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Radioactive Waste in Perspective

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Cover credit: Radioactive waste disposal at Gorleben, Germany (Bundesamt für Strahlenschutz, Foto: Menkhaus), and CO_2 reservoir at Schwarze Pumpe, Germany (AFP photo/DDP/Michael Urban).

FOREWORD

The objective of this study is to provide an overview of the current management of radioactive and hazardous wastes. Its intended audience is policy makers and interested stakeholders.

This work has two themes that compare:

- radioactive and hazardous wastes and their management strategies in general; and
- the management of wastes arising from coal and from nuclear power generation in particular.

These two themes provide two distinct perspectives. The first illustrates that the disposal of radioactive waste is not a uniquely difficult issue, as is sometimes perceived. The second compares the wastes arising from two of the probable low-carbon baseload electricity generating technologies to be used in the future: nuclear power and coal-fired generation with carbon capture and storage. Neither technology is without its waste challenges, although they are very different, and both will rely to varying degrees on geological storage.

The goal of these comparisons is to illustrate similarities and differences in these wastes and their management. Aspects of the wastes and their management that are examined include the inherent hazards of the waste, risks posed, regulatory requirements applied, treatment and disposal methods, risk communication, and social acceptance of disposal facilities and practices.

The study has been carried out by an *ad hoc* group of experts under the guidance of the NEA Committee on Technical and Economic Studies on Nuclear Energy Development and the Fuel Cycle (NDC) with participation by the OECD Environment Directorate, the International Atomic Energy Agency (IAEA) and the Secretariat to the NEA Radioactive Waste Management Committee (RWMC). The study was also reviewed by the RWMC before publication.

Acknowledgements

The members of the *ad hoc* group of experts nominated by NEA member countries are listed in Appendix 5. The OECD Nuclear Energy Agency Secretariat would like to acknowledge the important contribution made by each member of the expert group. The group was co-chaired by Guy Collard of Belgium, Mariano Molina of Spain (who wrote Appendix 1: *Strategic Issues for Radioactive Waste*) and Joachim Wuttke of Germany (who wrote Appendix 2: *Strategic Issues for Hazardous Waste*). Sten Bjurström wrote Appendix A3.2: *A Case Study on Mercury Containing Waste.* Torsten Eng, Administrator in the NEA Nuclear Development Division was the Secretary to the *ad hoc* Group and prepared the main report. Stan Gordelier, former Head of the Nuclear Development Division provided a substantial review of the text in response to the numerous comments that had been received on earlier versions of the report. Betsy Forinash of the NEA Radiological Protection and Radioactive Waste Management Division provided helpful comments. Jan Horst Keppler and Ron Cameron of the Nuclear Development Division provided final collation and review. The NEA is also grateful for the expert input to the group from the OECD Environment Directorate and from the UN International Atomic Energy Agency.

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KEY POINTS FOR POLICY MAKERS

In OECD countries, both radioactive and hazardous wastes (a term used in this report for potentially dangerous non-radioactive wastes) are strongly regulated and safely managed. The principles applied to the management of both waste types are essentially the same.

The safe disposal of radioactive waste is not the uniquely difficult issue that is perceived by the media, much of the public and by many politicians:

- Radioactive waste is produced in much lower quantities than hazardous waste.
- Low-level and short lived intermediate-level wastes (LILW-SL) are already being disposed to repositories in many countries. On a volumetric basis, some three quarters of all the radioactive waste created since the start of the nuclear industry has already been sent for disposal.
- Whilst concern is expressed that some radioisotopes in waste decay so slowly that they remain potentially dangerous for very long periods of time, some hazardous wastes (e.g. mercury, arsenic) have infinite lives.

Radioactive wastes arise from the nuclear industry, from other industrial sources and from medical applications. The eventual safe disposal of all categories is a necessity with or without any further construction of nuclear power plants.

There is a worldwide consensus amongst technical experts in the field that properly established deep geological disposal is an entirely appropriate management approach for high-level waste and spent nuclear fuel (HLW/SF). While facilities exist in many countries for LILW-SL there is, as yet, no facility for HLW/SF.

Opinion polls clearly show that the issue of radioactive waste disposal features strongly in the public's often negative opinion of nuclear energy. Neither governments nor the nuclear industry have been able to effectively communicate the risks and benefits of nuclear power and waste disposal in a manner that could secure public acceptance of disposal facilities.

Even though hazardous wastes are produced in much larger quantities and arise from a much larger number of sources than do radioactive wastes, arrangements for their safe management and disposal have not attracted the same degree of public and political attention.

The hazardous waste management industry has been more successful in implementing final disposal arrangements than its radioactive waste counterpart. Indeed, over recent time the hazardous waste industry has concluded that deep geological disposal of some infinitely lived wastes is an appropriate disposal methodology, following the approach that the radioactive waste community have been endeavouring to pursue for many years. In contrast to radioactive waste, deep geological disposal for some especially hazardous long lived wastes has already been successfully achieved in some countries.

The facts that hazardous wastes are produced in much larger quantities and come from a much more numerous and diverse set of sources have provided strong driving forces for the resolution of hazardous waste disposal issues. In contrast, the much smaller quantities of radioactive waste, mainly arising from a very limited number of producers, has meant that storage has been a safe and economically viable option to date. This has reduced the necessity to establish final disposal arrangements and resulted in the deferral of potentially contentious decisions.

With the growing concerns on CO_2 emissions and climate change, it is probable that there will be a growth in nuclear energy generation and also in carbon capture and storage (CCS) technology applied to coal and gas fired power stations. CCS is still under development and is not yet commercially available, but it is believed to hold considerable promise. Both nuclear energy and those fossil fired technologies equipped with CCS will rely on deep geological disposal for their important waste streams, albeit that CO_2 is not considered to be a hazardous waste. In the case of radiological waste, the containment is based on a combination of a solidified waste form, engineered and geological barriers. In the case of CCS, the waste form is a supercritical fluid and containment relies only on geological barriers.

The reliance of both technologies on geological disposal provides both an interesting parallel and a contrast, particularly in view of the significant difference in quantities and engineered barriers. However, the consequences of repository failure differ significantly between the two technologies. Given the solidified nature of the radioactive waste form a catastrophic major release is virtually impossible and the concern relates to health consequences of very slow releases via groundwater transmission in the very long term. In contrast, a catastrophic release of $CO_{2,}$ whilst unlikely, is possible for CCS if, for example, there were to be a pipeline transportation or injection well capping failure. Such a release could result in deaths in any local community. However, slow long term CO_2 release is more probable, but would have negligible health consequences beyond that of contributing to global warming.

EXECUTIVE SUMMARY

Each year society produces 8 000-10 000 Mt of waste worldwide (excluding overburden from mining and mineral extraction wastes, which are not usually counted as a waste). Of this about 400 Mt is hazardous waste and about 0.4 Mt is radioactive waste, which is mainly currently being generated by the world's nuclear power plants and their fuel cycle support facilities.

The objective of this NEA study is to put the management of radioactive waste into perspective, firstly by contrasting features of radioactive and hazardous wastes, together with their management policies and strategies and secondly by exploring the wastes resulting from the most important future alternative technology for generating low carbon release electricity. Hence the study has two themes that aim to offer policy makers a broad perspective on the similarities and differences between:

- Radioactive and hazardous wastes and their management strategies
- Management of wastes from coal and from nuclear power generation

Direct comparisons between radioactive and hazardous waste management must be done very cautiously because the very different hazard characteristics of the two waste types require different processing techniques to assure safety. However, there is a fundamental and essential similarity: both radioactive and hazardous wastes have the potential, if not managed appropriately, to cause environmental harm and to damage human health.

Similarly, there are significant differences between the wastes produced by different power generation sources and again any comparisons must be undertaken cautiously.

Theme 1 – Radioactive and hazardous wastes and their management strategies

In volume terms, the global generation rate of hazardous waste is up to three order of magnitude higher than that of radioactive waste from the nuclear power industry; almost all industries and households generate hazardous waste, but most radioactive waste comes from a very few sources – primarily electricity generation.[1] Taking the United States as an example, there are in the order of 100 times more large hazardous waste generators than radioactive waste generators.

Radioactive wastes, particularly those generated by nuclear power plants, also have well-known constant characteristics, which is a considerable advantage in being able to predict their behaviour when disposed to a repository. Waste characteristics, and therefore management strategies, are

1. Radioactive waste is also generated in very significant quantities by military activities, by research and development, medical applications and by various other non-nuclear industries. This report focuses on the civil uses of nuclear technology for the production of electricity. Some waste streams are both radioactive and toxic (so called mixed wastes), presenting management difficulties from both aspects. It should also be recognised that some radioactive waste streams contain lead and stable lead will ultimately be the natural decay product of some radio-nuclides. Lead is, itself, a hazardous material in waste.

fundamentally different between hazardous waste (which can have a range of hazardous characteristics making it flammable, oxidising, corrosive, reactive, explosive, toxic (including carcinogenicity) or ecotoxic) and radioactive waste which, in broad terms, has only radioactivity (which can cause serious tissue damage or fatalities at high doses and which may cause cancer in the long term at lower doses) as a hazard. Radioactivity decays predictably over time (albeit that for some isotopes this is over a very long timescale), so the hazard associated with radioactive waste continuously reduces. Whilst much hazardous waste can be fully treated to pose virtually no hazard before it is disposed, the intrinsic hazards in some hazardous wastes remain for all time. In this sense there is a parallel between the most difficult wastes arising from the two categories; longevity is not unique to radioactivity.

