show that the disposal system will satisfy all regulatory and other constraints within reasonable limits. It is believed that the most efficient way to achieve this objective is to follow the safety assessment approach outlined in Figure 6–1 and the decision analysis framework discussed in Chapter 9. As shown in Figure 6–1 the analyst will reach a point where he or she has to decide whether the safety case was adequately demonstrated or not. In the latter case, the analyst will have to assess what additional information will be needed. In many cases this will entail the acquisition of more detailed system-specific data, especially on those parameters and assumptions that have the greatest impact on the system's performance. However, as shown by the discussion of data worth analysis in Chapter 9, there is a cost-effective limit on obtaining the additional data. The limit is, however, not fixed but varies with the type of data and the purpose of the investigation. It may therefore be necessary to perform a decision analysis of the nuclear liabilities at this point to see if the additional costs can be justified, before proceeding with the acquisition of the data.

An important feature of the proposed iterative approach is that it allows one to evaluate some elements of the disposal concept before money is spent on any construction work. This applies in particular to the features, such as the materials to use in the construction of the repository, the engineered barriers and the waste packages, whose physical and chemical properties are well understood, or can be determined in the laboratory. The influence that different geometries of the repository, engineered barriers and waste packages may have on the regulatory compliance criteria can also be evaluated at this stage and a few selected for further investigation. The possibility that the system may fail and the conditions under which a failure will occur can also be investigated and used to estimate the corresponding nuclear liabilities, which can then be used to refine the decision analysis. This approach will ensure that only those components and their properties that are really necessary to ensure regulatory compliance will be included in the concept design. This can cause a considerable potential reduction in the nuclear liabilities, the construction and assessment costs.

A post-closure assessment ends technically the moment a disposal system is licensed and in operation. However, it would be foolish to stop all investigations of the system at that stage and not proceed with a pre-closure assessment, which the discussion in Chapter 1 indicates should form an integral part of the post-closure assessment. One reason why this may be necessary is that it is very difficult to include the consequences of a vehicle or other accident on the site that may affect the post-closure assessment adversely. This will clearly lengthen the institutional control period of the site and add to its nuclear liabilities, but one can once again try to reduce this with a detailed decision analysis of the accident's consequences.

Any waste disposal system has a finite life span, determined by the capacity of the repository, at which time the system has to be closed and decommissioned. As used here the term decommissioned refers to the actions that need to be taken, after its closure, with adequate regard for the health and safety of workers and members of the public and the protection of the environment (IAEA, 1993*b*). In the case of a disposal system this involves the orderly decontamination, dismantling and removal of all the surface and subsurface support facilities, erected during its operational life, the back-filling and sealing of all tunnels, shafts, exploratory and monitoring boreholes and possibly the restoration of the site as close as possible to its original environmental state. Although the repository will have to be closed at a specific moment in time, it may sometimes be advantageous to delay the decommissioning of site, especially when the results of a post-closure assessment is affected negatively by an accident on the site. However, one should be able to control the intensity and duration of the observations and the associated nuclear liabilities by a detailed analysis of the large volume of information on the site's behaviour that should be available at that time.

10.3 RECOMMENDATIONS

A post-closure safety assessment of radioactive waste disposal systems is an extensive exercise that requires input from various scientific, social and economic disciplines. No attempt was therefore made to describe the actual implementation in detail. However, it is believed that the thesis lie the foundation for a post-closure assessment procedure that will be within the financial abilities of South Africa, and able to clarify the post-closure behaviour of Vaalputs and to select a disposal site for the accumulated spent fuel. The same methodology can also be used to decide whether Thabana should be closed permanently and cleaned up, or developed into a permanent disposal site for radioactive waste, that will not impose undue burdens on future generations.