The unit costs of managing hazardous waste are considerably lower than for managing radioactive waste. Hazardous waste management is generally carried out on a commercial basis with immediate payment for services received; for radioactive waste, funds are generally built up from electricity generation revenues to pay for future disposal in facilities that may not yet exist. In most cases, market forces drive early implementation of hazardous waste management facilities in a way that is not seen for radioactive waste.

The implementation time for hazardous waste management facilities is generally much shorter than for radioactive waste facilities; gaining socio-political acceptance for hazardous waste disposal appears easier than achieving acceptance for geological disposal of radioactive waste. This may be due to differing public perceptions regarding the risks posed by radioactive and hazardous waste disposal facilities.

Theme 2 – Management of wastes from coal and from nuclear power generation

In 2007, about 40% of the world's electricity came from coal and 14% from nuclear generation. Globally, coal generation produces about 11 000 Mt/a (1 700kt/TWh) of wastes (including 10 500Mt/a of CO_2; 1 600kt/TWh) and additionally some 20 000 Mt/a (3 000kt/TWh) of mining wastes. Nuclear generation, taking into account the wastes from plants that will eventually be decommissioned, produces <0.5 Mt/a of wastes (<0.2kt/TWh) and 45Mt/a (<8kt/TWh) of mining and uranium milling wastes. Unlike nuclear power, most of the waste products from coal generation are disposed directly into the environment. There is global concern about the climate change effects of CO_2 emissions from fossil fired electricity generation, and air pollution from coal-fired electricity production includes a mixture of species potentially damaging to health and the environment.

In the vast majority of countries, all solid waste from coal-fired generation is allowed to be disposed to landfill. A considerable proportion of nuclear power solid wastes (very-low level, VLLW) can be considered for disposal at simple landfill facilities; only about 2% of nuclear power waste is high-level waste (HLW) or spent fuel (SF), which contain most of the radioactivity, and for which no disposal facilities are currently available.[2]

Carbon capture and storage (CCS) are technologies under development to extract carbon dioxide from the exhaust stream of large stationary centres of fossil fuel combustion and prevent it from dispersing into the atmosphere. Both coal with CCS and nuclear power rely on deep geological repositories as their waste management solution. Waste from CCS would be disposed as a supercritical

2. Some long-lived intermediate level waste will also require geological disposal, but HLW/SF contains the vast majority of the radioactivity (~97%) and is the most contentious waste stream.

fluid[3] contained only by natural barriers whilst waste from nuclear power would be disposed as a solidified and encapsulated product contained by both engineered and natural barriers.

CO_2 is not considered to be a hazardous waste. A large prompt release (for example from a CCS well cap failure or a transmission pipeline break) could, however, constitute a major risk including potential fatalities. Putting aside these potential accidental releases, the main issue is the long term retention of the CO_2 if the technology is to be effective in combating climate change. CO_2 has been injected into oil reservoirs for almost 40 years to enhance oil recovery without detectable losses of CO_2 over these timescales. However, measurement accuracy is insufficient to provide confidence for CO_2 retention in the longer term. If there were to be long-term leakage, the impact on climate change would simply be deferred rather than eliminated. A key issue for investors will be the extent of their liability for long-term monitoring and potential remediation.

Geological disposal of CO_2 may prove to become more contentious in the future; NGOs such as Friends of the Earth International and Greenpeace International support neither CCS nor nuclear power as a means to combat climate change. It is possible that CCS may, in future, suffer from the same public acceptance difficulties that have slowed progress in radioactive disposal.

Lessons learned

Both hazardous and radioactive wastes are generally well managed in OECD countries, although the public commonly perceives that both radioactive and some hazardous waste management are high-risk activities. However, there are many examples of hazardous wastes (including toxic and biohazard wastes) and radioactive wastes being safely disposed. Although large numbers of hazardous waste landfills exist worldwide, most countries with radioactive waste disposal capabilities have only a few near-surface facilities for low- and intermediate-level radioactive waste (LILW), although the disposal approaches and technological solutions are similar. The lower number of radioactive waste facilities is due partly to the fact that the volumes of waste requiring disposal are much smaller. Currently, there is no disposal facility in operation in the world for high-level waste (HLW) or spent fuel (SF), which are very small in volume but contain a very high proportion (~ 97%) of the radioactivity produced in the nuclear fuel cycle. As such it is the waste stream which attracts the most attention and it is regarded as the most problematic. There are also disposal issues associated with long lived intermediate level waste that need to be addressed since much of this may also need deep geological disposal.

In view of the larger number of hazardous waste facilities as well as the lack of disposal facilities for HLW, it would appear that the economic and other driving forces in place for implementation of strategies for hazardous waste management have been more effective in overcoming implementation obstacles, but the driving forces to implement radioactive waste management strategies have been much less effective.

The huge amount of hazardous waste generated by society means that timely decision-making on the implementation of management facilities was essential if countries' industrial capabilities were not to come to a halt. There was therefore a clear national economic, and hence political, imperative to implement hazardous waste management processes, including disposal. The volumes of radioactive waste are relatively small, allowing the nuclear industry historically to manage them safely and

3. A supercritical fluid is any substance at a temperature and pressure above its critical point. Such fluids have properties of both gases and liquids; they can diffuse through solids like a gas and dissolve materials like a liquid.

economically using surface storage. Hence the national industrial capabilities were not broadly understood to be threatened by inaction and the same imperatives have not applied.

Because of the widespread generation of hazardous wastes there are market opportunities for the development of hazardous waste treatment and disposal. The same is currently not true for radioactive wastes, where the generators usually treat their waste in-house and, in many cases, temporarily store it on their own sites for eventual disposal without further treatment.

Although the technology is clearly still in its infancy, economic driving forces appear to have arisen for CCS plant proposed for coal-fired power stations. A methodology is available to assess the effect of CCS on greenhouse gas emissions, enabling countries to report emissions reductions due to CCS and providing the basis for its inclusion in emissions trading schemes.

One important factor, which appears to make timely decision-making less difficult for hazardous, compared with radioactive, waste disposal is that the public perceives a lower level of risk for hazardous waste management. A significant reason may be the difference in familiarity between radioactive and non-radioactive waste types. Many common household items such as constituents of refrigerators, fluorescent tubes and batteries are generally classified as hazardous wastes when they are disposed, and potentially toxic chemicals like wood preservatives and pesticides are in common household use. Thus, the public is broadly familiar with many types of hazardous materials that generate or may become wastes and can see a direct correlation with its lifestyle and personal convenience. Such familiarity does not generally exist for radioactive waste, as it is generated and managed by small numbers of people on relatively few sites. While people recognise that they rely on electricity, the source of power generation is remote from their everyday lives. Context and evolving views of public participation in decision making are also important; a new hazardous waste disposal facility is now likely to face considerably more opposition than in the past.

Another factor may be that the public recognises that management of large volumes of hazardous waste is a by-product of the economic activities that are necessary to maintain a modern industrial society. Many members of the public work at facilities or in industries generating these wastes. In general, the public wants to maintain the lifestyle that an industrial society provides and is therefore inclined to accept the risks associated with hazardous waste.

In contrast, for many people nuclear power represents complex technology that is difficult to understand and has not been seen as necessary by many for maintaining their desired standard of living (there are alternative sources for electricity generation). A 2005 Eurobarometer poll showed that disposal of radioactive waste was seen by many Europeans as a significant reason to oppose nuclear power. A majority of citizens in 16 of the (then) 25 EU countries said they would support nuclear power if the waste problem was solved, whilst a majority in only 8 countries would support nuclear with the waste issue unresolved. In addition, 92% of Europeans agreed that a solution for highly radioactive waste should be developed now and not left for future generations and 79% thought that the delay in making decisions in most countries means there is no safe way of disposing of highly radioactive waste.

These data clearly show the importance of the perceived risks of radioactive waste management and the impact of this perception on both the progress of implementing HLW/SF disposal facilities and on the acceptability of continuing or further expanding nuclear power generation. Support for nuclear energy will therefore be expected to increase when radioactive waste disposal facilities become available for HLW/SF.

Chapter 1

INTRODUCTION

Radioactive waste disposal, and in particular the inability of the nuclear energy community to establish any repository for high-level waste and spent fuel (HLW/SF) is one of the factors that significantly influence public and political acceptability of this energy technology. In many quarters the safe handling and disposal of radioactive waste is regarded as somehow uniquely difficult. The objective of this study is to consider radioactive waste in the wider context of the conventional hazardous waste disposal issues of a modern industrial society and in this way to allow a more balanced perspective of the issues involved. A second theme then also explores the waste issues associated with the probable future major low carbon release alternative electricity generating technology, coal fired generation equipped with carbon capture and storage.

Whilst the vast majority of civil (i.e. non-military) radioactive waste comes from nuclear power production, there are many other sources from medical, industrial and agricultural uses. Whether or not a particular country chooses to develop or continue with nuclear electricity generation, radioactive waste currently exists and needs to be appropriately managed and eventually disposed. The perspective presented here should help to put that need in context.

It is recognised that both radioactive and chemically toxic wastes are hazardous. However, throughout this document the term *hazardous* is used to describe wastes that are chemically toxic or carcinogenic but that are not radioactive. The term *radioactive* is used to describe wastes that are hazardous primarily because they emit ionising radiation. Some radioactive wastes contain chemically toxic substances (making them *mixed waste* in some countries). This additional complexity has not been directly addressed in this study, since the emphasis is on disposal, at which point the radioactive waste will be encapsulated in solid form.

1.1 Background

The current global waste production rate is 8 000-10 000 Mt/a (excluding overburden from mining and mineral extraction wastes), of which about 400 Mt/a is hazardous waste and about 0.4Mt/a is radioactive waste from nuclear power plants and their fuel cycle support facilities (excluding mining and extraction wastes). Protection of human health and the environment and consideration for future generations are key components of the principles for managing both radioactive and hazardous waste – it is clear that both waste types are generally well managed in OECD countries.

Nonetheless, there is ongoing debate globally about disposal of both hazardous and radioactive wastes (see appendices). Those countries having radioactive waste disposal capability have only a few near-surface facilities, whilst large numbers of hazardous waste landfills exist worldwide. Currently, there is no geological disposal facility in operation in the world for HLW/SF.[1]

Public acceptance plays an increasing role in the decision-making procedure for siting new waste disposal facilities and this depends heavily on risk perception, which is therefore an important consideration for decision makers. Societal acceptance of risk depends on perceptions of risk and benefit, and these perceptions are only partially based on scientific evaluations. The public generally perceives that both radioactive and some hazardous waste management are high-risk activities, recognising that the materials pose high inherent hazards and must be handled carefully to avoid injuries.

Direct comparisons between radioactive and hazardous waste management must be done very cautiously because the very different hazard characteristics of the two waste types require different processing techniques to assure safety. However, there are fundamental and essential similarities: both radioactive and hazardous wastes have the potential, if not managed appropriately, to cause environmental harm and to damage human health; for wastes disposed to a repository, the primary concern for both types is the risk presented by transfer to the biosphere through water transport.

However, there are many examples of hazardous wastes (including toxic and biohazard wastes) being treated and safely disposed (indeed, this is also true of radioactive wastes with the exception of HLW/SF). This demonstrates, at least in principle, that secure disposal of inherently dangerous substances can be achieved in properly designed facilities and that the public will accept their construction In the past, the nuclear energy industry has successfully capitalised on experience and lessons learned from other industries, for example in reducing nuclear power plant capital costs. It is to be expected that experience from the hazardous waste management sector might also be applicable to radioactive waste management, even though the two waste types are significantly different.

1.2 Objectives and scope

Against this background, the objective of this study is to provide a perspective on the current management of radioactive waste. The intended audience for this work is policy makers.

The study has two themes that draw comparisons between:

- Radioactive and hazardous wastes and their management strategies.
- Wastes coming from coal and from nuclear power generation, both of which technologies are likely to be major components of the global energy mix for the foreseeable future and which, with the potential advent of carbon capture and storage (CCS), have similar needs in terms of deep geological disposal of some of the arising wastes.

1.2.1 Theme 1 – Radioactive and hazardous wastes and their management strategies

The comparison between radioactive and hazardous wastes and their management strategies is intended to provide policy makers with a broad perspective on the similarities and differences between the waste types in the following areas:

- waste types: definitions, quantities and sources;

1. However, given the low volumes of waste, some three quarters of the radioactive waste from all sources so far generated has been sent for disposal.

- risks and hazards;
- ethics and management principles;
- legislation and organisation;
- waste management approaches before disposal;
- management and disposal options;
- licensing and safety assessment for disposal;
- costs and financing.

The scope of this theme is:

- the wide spectrum of solid hazardous wastes that arise in a modern industrial society;
- solid radioactive waste generated from civilian sources, primarily nuclear power production;[2]
- developments in the management of mercury containing wastes, used as an example of a particular hazardous waste stream.

This theme neither includes gaseous or liquid effluents nor waste from military uses of nuclear power.

1.2.2 Theme 2 – Wastes arising from coal and from nuclear power generation

This theme is intended to provide policy makers with a broad perspective on the similarities and differences between management of wastes from nuclear and from coal generation in the following areas:

- waste quantities;
- waste properties and disposal;
- recycling waste to extract economic value;
- impact on climate change;
- economic issues;
- development status;
- safety;
- regulation;
- stakeholder issues.

2. This report covers all types of radioactive waste generated in the civil nuclear fuel cycle and focuses in particular on the disposal of HLW/SF, which contains the vast majority of the radioactivity and is the most contentious. Wastes from the mining and milling of uranium ores are considered in terms of the quantities produced. The report does not deal with radioactive waste generated by military activities, although this is mentioned in some places for the sake of completeness. The report does not deal either with naturally occurring radioactive materials (so called NORM) which can be generated in significant quantities by other non-nuclear industries.

Nuclear power and coal generation with CCS are both seen in many countries as elements in a portfolio of technologies to reduce the impact of climate change. Comparison between wastes arising from coal and from nuclear power generation should not therefore imply that nuclear power and coal generation with CCS are necessarily in competition or mutually exclusive; it is likely that both will be needed in considerable quantities to achieve the necessary reduction in emissions of climate change gases**.** It should be noted that both nuclear power and CCS depend for success on the implementation of geological disposal for their waste products albeit that carbon dioxide is not considered to be a hazardous waste.

1.3 Exclusion: numerical comparisons of risk

In OECD countries, there has been a convergence of approaches to managing radioactive and hazardous waste over the past two decades with the hazardous waste industry now employing practices for final disposal developed for radioactive waste. However, no detailed numerical comparison between the risks associated with radioactive and hazardous waste has been made, primarily because the two waste types have very different hazard characteristics.

Both radioactive and hazardous waste facilities place strict requirements on construction standards of their engineered barriers and, depending on the nature of the facility, on the surrounding geology. Both also impose strict acceptance criteria for the disposed wastes. For radioactive waste it is then normal practice for the safety assessment to be extended to include a probabilistic analysis of the risk to the most exposed group at some varying time in the future, on the assumption that the engineered barriers will not provide perfect retention forever. Such analyses are enabled by the simpler range of wastes disposed and the assumption of a linear relationship between radiation dose and risk. For hazardous wastes the more complex positions with respect to the wastes disposed and the exposure/risk relationships means that reliance is placed on construction, acceptance and treatment standards and geology, and probabilistic risk analysis is not generally conducted. To date, very little international research has been conducted in this area and detailed evaluation of this matter is hence outside the scope of this study.

1.4 Report structure

The report consists of five chapters and six appendices.

Chapter 1, this chapter, introduces the report, providing background information on its objectives and scope.

Chapter 2 compares radioactive and hazardous waste management, under the headings shown in Section 1.2.1. A summary is provided in tabular form (Table 2.1) of the similarities and differences between both hazardous and radioactive wastes. A case study on the management of mercury as an example of a highly toxic hazardous waste is summarised in this chapter, which also discusses opportunities and challenges for both waste types.

Chapter 3 offers a broad perspective on the similarities and differences between management of wastes from nuclear and from coal generation, comparing the issues set out in Section 1.2.2.

Chapter 4 summarises the differences between "expert" and public perceptions of risk and the public's attitude to radioactive waste management.

Chapter 5 presents a concluding discussion for each of the two main themes and suggests some lessons that may be drawn from the study.

Appendices 1 and 2 describe the strategic issues for the management of radioactive waste and hazardous waste respectively in detail, providing an overview of the current management of these waste types. Although the two Appendices have the same general structure, the contents are treated differently. Appendix 1 provides information on radioactive waste management from an international perspective, augmented by a few national examples. Hazardous waste is described in Appendix 2 mainly using representative examples taken from Germany and the United States.

Appendix 3 presents case studies. These show how coal ash and carbon dioxide (as primary wastes from coal-fired electricity production) are managed, including a discussion on CCS. This Appendix also includes the detail or the case study of mercury waste, as an example of a highly toxic hazardous chemical waste.

Appendix 4 discusses risk, risk perception and the public's attitude to radioactive waste management, matters that are crucial for an understanding of how society sees and manages its waste.

These four Appendices contain comprehensive sets of references to which the reader is directed for further information. To make the report easier to read, these extensive references have not been reproduced in Chapters 1 to 5.

Appendix 5 presents a list of participants involved in the study from Belgium, the Czech Republic, Germany, Hungary, Italy, Japan, Korea, Russian Federation, Spain, Sweden, Switzerland and the United States, together with a representative from International Atomic Energy Agency (IAEA) and a hazardous waste expert from the OECD Environment Directorate. Appendix 6 provides a glossary of the acronyms used in the study.

Chapter 2

THEME 1 – RADIOACTIVE AND HAZARDOUS WASTES IN PERSPECTIVE

Detailed discussions on radioactive and hazardous wastes and their management strategies are presented in Appendices 1 and 2. The purpose of this chapter is to summarise the issues considered in those appendices, drawing comparisons between management strategies for the two waste types.

2.1 A comparison between radioactive and hazardous wastes and their management strategies

This Section addresses the first theme of the NEA study, comparing radioactive and hazardous wastes and their management strategies, and aims to summarise some of the similarities and differences between these two types of waste under the headings set out in Section 1.2.1. The section concludes by describing some opportunities and challenges for future management of these two waste types.

Direct comparisons between radioactive and hazardous waste management must be done very cautiously because the very different hazard characteristics of the waste types require different processing techniques to assure safety. However, there is a fundamental and essential similarity: both radioactive and hazardous wastes have the potential, if not managed appropriately, to cause environmental harm and to damage human health.

2.1.1 Definitions of waste types

Before considering similarities and differences, it may be helpful to summarise what is meant by "radioactive" and "hazardous" waste. More details are presented in Sections A1.1 and A2.1.1.