As mentioned in Chapter 1 a preliminary version of the methodology has already been used to assess the suitability of the BOSS concept for the disposal of spent nuclear sources. Although the methodology has been refined since that assessment, it is felt that the procedure used to generate scenarios for the assessment of uncertainties in the future evolution of the site can be improved. No attempt was made to incorporate the decision analysis framework, introduced for the first time in this thesis, in a consistent and logical manner into the safety assessment methodology.

The safety assessment methodology need not be restricted to the waste disposal system as such, but can also be used by regulatory authorities to justify their decisions to the owner or operator of the disposal site and the public (Kozak, 1994). This includes aspects, such as granting a licence for a specific site, or an order to close the site, restrictions placed on land usage after the end of the institutional control period, and the durability requirements for intrusion resistant covers (Siraky, 1996).

APPENDIX A

THE INTERNATIONAL ISAM LIST OF FEATURES, EVENTS AND PROCESSES (FEPS) FOR A RADIOACTIVE DISPOSAL SYSTEM

A.1 DEFINITION OF THE DISPOSAL SYSTEM DOMAIN

Layers 1, 2 and 3 of this international list are all defined relative to the following definition of the Disposal System Domain.

The disposal system domain is defined as the spatial and temporal domain that consists of:

(a) the wastes,

(b) the engineered and natural barriers expected to contain the waste,

(c) the potentially contaminated geology and surface environment,

(d) the geology, surface environment and human behaviour necessary to provide an estimate of the movement and the exposure of man to the radionuclides, following the closure of the repository.

A.2 LAYERS AND CATEGORIES OF THE LIST

Layer 0 Assessment Context

Assessment context factors are factors that the analyst will consider in determining the scope of the analysis; these may include factors related to regulatory requirements, definition of desired calculation end-points and requirements in a particular phase of assessment. Decisions at this point will affect the phenomenological scope of a particular phase of assessment; i.e. what "physical FEPs" will be included. For example, some classes of future human actions or extreme future events unrelated to the repository may be excluded.

Layer 1 External Factors

External Factors are FEPs with causes or origins outside the disposal system domain, i.e. natural or human factors of a more global nature and their immediate effects. Included in this layer are decisions related to repository design, operation and closure since these are outside the temporal bound of the disposal system domain.

In general, external factors are not influenced, or only weakly influenced, by processes within the disposal system domain. In developing models of the disposal system domain, external factors are often represented as boundary conditions or initiating events for processes within the disposal system domain. The following categories are used for this purpose.

1.1 Repository issues – Decisions on the design, waste allocation, and events related to the site investigation, operations and closure.
1.2 Geological processes and effects – Processes arising from the wider geological setting and long-term processes.
1.3 Climatic processes and effects – Processes related to global climate change and associated regional effects.
1.4 Future human actions – Human actions and regional practices in the post-closure period that can potentially affect the performance of the engineered and geological barriers, e.g. intrusive actions, but not the passive behaviour and habits of the local population in 2.4 below.
1.5 Other – all other FEPs not included in 1.1 to 1.4, e.g. meteorite impacts.

Note that there are only a few significant direct interactions between FEPs in the different categories of external factors, in general.

Layer 2 Environmental Factors Affecting the Disposal System Domain

The Disposal system domain environmental factors are FEPs processes that occur within the spatial and temporal domain, relevant to the estimation of the release and migration of radionuclides, and the effect that has on the evolution of the physical, chemical, biological and human conditions, controlling the exposure of man to the nuclides. (See also Layer 3). The following categories belong to this layer.

2.1 Waste and engineered barriers – The type, structure and nature of these components.
2.2 Geological environment – The hydrogeological, geomechanical and geochemical FEPs related to the pre-emplacement and modified state of the environment (and other long-term changes) caused by the presence of the repository.
2.3 Surface environment – The FEPs related to surface phenomenon, including near-surface aquifers and unconsolidated sediments, but excluding human activities and behaviour in 1.4 and 2.4.
2.4 Human behaviour – The habits and characteristics of individuals and population (in other words, the critical group for which exposures have to be calculated) that will impact on the performance of the engineered or geological barriers, but excluding the intrusive or other activities in 1.4.