Radioactive waste

Radioactive waste is defined by IAEA as "any material that contains or is contaminated by radionuclides at concentrations or radioactivity levels greater than the exempted quantities established by the competent authorities and for which no use is foreseen".

Several classifications could be used to describe radioactive waste. The system adopted by IAEA[1] combines the type of radiation emitted, the activity of the waste and its half-life[2] to present an easy method of classification based on the following main categories:

- Exempt waste (EW): excluded from regulatory controls because radiological hazards are negligible.

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1. In late November 2008, after the text of this document had been prepared, the IAEA published a new Draft Safety Guide (DS390), in which it proposes six classes of radioactive waste.

2. Each radioactive element has its characteristic half-life ($t_{1/2}$), which is the time taken for half of its atoms to decay. In the classification scheme of IAEA two kinds of radioactive waste are distinguished: short-lived waste, whose predominant activity is defined by radionuclides with $t_{1/2} < 30$ years and long-lived one, where $t_{1/2} > 30$ years.

- Low- and intermediate-level waste (LILW): radioactivity levels are above those for exempt waste and the thermal power is below about 2 kW/m^3; IAEA recognises two sub-categories of LILW.[3]

  – Short-lived waste[4] (LILW-SL): primarily contains short-lived radionuclides, with long-lived radionuclide (including long-lived alpha emitter) concentrations restricted to an average of 400 Bq/g per waste package.

  – Long-lived waste[5] (LILW-LL): contains long-lived radionuclide concentrations that exceed limits for short-lived waste.

- High-level waste (HLW): contains sufficient concentration of radionuclides to produce heat generation that is greater than 2 kW/m^3; the typical activity levels are in the range of 5 x 10^4 to 5 x 10^5 TBq/m^3.

Some countries have different detailed interpretations of this classification method, in some cases based on acceptance criteria for national radioactive waste disposal facilities.

There are exceptions to most radioactive waste classification schemes for the following materials:

- mining and milling wastes: residues left from mining and extraction of uranium and other raw materials that contain naturally occurring radionuclides;
- environmental contamination: radioactively contaminated environmental media, such as soil and groundwater;
- spent nuclear fuel is considered as either a resource or a waste depending on which management strategy a country is using. See Appendix 1 for further details.

Hazardous waste

The OECD provides the following definition for waste: "wastes are substances or objects, other than radioactive materials covered by other international agreements, which: (i) are disposed of or are being recovered; or (ii) are intended to be disposed of or recovered; or (iii) are required, by the provisions of national law, to be disposed of or recovered". Hazardous wastes are also defined internationally elsewhere (e.g. Basel Convention and in European Union legislation), but in slightly differently ways to the OECD.

Hazardous waste can have a range of characteristics making it flammable, oxidising, corrosive, reactive, toxic or ecotoxic. Some examples of hazardous waste streams are wastes from the generation and use of biocides, wood preserving chemicals, organic solvents, polychlorinated biphenyls (PCBs). Some examples of hazardous constituents in waste are metal carbonyls, arsenic, cadmium, mercury, inorganic cyanides, acidic solutions or acids in solid form and asbestos. Hazardous wastes are often categorised and managed according to the nature of the hazard, although they may also be classified according to specific substances they contain or their origin (i.e., waste streams from a given industrial

3. In addition, some countries differentiate between low level and intermediate level waste on the basis of disposal site acceptance criteria.
4. Radioactive waste that does not contain significant levels of radionuclides with half-lives greater than 30 years, see *www-pub.iaea.org/MTCD/publications/PDF/Pub1155_web.pdf*
5. Radioactive waste that contains significant levels of radionuclides with half-lives greater than 30 years, see *www-pub.iaea.org/MTCD/publications/PDF/Pub1155_web.pdf*

sector or process). Different types of hazardous waste may exhibit one or several hazardous characteristics. For further details, see Appendix 2.

2.1.2 Comparison between radioactive and hazardous waste

The main similarities and differences between the two types of waste are summarised in Table 2.1. A brief description of the key similarities and differences is provided here. Further detail on these topics, together with comprehensive references, is provided in Appendices 1 and 2.

Quantities and sources

Globally, about 8-10 billion tonnes of waste are produced every year; this figure excludes wastes from mining overburden and the subsequent mineral extraction. Of this, about 400 million tonnes per year is hazardous waste. The current production rate of radioactive waste from nuclear electricity generation is about 0.4 million tonnes per year (excluding uranium mining and milling wastes): the current global generation rate of hazardous waste exceeds that of radioactive waste by three orders of magnitude. Further detail on the quantities of radioactive waste in the various classes referred to above is given in Appendix 1.

Whilst the vast majority of radioactive waste is produced by a relatively small number of easily identifiable generators (such as nuclear power plants, nuclear fuel facilities, etc.) hazardous waste is produced by tens of thousands of different generators in a range of industries that cover much of the industrial output of the developed world.

Risks and hazards

Radioactive waste has one primary hazardous characteristic: radioactivity, which can cause death or serious injury at high doses and has the potential to produce cancers in the longer term at low doses. Exposure to ionising radiation increases the risk of cancer in exposed persons in direct proportion to the degree of exposure. While debate continues with respect to exposure to very low levels of radiation, it is generally assumed that there is no threshold (the linear no threshold, LNT, assumption). The chemical toxicity of some radioactive elements (such as uranium) and of stable nuclides (e.g. lead) is also a potential source of hazard but usually to a much lesser extent than that associated with radiological characteristics.

Hazardous waste can contain a spectrum of hazardous characteristics such that the waste may be explosive, flammable, oxidising, poisonous, infectious, corrosive, toxic to humans or ecotoxic and can have short and long-term effects on human health and the environment. Physical hazards such as chemical reactivity, ignitability or corrosivity pose acute hazards only, although these can result in property damage, serious injury or even death if the wastes are mismanaged. Regarding longer-term hazards, a number of hazardous waste constituents are also carcinogenic, or cause non-cancer toxicity to different organs over long low level exposure periods, while many others have thresholds for toxicity below which exposure is expected to have no adverse effects.

In terms of the longevity of their associated risks, radioactive isotopes decay according to well-understood physical laws, each with a specific half-life. For HLW, the timescale for the radioactivity to decay to around the level of the original uranium ore is around 100 000 years whereas for LILW, many of the isotopes have half-lives less than around 30 years. Some hazardous wastes (e.g. some organic chemicals) biodegrade and their hazards reduce over time. However, other hazardous substances, like toxic heavy metals, do not change their toxicity over time. These wastes can thus be theoretically considered as having an infinite "half-life".

The risks from radioactive wastes are easily aggregated for a mix of nuclides (even for different types of ionising radiation) to allow a comprehensive view of the total risk. It is much more difficult to achieve such an assessment for hazardous waste because the different risks posed by the wide range of hazard characteristics are not necessarily additive.

Ethics and management principles

Protection of human health and the environment and consideration for future generations are key components of the principles for managing both radioactive and hazardous waste.

There are many internationally accepted principles that most countries adopt in developing management strategies for radioactive and hazardous waste. The principles are listed below and are set out in detail in Appendices 1 and 2. In practice, most of these principles are, in effect, used in managing both waste types. For example, although public participation is not included in the nine IAEA principles, it is widely recognised as essential in developing disposal facilities for radioactive waste.

The Basel Convention: The Principles of Toxic Waste Management

- source reduction;
- integrated life-cycle;
- precautionary;
- integrated pollution control;
- standardisation;
- self-sufficiency;
- proximity;
- least transboundary movement;
- polluter pays;
- sovereignty;
- public participation.

IAEA Safety Fundamentals:[6] The Principles of Radioactive Waste Management

- protection of human health;
- protection of the environment;
- protection beyond national borders;
- protection of future generations;
- burdens on future generations;
- national legal framework;

6. The IAEA safety principles are embodied in the Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management (see Appendix 1), which can be regarded as the equivalent of the Basel Convention for hazardous wastes.

- control of radioactive waste generation;
- radioactive waste generation and management interdependencies;
- safety of facilities.

This report does not address in detail the importance of legislative frameworks for the strengthening of radioactive waste policies, and more particularly, how international and national legal frameworks have evolved to reflect national priorities and policies. However, it is indeed noteworthy that one of the accomplishments of the international nuclear legal regime is the fact that there is a common definition of most concepts applicable to radioactive waste disposal strategies.

As regards radioactive waste, the use of the IAEA system of classification of radioactive waste – which is internationally accepted and which combines the type of radiation emitted, the activity of the waste and its half-life – is an illustration of the ongoing process of harmonisation of the radioactive waste management legal terminology that is reflected in national waste policies. The 1989 Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal has played a similar role in establishing a comprehensive global framework with regard to non-radioactive hazardous wastes and has helped governments to define a set of potential waste management strategies. The same holds true for the 1972 London Dumping Convention and the 1992 OSPAR Convention which have served as drivers to introduce international environmental management principles into national policies.

In law-making, countries usually first consider what are the most appropriate strategies and policies to accomplish their objectives. Once the strategies and policies have been established, national legal frameworks are developed to reflect those national priorities and policies, as illustrated for example by the issue of stakeholder involvement. Stakeholder involvement has been inspired by growing opposition of the public and an increasing number of judicial proceedings against nuclear installations. Conscious about these societal developments, there has been an evolution in governmental policies of OECD member countries from a process known as "decide, announce and defend" to a process whereby the public is informed about the risks and opportunities of nuclear energy and is allowed to participate in the decisions concerning, for example, site selection for RW/SNF facilities.

This resulted in an increasing number of legal frameworks that support access of the public to nuclear information "transparency of nuclear information", including on the safety of RW/SNF facilities, and the adoption of more developed mechanisms for stakeholder participation in decision making on RW/SNF facilities, often through environmental impact assessments (EIAs). Stakeholder involvement, which is generally perceived in OECD member countries as a necessary condition for public acceptance of waste management policies, depends to a large extent on international and national legal instruments that guarantee the respect of the populations' rights to information and participation.

Legislation and organisation

Typically, there is a high level of state intervention in radioactive waste management with only a small number of national organisations involved (see Appendix 1). Because hazardous waste has a wide range of producers across many types of industries, all levels of government tend to be involved in its management with distributed responsibility across federal, regional and local authorities. A diversity of administrative frameworks deals with hazardous waste management, which is largely market-oriented within a regulatory framework.

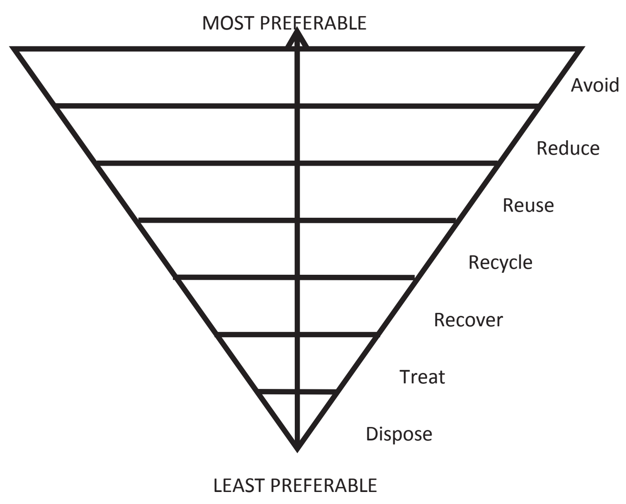
For both radioactive and hazardous waste, efforts are being made to achieve some degree of legislative harmonisation at an international level through, for example, international conventions and guidance.

Waste management approaches before disposal

Avoiding and reducing waste generation at source (waste prevention) is a primary aim of both radioactive and hazardous waste management. Figure 2.1 represents a commonly used waste hierarchy and shows the different means and options for managing wastes. For both waste types, the first objective is to minimise primary waste generation and to minimise the quantity and hazard of waste for disposal. However, the degree of applicability of the intermediate steps in the hierarchy is generally not the same for radioactive and hazardous waste.

In many industries, substitution of materials can often be used to avoid generation of certain hazardous wastes, but it is not generally possible to avoid radioactive waste generation in this way.

Figure 2.1: Typical waste management hierarchy



The wide range of materials in hazardous waste gives greater scope for recycling. For example, contaminated mercury or solvents can be distilled and used again, lead from automobile batteries can be smelted and reused, and incineration allows some wastes to be destroyed and the energy content of the waste to be recovered. Recycling is frequently adopted in hazardous waste management to maximise the use of available valuable resources and to minimise the risk from environmental harm. Although recycling of materials previously contaminated with radioactivity is feasible, it is rarely used. However, those countries that operate a closed fuel cycle recycle the uranium and plutonium recovered by reprocessing spent fuel (see Appendix 1).

For some hazardous wastes, a range of treatment options is available to reduce or eliminate hazards before disposal, e.g. incineration of toxic organic chemicals. However, the intrinsic hazard from radioactivity cannot be removed or reduced by chemical or physical treatment.[7] Similarly, the heavy metal hazard cannot be removed from hazardous wastes (although metals may be able to be recycled from the wastes). The main aim of radioactive waste treatment is to concentrate and stabilise the waste and minimise the probability of dispersion after disposal. Because radioactivity does decrease predictably with time, interim storage is often used to reduce the hazard before disposal by allowing short-lived radioactive materials to decay to significantly lower levels and by reducing heat loads that would otherwise necessitate special handling. Hazardous wastes are often stored to collect enough waste for the treatment process to be economic, but storage does not normally reduce the hazard.

Not all countries have the specialised facilities required to manage all kinds of hazardous wastes, therefore such wastes are moved between countries for pre-treatment. International regulations exist to manage transfrontier (transboundary) shipment of both radioactive and hazardous wastes. These movements occur regularly around the world to allow specialised treatment and disposal facilities to be used to manage specific hazardous waste streams. However, there is very little transfrontier shipment of radioactive waste, even for treatment, although spent fuel is regularly shipped between countries for reprocessing. In addition, spent fuel from some research reactors is shipped to other countries for storage and ultimate disposal to support non-proliferation efforts.

Management and disposal options

For both types of waste the best approach is to avoid their creation if at all possible. For wastes that already exist, the *concentrate and contain* option is used for radioactive waste (for short-lived radioactive wastes also the *delay and decay* option is practised), while the *eliminate or reduce the hazard* (incineration, chemical treatment etc) option is the primary strategy for hazardous waste. Less hazardous waste of both types is routinely disposed to landfills or near-surface facilities that depend mainly on engineered barriers to prevent adverse impacts on human health and the environment. In the case of higher activity or longer lived radioactive waste, safety relies on containment, isolation and multiple barrier concepts; in the case of hazardous waste, the elimination or reduction of hazard through effective treatment is the first option, followed by containment and isolation using multiple barriers.

Those countries with radioactive waste disposal capability have only a few near-surface facilities[8] to accommodate the relevant volumes of waste, whilst large numbers of hazardous waste landfills exist worldwide. Sweden and Finland have built underground disposal facilities for LILW. Geological repositories are also in licensing or under construction in Canada, Germany and Hungary.

The consensus in the scientific community is that disposal in stable geological formations is the best way to achieve the long-term management of long-lived radioactive waste. With a well designed and implemented geological disposal system, it is possible to achieve the required degree of isolation of radioactive waste from the biosphere, thus ensuring protection of human health and the environment without imposing undue burdens on future generations.

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7. It may be possible to reduce the quantity of certain radioactive isotopes by using transmutation technology. R&D on partitioning and transmutation technologies has been undertaken in some countries. However, much more R&D would be required to realise commercial utilisation.

8. However, as noted earlier, in volumetric terms, some three quarters of all of the radioactive waste from all sources generated since the start of the nuclear technologies has already been sent to disposal.

Currently, there is no geological disposal facility (repository) in operation in the world for civilian spent fuel and HLW. In the United States, a deep geological repository (WIPP) for long-lived defence-related transuranic waste with negligible heat generation is being operated near Carlsbad in New Mexico, United States. Three sites have been designated for construction of a geological disposal facility for HLW and spent fuel, at Yucca Mountain (a licence for construction authorisation was submitted to the US Nuclear Regulatory Commission in June 2008 and is currently under review), Olkiluoto (Finland) and recently at Forsmark in Sweden. Several other countries have officially announced their intention of achieving this solution in the near future, including Canada, France, Switzerland and the United Kingdom.

However, following the US 2008 Presidential election, the strategy for HLW disposal in the United States is under review. The Department of Energy's FY 2010 budget request identified the new Administration's intended termination of the Yucca Mountain repository project and it includes the funding needed to explore alternatives for nuclear waste disposal and to continue participation in the Nuclear Regulatory Commission's licence application process. All funding for the development of the Yucca Mountain facility and related infrastructure – such as further land acquisition, transportation access and additional engineering – has been eliminated. The DOE remains committed to meeting its obligations for managing and ultimately disposing of spent nuclear fuel and high level radioactive waste. To that end a "blue ribbon" panel of experts is being convened to evaluate alternative approaches.

In contrast, worldwide, final deep underground disposal is not a common management option for hazardous waste. However, in the United States, for instance, the deep well injection of liquid hazardous waste (which is not in the scope of this report), while conducted by only 3% of hazardous waste facilities, does account for almost 50% of all hazardous waste managed. Geological disposal is used in Europe, but not generally in other OECD countries, for disposal of extremely hazardous materials like mercury (see Appendix 3). Germany in particular has developed hazardous waste disposal facilities in salt domes and there is experience of using this medium in France and the United Kingdom.

For both hazardous and radioactive wastes, it is widely recognised that public participation in decision making related to waste disposal matters is essential. This matter is discussed further in Appendix 4.

Licensing and safety assessment for disposal facilities

This comparison of licensing and safety focuses on facilities for disposal. A licence and typically acceptance of an environmental impact assessment are required before construction and operation of either a radioactive or hazardous waste management facility is permitted. Statutory provisions and regulatory requirements mean that a safety assessment is required; for both waste types, the proposed site must be characterised before development of a disposal facility.

Achieving safety for a disposal facility, both during operation and after closure, is the paramount requirement in the licensing and regulatory system. For both hazardous and radioactive wastes, site specific Waste Acceptance Criteria are used to ensure that the characteristics of the waste and its package are compatible with requirements based on the safety assessment. Waste must be characterised before emplacement in a disposal facility to ensure that it meets acceptance criteria.

The long-term risk to human health from eventual migration of long-lived radioactivity from a waste disposal facility is usually calculated as the risk to a defined receptor (e.g. a hypothetical "most exposed” individual or group in the far future) of death from a radiation-induced cancer. Many countries have numerical limits or targets for this risk (or the equivalent in terms of an acceptable radiation dose), typically in the range of one in 100 000 to one in a million per year. For HLW and LILW-LL disposal sites, these quantitative safety assessments are typically performed for periods of up to one million years. For LILW-SL disposal, the safety assessments normally cover a few hundred years in recognition of the reducing radioactivity of the waste. The time taken into account in the safety assessment for underground hazardous waste facilities varies. In Germany, safety assessments cover periods of 10 000 to 50 000 years.