The interactions between the FEPs in this layer of categories are clearly very important for the assessment.

Layer 3. Radionuclide and Contaminant Factors

The radionuclide and contaminant factors include those FEPs that will directly affect the disposal system environment and the dose members of the critical group may receive from the given concentrations of radionuclides in the environment. There are essentially three categories in this layer.

3.1 Contaminant characteristics – The characteristics of radiotoxic and chemotoxic nuclide species that might be considered in a post-closure safety assessment.
3.2 Release and migration factors – The FEPs that directly affect the release and migration of radionuclides in the disposal system domain.
3.3 Exposure factors – Processes and conditions that directly affect the dose members of the critical group may receive from the given concentrations of radionuclides in the environmental media.

The boundaries between the different layers and categories are subjective and will depend on the individual analyst's concept of the disposal system. Nevertheless, this should not prevent a self-consistent assignment of FEPs within the International List itself or when mapping project FEPs to the International List.

A.3 THE DRAFT ISAM INTERNATIONAL FEP LIST (VERSION 1.0) IN CLASSIFICATION SCHEME ORDER.

Layer 0 Assessment Context

0.01 Assessment endpoints
0.02 Time scales of concern
0.03 Spatial domain of concern
0.04 Repository assumptions
0.05 Future human action assumptions
0.06 Future human behaviour (target group) assumptions
0.07 Dose response assumptions
0.08 Assessment purpose
0.09 Regulatory requirements and exclusions
0.10 Model and data issues

Layer 1 External Factors

1.1 REPOSITORY ISSUES
1.1.01 Site investigation

1.1.02 Excavation and construction
1.1.03 Emplacement of wastes and back-filling
1.1.04 Closure e.g. capping
1.1.05 Records and markers, repository
1.1.06 Waste allocation
1.1.07 Repository design
1.1.08 Quality control
1.1.09 Schedule and planning
1.1.10 Administrative control, repository site
1.1.11 Monitoring of repository
1.1.12 Accidents and unplanned events
1.1.13 Retrievability

1.2 GEOLOGICAL PROCESSES AND EFFECTS

1.2.01 Tectonic movements and orogeny
1.2.02 Deformation, elastic, plastic or brittle
1.2.03 Seismicity
1.2.04 Volcanic and magmatic activity
1.2.05 Metamorphism
1.2.06 Hydrothermal activity
1.2.07 Erosion and sedimentation
1.2.08 Diagenesis
1.2.09 Salt diapirism and dissolution
1.2.10 Hydrological and hydrogeological response to geological changes

1.3 CLIMATIC PROCESSES AND EFFECTS

1.3.01 Climate change, global
1.3.02 Climate change, regional and local
1.3.03 Sea level change
1.3.04 Periglacial effects
1.3.05 Glacial and ice sheet effects, local
1.3.06 Warm climate effects (tropical and desert)
1.3.07 Hydrological and hydrogeological response to climate changes
1.3.08 Ecological response to climate changes
1.3.09 Human response to climate changes
1.3.10 Other geomorphological changes

1.4 FUTURE HUMAN ACTIONS

1.4.01 Human influences on climate
1.4.02 Motivation and knowledge issues (inadvertent and deliberate human actions)
1.4.03 Un-intrusive site investigation
1.4.04 Drilling activities (human intrusion)
1.4.05 Mining and other underground activities (human intrusion)
1.4.06 Surface environment, human activities
1.4.06.01 Surface Excavations
1.4.06.02 Pollution
1.4.06.03 Site Development
1.4.06.04 Archaeology
1.4.07 Water management (wells, reservoirs, dams)
1.4.08 Social and institutional developments
1.4.09 Technological developments
1.4.10 Remedial actions
1.4.11 Explosions and crashes