Institutional controls, including post-closure monitoring for (in the case of HLW) several decades at a minimum, are usually a central component in an acceptable safety case for radioactive waste disposal. These controls also help to address safety concerns over inadvertent or intentional human intrusion. Indeed, some concepts for deep repositories include plans for institutional oversight hundreds of years into the future. However, a central tenet of deep geological disposal is that its safety can be assured over very long times without relying on the continuation of monitoring or other interventions by future societies. Deep repository design philosophy is that safety is assured passively (i.e. without the need for further monitoring or intervention) once the repository is closed. Nevertheless, for the purposes of reassurance, institutional controls for radioactive waste disposal are foreseen in all OECD countries.

Hazardous waste landfills are monitored for typically 30 years after closure for gas and leachate evolution. After this time, and based on the monitoring results, the competent authorities decide whether the institutional period should be extended. An informal rule of thumb says that institutional control will be maintained for at least a century.

Some OECD countries require that future deep geological HLW/SF disposal facilities make provisions for retrievability, the ability to take the waste out of the disposal facility, sometimes even after closure, for reasons of safety (for instance, if observation results do not fit with the predicted values from modelling and simulation in the safety assessment) or otherwise (for instance, if techniques were to be developed to recover or recycle certain materials or if other significant treatment and disposal technologies were developed and demonstrated to be feasible). Nevertheless, the application of such retrievability concepts – and the degree to which they would be legally required – varies widely. The requirement may add significant complexity and effectively rules out some options acceptable for hazardous waste, such as deep well disposal.

No similar legal retrievability provisions are in place for hazardous wastes; such wastes are sometimes recovered from surface or shallow landfill disposal[9] to allow their constituent materials to be recycled when new industrial developments find cost effective means to do so.

Costs and financing

Most LILW radioactive waste management facilities have a limited range of acceptable waste forms for which they charge fixed rates. These rates typically depend on radioactivity level, dose rate, isotopic composition, volume, container weight, etc. Because hazardous waste can have so many hazard characteristics, it is difficult to provide typical costs since the fee varies hugely for different waste types and treatment options used. However, it is clear that waste management costs per unit mass are much higher for radioactive wastes than for hazardous wastes. The cost of HLW/SF disposal is estimated in the range of 300 000-600 000/tonne (400 000-800 000 USD/tonne at May 2009

9. Shallow landfill here includes underground near surface (a few tens of metres) disposal.

exchange rates). However, this high cost has little effect on the economics of nuclear power (total waste management costs for nuclear power stations are estimated at 0.04-0.16 US cents/kWh). Costs at this level per tonne would be unaffordable for most hazardous wastes. Examples from Germany indicate that underground waste disposal of some of the most hazardous wastes in salt rock are typically 250/tonne.

Both radioactive and hazardous waste management adopt the "Polluter Pays" principle. Hazardous waste management is generally carried out on a commercial basis with immediate payment for services provided. Facilities for managing radioactive waste are not always available nationally (no HLW/SF disposal facility is available globally) and funds are generally built up from electricity generation revenues to pay for future disposal. The facility may even be developed by the government and its costs may be pre-paid by the waste producer, not recovered through charging disposal fees. Therefore, not only the costs but also the entire funding and economic frameworks may be very different for the two waste types.

US regulations also require that hazardous waste treatment and disposal facility operators provide some form of financial assurance to support closure of the faculty at the end of its useful life. This may be in the form of a trust fund, a bond, a letter of credit, or by purchase of an insurance policy.

Table 2.1: Comparison between radioactive and hazardous wastes and their management

	Radioactive waste	Hazardous waste
Definitions, quantities and sources		
Definitions	• Both radioactive and hazardous wastes have internationally agreed definitions, with scope for national interpretation.	
Estimated global annual generation rate	• ~0.4 million tonnes from the nuclear power industry, of which 10 000 tonnes is HLW/SF (plus some 23Mt of lightly active milling wastes).	• 400 million tonnes (Excluding mining and mineral extraction wastes).
Main generation routes	• Primarily electricity production. • Other minor sources include healthcare, R&D and agriculture.	• Wide range of industries, including chemical, pharmaceutical, oil and gas, healthcare, mining, refining, steel and glass production.
Numbers of facilities generating and managing waste	Small number of waste generators and disposal sites (but note that disposal sites for LILW-SL exist in many countries and more than 75% of radioactive waste generated from all sources so far has been sent for disposal). *Example: United States* • 132 power stations (operational and shutdown); • 4 major disposal facilities.	Large numbers of generators and disposal sites or treatment centres. *Example: United States* • 16 000 large quantity waste generators;[10] • 600 treatment, storage, and disposal facilities; • Additional large numbers of small and medium generators.
Classification	• Internationally agreed classification systems exist for both waste types, and/or/specific country classification systems based on each country's statutory and regulatory framework.	
Risks and hazards		
Primary hazard characteristics	• Radioactivity (can induce cancers; probability proportional to dose). • Small quantities of wastes e.g. HLW/SF contain toxic heavy metals. • HLW/SF has potential for criticality hazard (minimised by design).	• Range of hazardous characteristics, including explosive, flammable, oxidising, poisonous, infectious, toxic – some hazardous waste have synergistic potentials. • Some hazardous waste constituents are carcinogenic.

10. Under the United States Resource and Conservation Recovery Act, large quantity generators are defined as those that generate 1 000 kg or more of hazardous waste per calendar month or 1kg or more per month of acutely hazardous waste.

	Radioactive waste	Hazardous waste
		• Some constituents have thresholds for toxic effects. • Many hazards can be mitigated or completely removed by proper treatment before disposal.
	• Health effects are generally long-term for both waste types: high exposures are required for acute effects and for immediate fatality.	
Exposure routes	• Inhalation, ingestion, external (including non-contact) exposure (also the relevant effect for criticality).	• Inhalation, ingestion, dermal. • External exposure to corrosive and reactive wastes, mechanical/thermal via explosion and/or fire.
Ease of hazard identification	• Easy to detect general radioactivity with low cost contamination and dose rate monitors; more specific characterisation might be more costly.	• Hazardous waste identification often requires complex, expensive laboratory analysis.
Evolution of risk	• All radioactive wastes reduce their hazard over time although, for HLW, the timescale for the activity to decay to around the level of the original uranium ore is in the order of 100 000 years. • Radioactive isotopes decay with known half-life according to well understood physical laws.	• Some organic wastes biodegrade and naturally reduce in hazard. • Some hazardous wastes (e.g. heavy metals) remain toxic indefinitely.
Ethics and principles		
Management principles	• Protection of human health and the environment and consideration for future generations are key components of principles for managing both radioactive and hazardous waste. • Most countries adopt internationally accepted principles to develop both radioactive and hazardous waste management strategies; the principles are, in effect, the same for both waste types.	
Legislation and organisation		
Legislation and regulation	• Both radioactive and hazardous waste management are subject to extensive national legislation and standards. • Both are also subject to international agreements.	
Organisational structures	• Organisation usually involves regulator, waste generator and implementer of waste management solutions; these three bodies are usually independent.	• All levels of government tend to be involved in hazardous waste management with distributed responsibility across federal, regional and local authorities.

	Radioactive waste	Hazardous waste
Organisational structures (cont'd)	• Typically, there is a high level of state intervention in radioactive waste management.	• A diversity of administrative frameworks deal with hazardous waste management, which is to a large extent market oriented.
Waste management approaches before disposal		
Waste minimisation	• Reducing waste generation at source is a primary aim of both radioactive and hazardous waste management.	
Substitution	• For new reactors, construction materials are chosen to reduce eventual radioactivity hazards caused by activation.	• Substitution of materials is commonly used to avoid or reduce certain waste hazards.
Reuse and recycling	• In some countries, material previously contaminated with radioactivity is recycled and reused; in other countries this is less often done, primarily because of public concerns. • Limited quantities of wastes, e.g. from decommissioning, have been recycled into the nuclear industry as shielding materials. • Spent fuel (which is not a waste until so declared) can be, and is routinely, recycled by reprocessing to recover and reuse its fissile content.	• Reuse and recycling are frequent to maximise use of available valuable resources and to minimise the risk for environmental harm. • The wide range of materials in hazardous waste gives increased scope for recycling. • Incineration allows the energy content of waste to be recovered. • Removal of hazardous wastes from surface or near surface repositories is sometimes done to recycle materials contained in the waste back into the economic cycle.
Treatment before disposal	• The intrinsic hazard from radioactivity cannot be removed or reduced by incineration or chemical treatment before disposal. • Storage (see below) for decay. • The main aim of treatment is to concentrate the waste and minimise the risk for dispersion after disposal.	• A range of treatment options is available to reduce or eliminate hazard before disposal, e.g. incineration of organic chemicals or ignitable waste. • Hazardous waste treatment also uses chemical methods to destroy or reduce the hazard or to concentrate the material.
Storage	• Interim storage, sometimes for several decades, is often used to reduce radioactivity by allowing it to decay before disposal. • This reduces the potential environmental hazard and the radiation dose to power plant and disposal facility operators.	• Hazardous wastes are typically allowed to be stored for up to one year to collect enough waste for the treatment process to be economical. • Long-term storage is not typically permitted; it does not, in most cases, reduce the hazard.

	Radioactive waste	Hazardous waste
Transfrontier/ transboundary shipment	• International regulations exist to manage transfrontier shipment of both types of waste.	
	• There is little transfrontier shipment of radioactive waste • Some spent fuel is moved between countries for reprocessing; in general contracts require the resultant HLW to be returned to the country of origin.	• Transfrontier shipments occur regularly around the world to allow specialised treatment and disposal facilities to be used to manage specific hazardous waste streams.
Disposal options		
Public participation	• An important factor in assessing waste disposal options is the public's perception and acceptance of risk; therefore, public participation in decision making related to waste disposal matters is essential.	
Options and experience for disposal of lower risk wastes	• For both waste types, containment of waste is achieved primarily through stabilisation of the waste forms and engineered barriers.	
	• In case of radioactive waste, safety relies on containment, isolation and multiple barrier concepts until radioactive decay removes the risk. • Countries with radioactive waste disposal capability have only a few near-surface disposal facilities (although waste volumes are significantly smaller than those for hazardous waste). • Sweden and Finland have built underground disposal facilities for LILW. • A deep geological repository (WIPP) for long-lived defence-related transuranic waste is being operated in the United States.	• Large numbers of hazardous waste landfills exist worldwide. • In the case of hazardous waste the elimination or reduction of hazard is the first option, followed by isolation and containment using multiple barriers.
Options and experience for disposal of higher risk wastes	• The consensus in the scientific community is that disposal in stable geological formations is the best way to achieve the long-term management of long-lived radioactive waste. • Many countries plan to develop deep facilities to dispose of HLW/SF; Finland, Sweden and the United States[11] have chosen sites for their HLW/SF disposal facilities.	• Deep underground disposal is not a normal management option for solid hazardous waste. • Geological disposal is used in Europe, for disposal of extremely hazardous materials like mercury. • Germany in particular has developed hazardous waste disposal facilities in salt rock, there is experience also in France and the United Kingdom. • Geological disposal is not generally used in other OECD countries, although deep-well injection for hazardous liquids is used in the United States.

11. However, the United States will now be evaluating alternative approaches for its waste management programme.

	Radioactive waste	Hazardous waste
Options… (cont'd)	• There is significant salt rock engineering experience through use of this geology to store hydrocarbon fuels (as new products, not wastes).	
Licensing and safety assessment for disposal		
Licensing	• In OECD countries a licence is required from the competent authorities to construct and operate any radioactive or hazardous waste management facility, including a disposal facility. • The licensing process checks that the safety assessments are technically and scientifically correct and sufficient so that protection of the public and environment can be reasonably assured.	
	• In some OECD countries (e.g. Hungary, Finland) preliminary legislative authorisation for underground waste disposal facilities must be obtained before an application is made for a construction permit.	• Preliminary legislative authorisation is not normally required for hazardous waste disposal facilities.
Safety assessment	• Statutory provisions and regulatory requirements mean that for both waste types extensive safety assessment is required before disposal sites can be built and operated. • The assessment of risk typically requires identification and assessment of: o the disposed waste and its containment within the disposal facility; o the pathways by which substances from the wastes may reach the biosphere; o the impact on human health and the environment of substances that may reach and be transported through the biosphere. • Safety relies on containment, isolation and multiple barrier concepts. • The proposed disposal site must be characterised before development and each batch of waste characterised before disposal to ensure that it meets acceptance criteria.	
Risk assessment and facility risk target	• Achieving safety during all phases of the lifecycle of a disposal facility, including after its closure, is a paramount consideration in the licensing and regulatory system. • The regulations applicable to radioactive waste and to hazardous waste share the requirement of developing a safety case for the consideration of the regulator before a license can be granted. • In both cases, the safety case is an integration of arguments and evidence that describe and substantiate the claim that the disposal facility will be safe during operation and after closure and beyond the time when reliance can be placed on active control of the facility. For radioactive waste a quantitative risk assessment is normal practice as part of such substantiation.	
	• Many countries have numerical limits or targets for risk (or the equivalent in radiation dose terms) typically in the range of one in 100 000 to one in a million per year . • In some OECD countries, for HLW and LILW-LL disposal sites,	• Numerical calculations of probabilistic risk are not normally feasible. Safety is based on construction, waste acceptance and treatment standards, geology and monitoring.

	Radioactive waste	Hazardous waste
Risk… (cont'd)	these quantitative safety assessments are typically performed for time periods of about 10 000 up to 1 million years. • For LILW-SL disposal, the safety assessments normally cover a few hundred years (in recognition of the increased radioactive decay rate of the waste).	• The time period taken into account in the safety assessment for underground hazardous waste facilities varies; in Germany safety assessment cover periods of 10 000 to 50 000 years.
Waste acceptance criteria	• Site specific Waste Acceptance Criteria are used to ensure that the characteristics of the waste and its package are compatible with requirements based on the safety assessment. • The assessment may be subject to restrictions, for example, that certain hazardous wastes will not be disposed to the facility, or that the quantity of radioactivity in the waste is below a defined level.	
Disposal site post-closure monitoring and institutional control	• Institutional controls, including post-closure monitoring, for (in the case of HLW) centuries are usually a central component in a safety assessment for radioactive waste disposal. • These controls also address safety concerns over inadvertent or intentional human intrusion. • Institutional controls are foreseen in all OECD countries.	• Hazardous waste landfills are typically monitored for a minimum of 30 years after closure for gas evolution, leachates, etc. • After this time, and based on the monitoring results, the competent authorities decide whether the institutional period should be extended. • Many landfill experts expect that administrative control would be prolonged for at least a century.
Retrievability	• Some OECD countries considering deep geological disposal of HLW/SF have legal provisions for retrievability – the ability to take the disposed waste out of the disposal facility; others are considering this possibility.	• No similar legal provisions are in place for hazardous wastes; such wastes are sometimes recovered from surface or near surface disposal facilities to allow their materials to be recycled when new industrial developments find cost effective means to do so.
Costs and financing		
Costs	• Disposal of HLW/SF is estimated in the range of 300 000 to 600 000/tonne (400 000-800 000 USD/tonne at May 2009 exchange rates). • Many waste disposal facilities charge fixed fees, which depend on radioactivity level, dose rate, isotopic composition, volume, container weight etc., for a limited range of acceptable waste forms.	• Because hazardous waste can have so many hazardous characteristics, it is difficult to provide representative costs. • The fee varies hugely for different waste types and treatment options used. • Examples from Germany indicate that geological waste disposal of some of the most hazardous wastes in salt rock are typically 250/tonne.

	Radioactive waste	Hazardous waste
Financing	• Both radioactive and hazardous waste management adopt the "Polluter Pays" principle.	
	• Facilities for managing radioactive waste are not always available nationally (no HLW/SF disposal facility is available globally) so future financing is required. • Funds are generally built up from electricity generation revenues to pay for future disposal: United States levies the equivalent of (per kWh) USD 0.001 (0.0008), Sweden SEK 0.01 (0.001) and Japan Yen 0.13 (0.001).	• Industrial hazardous waste management is normally carried out on a commercial basis with immediate payment for services provided. • Some nations require financial assurance (e.g., a bond or insurance) for disposal facility closure.
	• Not only the costs but also the entire funding and economic frameworks may be very different for the two waste types.	

2.1.3 *The management of mercury waste – A case study*

Mercury is an example of a highly toxic, hazardous chemical. The case study presented in Appendix A3.2 describes the production rates and sources of mercury and explains some of its hazard characteristics. The aim is to present a perspective on the management and eventual geological disposal of highly toxic mercury waste streams.

The annual global contribution to the mobilised pool of mercury has been estimated as 13 500 tonnes. To provide a perspective, this amount is in the same order of magnitude as the annual global HLW/SF arising from the world's nuclear power plants. Because the hazard from mercury does not diminish with time, when it is disposed of it must be isolated from man and the environment, effectively forever. In order to cope with safety requirements over long periods, without the need for monitoring and intervention, the trend for managing mercury waste is towards deep disposal. The isolation needed for mercury wastes is therefore of a similar nature to, but even more demanding than those for high-level radioactive waste.

Mercury waste provides a useful comparison with radioactive waste in that:

- It has a significant health impact if inappropriately managed.
- Mercury and mercury containing compounds will always remain toxic: they are typical of hazardous chemical substances requiring long term safe management and disposal – in this sense they present similar challenges to the management of radionuclides of especially long half lives.
- In a number of countries the management of mercury and similar wastes has adopted the same route as that proposed for long-lived radioactive waste: deep geological disposal.

Health effects

Mercury and its compounds can have a significant impact on health on local, regional and global scales since it can be highly toxic to humans, ecosystems and wildlife. High doses can be fatal, but relatively low doses can also have serious adverse impacts to the developing nervous system. There are indications of possible harmful effects on the cardiovascular system and the immune and reproductive systems, although there are exposure thresholds below which no adverse health effects are expected to occur. Mercury has not been found to be carcinogenic. Possible routes for intake and damage are connected to its chemical form, methyl mercury being the most hazardous.

Inappropriate management of mercury has caused a variety of significant impacts on human health and the environment throughout the world. As examples, the Minimata disease in Japan was caused by spilled mercury that bio-accumulated in fish and other seafood, a main source of food for local people; 3 000 people were affected. In Iraq mercury poisoning affected some 6 000 people due to consumption of seed that had been treated with fungicides containing mercury.

Management of wastes containing mercury

Some mercury can be recovered from waste for reuse. While many devices that have typically used mercury have been replaced with mercury-free alternatives (e.g., thermometers, switches, medical devices such as sphygmomanometers), there remain some legitimate uses for mercury, such as in lamp manufacture. Recovery and reuse of the mercury can reduce mining of new mercury to supply these needs. The US waste regulations require mercury recovery for reuse from wastes containing more than 260 mg/kg mercury.