1.5 OTHER

1.5.01 Meteorite impact
1.5.02 Miscellaneous and FEPs of uncertain relevance

Layer 2 Environmental Factors Affecting the Disposal System Domain

2.1 WASTES AND ENGINEERED FEATURES

2.1.01 Inventory, radionuclide and other material
2.1.02 Waste form materials and characteristics
2.1.03 Container materials and characteristics
2.1.04 Buffer and backfill materials and characteristics
2.1.05 Engineered barriers system e.g. caps
2.1.06 Other engineered features materials and characteristics
2.1.07 Mechanical processes and conditions (in wastes and EBS)
2.1.08 Hydraulic and hydrogeological processes and conditions (in wastes and EBS)
2.1.09 Chemical and geochemical processes and conditions (in wastes and EBS)
2.1.10 Biological and biochemical processes and conditions (in wastes and EBS)
2.1.11 Thermal processes and conditions (in wastes and EBS)
2.1.12 Gas sources and effects (in wastes and EBS)
2.1.13 Radiation effects (in wastes and EBS)
2.1.14 Nuclear criticality
2.1.15 Extraneous materials

2.2 GEOLOGICAL ENVIRONMENT

2.2.01 Disturbed zone, host lithology
2.2.02 Host lithology
2.2.03 Lithological units, other
2.2.04 Discontinuities, large scale (in geosphere)
2.2.05 Contaminant transport path characteristics (in geosphere)
2.2.06 Mechanical processes and conditions (in geosphere)
2.2.07 Hydraulic and hydrogeological processes and conditions (in geosphere)
2.2.08 Chemical and geochemical processes and conditions (in geosphere)
2.2.09 Biological and biochemical processes and conditions (in geosphere)
2.2.10 Thermal processes and conditions (in geosphere)
2.2.11 Gas sources and effects (in geosphere)
2.2.12 Undetected features (in geosphere)
2.2.13 Geological resources

2.3 SURFACE ENVIRONMENT

2.3.01 Topography and morphology
2.3.02 Soil and sediment
2.3.03 Aquifers and water-bearing features, near surface
2.3.04 Lakes, rivers, streams and springs
2.3.05 Coastal features
2.3.06 Marine features
2.3.07 Atmosphere
2.3.08 Vegetation
2.3.09 Animal populations
2.3.10 Meteorology
2.3.11 Hydrological regime and water balance (near-surface)
2.3.12 Erosion and deposition
2.3.13 Ecological, biological and microbial systems
2.3.14 Animal and plant intrusion leading to vault and trench disruption

2.4 HUMAN BEHAVIOUR

2.4.01 Human characteristics (physiology, metabolism)
2.4.02 Adults, children, infants and other variations
2.4.03 Diet and fluid intake
2.4.04 Habits (non-diet-related behaviour)
2.4.05 Community characteristics
2.4.06 Food and water processing and preparation

2.4.07 Dwellings
2.4.08 Wild and natural land and water use
2.4.09 Rural and agricultural land and water use (including. fisheries)
2.4.10 Urban and industrial land and water use
2.4.11 Leisure and other uses of environment

Layer 3. Radionuclide and Contaminant Factors

3.1 CONTAMINANT CHARACTERISTICS

3.1.01 Radioactive decay and in-growth
3.1.02 Chemical and organic toxin stability
3.1.03 Inorganic solids and solutes
3.1.04 Volatiles and potential for volatility
3.1.05 Organics and potential for organic forms
3.1.06 Noble gases

3.2 CONTAMINANT RELEASE and MIGRATION FACTORS

3.2.01 Dissolution, precipitation and crystallisation, contaminant
3.2.02 Speciation and solubility, contaminant
3.2.03 Sorption and desorption processes, contaminant
3.2.04 Colloids, contaminant interactions and transport with
3.2.05 Chemical and complexing agents, effects on contaminant speciation and transport
3.2.06 Microbial, biological and plant-mediated processes, contaminant
3.2.07 Water-mediated transport of contaminants
3.2.08 Solid-mediated transport of contaminants
3.2.09 Gas-mediated transport of contaminants
3.2.10 Atmospheric transport of contaminants
3.2.11 Animal, plant and microbe mediated transport of contaminants
3.2.12 Human-action-mediated transport of contaminants
3.2.13 Food chains, uptake of contaminants in

3.3 EXPOSURE FACTORS

3.3.01 Drinking water, foodstuffs and drugs, contaminant concentrations in
3.3.02 Environmental media, contaminant concentrations in
3.3.03 Non-food products, contaminant concentrations in
3.3.04 Exposure modes
3.3.05 Dosimetry
3.3.06 Radiological toxicity and other effects
3.3.07 Non-radiological toxicity and other effects
3.3.08 Radon and radon daughter exposure

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SUMMARY

Radioactive waste in South Africa is generated through the nuclear fuel cycle and the application of radioactive materials in industry, science and medicine. The radioactive waste is presently disposed at Vaalputs in Bushmanland and Thabana at Pelindaba in near-surface disposal facilities. No strategy exists at present for the disposal of high level waste.

The objective of radioactive waste management and its underlying principles is to ensure that human health and the environment are protected at all times, without imposing an undue burden on future generations. This implies that, before any long-term management strategy of radioactive waste disposal can be implemented, the impact of the disposed waste must be determined as a function of time—a procedure referred to as *post-closure safety assessment.*

In this thesis, a methodology to perform post-closure safety assessments of radioactive waste disposal systems in South Africa and other parts of Africa is described. Not only will it contribute significantly to reassess the suitability of current waste disposal practices, but also lays the foundation for future disposal practices.

The proposed methodology—an integrated approach to radioactive waste management—is aimed at:

(a) ensuring the safety of the present public and future generations,

(b) enhancing the public acceptance of the methodology,

(c) keeping the expenditure associated with the implementation of the methodology at a minimum.

The methodology recognises the interdependence between operational phase activities and the post-closure behaviour of the disposal system. It is an iterative process that considers site-specific, prospective evaluations of the post-closure phase to ensure that the disposal system will comply with internationally accepted criteria, within reasonable limits. Provision is therefore made to identify the data, design and other needs that will contribute towards the achievement of this objective. The first step in this procedure is to identify those internal and external features, events and processes that can be used to predict how radioac-

tive material may escape from the disposal facility, along which paths will it migrate and how it may impact humans. Various conceptual and mathematical models that can be used to develop appropriate scenarios of these processes and to compare the results with site-specific data are discussed in the thesis.

The cost to develop a waste disposal system, the disposal of the waste and the pre- and post-closure assessments of the system, or so-called nuclear liabilities, can be astronomically high. Combining the post-closure assessment of the system with the decision analysis framework discussed in the thesis can reduce these costs considerably.

Post-closure assessments of radioactive waste disposal systems have in the past often been interpreted as an exercise to predict the exact behaviour of the system far into the future. However, as pointed out in the thesis this is not possible, even with the technology available today. The more pragmatic approach, advanced in the thesis, is that modern technology is able to demonstrate to reasonable members of the public that such a system will be safe. Nevertheless it is recognized that the methodology cannot be implemented without the active participation of the public. It is therefore envisaged that the proposed methodology will be implemented with the close co-operation of the public, particularly those living near the site where the disposal system will be implemented.

OPSOMMING

In Suid Afrika word radioaktiewe afval gegenereer deur die kernbrandstofsiklus en die gebruik van radioaktiewe bronne in die industrie, die wetenskap en geneeskunde. Lae en middel energie radioaktiewe afval word tans weggedoen in oppervlak wegdoeningsfasiliteite by Vaalputs in die Boesmanland en geberg in vlak bergingsfasiliteite by Thabana by Pelindaba. Geen strategie bestaan egter tans vir die wegdoening van hoë energie afval nie.

Die doel van radioaktiewe afvalbestuur en die onderliggende beginsels daarvan, is om te verseker dat die mens se gesondheid asook die omgewing te alle tye beskerm word, sonder om onnodige druk op toekomstige geslagte te plaas. Dit impliseer dat, alvorens enige langtermyn bestuurstrategie vir radioaktiewe afval geïmplementeer kan word, moet die impak van die gebergde afval as 'n funksie van tyd bepaal word—'n prosedure waarna verwys word as 'n nasluiting veiligheidsvasstelling (post-closure safety assessment).

'n Metodiek om so 'n nasluiting veiligheidsvasstelling vir radioaktiewe afval wegdoeningsisteme in Suid Afrika en ander dele van Afrika uit te voer, word in hierdie proefskrif beskryf. Nie alleen bied dit 'n beduidende bydrae tot die herevaluering van bestaande wegdoeningsisteme nie, maar verskaf dit ook 'n basis vir toekomstige wegdoeningspraktyke wêreldwyd.

Die voorgestelde metodiek—'n geïntegreerde benadering tot radioaktiewe afvalbestuur—het ten doel om:

(a) die veiligheid van huidige en toekomstige geslagte te verseker,

(b) publieke aanvaarding van die metodiek te bevorder,

(c) finansiële uitgawes geassosieer met die implementering van die metodiek te minimeer.

Die metodiek aanvaar die interafhanklikheid wat daar bestaan tussen aktiwiteite binne die operasionale fase en die nasluitingsgedrag van die sisteem. Dit is 'n iteratiewe proses wat bestaan uit terrein spesifieke, toekomstige evaluasies van die nasluiting fase, met die doel om te verseker dat die wegdoeningsisteem voldoen aan internasionaal aanvaarbare kriteria, gegee sekere redelike beperkings. Gevolglik word voorsiening gemaak vir die identifisering van data, ontwerp en ander behoeftes wat 'n bydrae sal lewer om die doelwit te bereik. Die

eerste stap in hierdie prosedure is om alle interne en eksterne eienskappe, gebeurtenisse en prosesse (features, events and processes) te bepaal wat gebruik kan word om te voorspel hoe radioaktiewe materiaal vanuit die wegdoeningsfasiliteit kan ontsnap, langs watter paaie dit sal beweeg en wat die gevolglike impak op die mens sal wees. Verskeie konsepsuele en wiskundige modelle wat gebruik kan word om die prosesse voor te stel en die resultate met terrein spesifieke data te vergelyk, word in die proefskrif bespreek.

Die koste om so 'n radioaktiewe afval wegdoeningsisteem te ontwikkel, die wegdoening van die afval en die ondersoeke wat nodig is om vas te stel of so 'n sisteem veilig sal wees, algemeen bekend as kernaanspreeklikheid, kan astronomies hoog wees. Soos aangetoon in die proefskrif kan hierdie kostes egter aansienlik verminder word deur die nasluitingsevaluasie van die sisteem met 'n besluitnemingsanalise raamwerk te kombineer.

Nasluitingsevaluasies van radioaktiewe afval wegdoeningsisteme is dikwels in die verlede geïnterpreteer as 'n oefening wat daarop ingestel is om die werklike verre toekomstige gedrag van die sisteem te voorspel. Soos in die proefskrif aangedui word, is dit egter nie moontlik nie, self nie eers met die tegnologie wat vandag beskikbaar is nie. Wat die moderne tegnologie wel in staat is om te doen, is om aan te toon dat so 'n sisteem binne aanvaarbare perke veilig is. 'n Meer pragmatiese benadering word dus in die proefskrif voorgestel, nl. om aan redelike persone aan te dui dat so 'n sisteem wel veilig sal wees. Die metodiek kan dus nie geïmplementeer word sonder die aktiewe deelname van die publiek nie. Die sukses van die voorgestelde metodiek sal dus hoofsaaklik bepaal word deur die samewerking van die publiek, veral dié persone wat naby 'n terrein woon wat vir so 'n wegdoeningsisteem geoormerk is.