A programme on mercury waste and its environmentally sound management is being carried out under the Basel Convention, including production of draft technical guidelines to facilitate safe management. The United Nations Environment Programme is carrying out a comprehensive programme to understand mercury issues with a view to reducing risks for humans and the environment. The EU also has a strategy which includes looking for long term disposal solutions.[12]

Disposal strategies and technologies currently differ significantly between countries. Waste containing mercury has been disposed in specially engineered landfill, underground caverns and near surface pits. Increasingly there is a trend to its disposal deep underground in stable geological formations. In 2005, Sweden was the first EU country to pass legislation requiring deep geological disposal for all waste with mercury content above 0.1%. Sweden is currently building a disposal facility in granite rock connected to a deep mine. Deep geological disposal of long-lived hazardous wastes is currently carried out in deep (700 m) salt formations in Germany where four mines are in use. Facilities are being developed in several countries to allow the long-term safety without the need for monitoring and intervention.

Mercury and its compounds are highly toxic and present risks to human health and the environment over long periods that require some precautions that are similar in some ways to those needed for long-lived radioactive waste, particularly safe permanent disposal. The parallels with HLW/SF management are clear, but for these toxic waste streams faster progress to implementation has been possible.

2.1.4 *Opportunities and challenges*

Hazardous waste management

Hazardous waste management options are assessed by use of the waste management hierarchy and waste management principles. The primary requirement is to avoid or minimise waste generation. If waste cannot be avoided it should be reused, recycled or recovered so far as practicable. Only if this is not possible should the option of disposal, after pre-treatment if necessary, be used. At all stages in the process adequate facilities must be available for waste treatment, recovery and disposal to protect human health and the environment. Some hazardous waste (such as mercury) needs to be isolated from the biosphere for geological time. In some countries, these are disposed of in deep disposal facilities of suitable geology which are similar to those envisaged for HLW/SF. The issue of retrievability of deep geologically disposed wastes has not arisen for hazardous materials.

A modern waste management system can only be effective if those responsible for the generation of hazardous waste accept responsibility for, and bear the costs of, its management and disposal. Consequently, waste generators from trade and industry are required to accept responsibility for the management and disposal of their hazardous waste. However, household hazardous wastes are generally exempted from this rule, since municipalities normally include these costs in household taxes.

12. Regulation (EC) No. 1102/2008 was published in October 2008, after the text of this study had been prepared. This requires that waste metallic mercury is to be stored in salt mines or deep underground hard rock formations providing an equivalent level of safety.

Opportunities for improving hazardous waste management might include development of a vision of sustainability that could serve as long-term guidance for development of hazardous waste policies. Over the last few decades, hazardous waste management has been dominated by first recovering material and energy as far as possible and second by developing environmentally sound management strategies for remaining residues. The challenge in the future is to regard waste as a resource that should be used efficiently while at the same time preventing release to the environment. This new challenge may also include the retrieval of waste disposed of in the past.

Previously, R&D was carried out to develop waste treatment and disposal techniques and to develop cleaner management methods. In the future, R&D is likely to focus on enhancing resource use efficiency, substituting non-hazardous materials for hazardous materials when producing goods, and retrieving previously disposed wastes for recycling. The goal is to move from waste management to resource management.

Radioactive waste management

Public acceptance is judged to be the primary challenge now and into the future, especially for geological disposal of HLW/SF. The NEA Radioactive Waste Management Committee (RWMC) has already noted

> "…confidence by the technical community in the safety of geological disposal is not, by itself, enough to gain public confidence and acceptance. There is consensus that a broadly accepted national strategy is required. This strategy should address not only the technical means to construct the facility but also a framework and roadmap allowing decision makers and concerned public the time and means to understand and evaluate the basis for various proposed decisions and, ultimately, to gauge whether they have confidence in the level of protection that is being indicated by the implementing organisation and evaluated by the regulator through its independent review."

Other near term challenges fall into four categories: technology, legislation, policy making and regulatory concern. In the area of technology, radioactive waste management has sufficient scientific and technical knowledge and experience safely and reasonably to fulfil its goals. Nevertheless, the implementation of waste management solutions will always be accompanied by uncertainties that can be reduced by further R&D. Knowledge retention, for example of waste and facility characterisation and facility operation, will be an important challenge into the institutional control period.

In the area of legislation, there is a consensus that radioactive waste management is an issue that is being adequately addressed in OECD countries. Legislation requires progressive adaptation to new societal situations and technical developments, basically arising from the expected implementation of national policies on HLW/SF disposal. In this context, a key issue will be the legislative and regulatory definition of the concepts of reversibility and retrievability of a repository. Again, in the words of NEA RWMC

> "…reversibility and retrievability are considered by some countries as being important parts of the waste management strategy… There is general recognition that it is important to clarify the meaning and role of reversibility and retrievability for each country, and that provision of reversibility and retrievability must not jeopardise long-term safety."

There can be no doubt that the regulatory framework for radioactive waste disposal is clear, well established and comprehensive. There is a widespread perception, however, that radioactive waste management (like energy policy overall, and policies regarding nuclear power in particular) would benefit from more continuity and stability on the part of decision makers and greater independence from day-to-day political concerns. This would be expected to allow better use of allocated resources and result in reduced implementation timescales, although continuing public concerns about radioactive waste disposal make it very difficult for political decision makers to disregard shorter term political concerns.

Disposal of LILW is an internationally tested practice either in surface facilities or in deeper repositories. There is considerable regulatory experience in this area that has been shared and contrasted in international organisations like the NEA and the IAEA and that is helping countries that are new to LILW repositories. However, no underground repository for HLW/SF has yet been licensed and although the first application was for such a facility was submitted in June 2008 by the US DOE for the Yucca Mountain repository the US will be evaluating alternative approaches for its waste management programme. The complexity of the documentation involved in the submissions for this type of facility is considerable.

Chapter 3

THEME 2 – THE OUTLOOK FOR WASTES ARISING FROM COAL AND FROM NUCLEAR POWER GENERATION

This chapter addresses the second of the themes considered in this report. This is regarded as an important consideration in that society's need for electricity has to be satisfied. There is a choice to be made with respect to the balance of technologies that meet this need whilst recognising the constraints imposed by the need to avoid climate change. As will be seen in Chapter 4 and Appendix 4, radioactive waste disposal is a key factor in the public's antipathy to nuclear energy. Diminishing the role of one technology because of a disadvantage (in the case of nuclear energy, the need to manage radioactive waste) without considering the equivalent disadvantages of any replacement will not lead to a rational decision. There are, of course, many other factors than just waste issues in making such a technology choice, but here waste is the focus. In practice, meeting the necessary CO_2 reduction targets identified by organisations such as the Intergovernmental Panel on Climate Change (IPCC) will be extremely challenging and both CCS and nuclear energy are likely to be needed in significant quantities.

In 2005, about 40% of the world's electricity came from coal and 15% from nuclear generation. The wide availability of coal means that it will continue to be used and projections suggest that its use will increase significantly as world energy demand continues to grow; globally, coal and nuclear are expected to be two of the primary sources of base load electricity in the future. It is therefore of considerable interest to put radioactive waste from nuclear generation into perspective with wastes from coal generation. A typical 500 MWe coal fired power plant burns about 2 Mt/a of coal and around 3.2 Gt of coal is used for electrical power generation per annum globally. In order to avoid the serious environmental damage that will result from climate change the technologies of carbon capture and storage (CCS) are being developed for coal and other stationary large scale fossil fuel use. The objective of these developments is to capture the carbon dioxide produced in combustion, compress it and transport it to suitable geological formations for deep underground disposal as a supercritical fluid.

The aim of this chapter is to provide a broad comparison between the management of wastes from coal and from nuclear power production. Coal ash and carbon dioxide are the main waste products from combustion of coal to generate electricity; current management of ash and possible future management of CO_2 via CCS are discussed in Appendix A3.1 and A3.3 where further details, including references to the matters discussed here, can be found. A detailed discussion of radioactive waste management can be a found in Appendix 1. Similarities and differences between these two types of waste are summarised overall in the following areas:

- waste quantities;
- waste properties and disposal;
- recycling waste to extract economic value.

The hazardous nature of radioactive waste, if not appropriately managed, is well recognised in society. However, as described elsewhere in this report, it is produced in relatively low quantities and a management philosophy of concentrate and contain is practicable. CO_2 is not regarded as a hazardous waste and at low concentrations is not dangerous, but it is produced in very large quantities and it is recognised as the major contributor to global warming. Coal ash contains a number of hazardous materials at low levels, but it is produced in such large quantities that the sum total entering the wider environment is significant and again concentrate and contain is not a practicable approach. Hence the two technologies present very different waste management challenges.

The scope of this study does not include detailed comparison between the health and environmental consequences of disposal of waste products from coal and nuclear generation. Because nuclear power and CCS are both generally seen as means to reduce the impact of climate change and both are likely to be necessary in significant quantities, further paragraphs are intended to paint a general comparison between these two technologies as follows:

- impact on climate change;
- economic issues;
- development status;
- safety;
- regulation;
- stakeholder issues.

As noted in Section 2.1, direct comparisons between the management of radioactive and other waste must be done very cautiously because of the very different characteristics of the waste types. However, there is again a fundamental and essential similarity: all wastes have the potential, if not managed appropriately, to cause environmental harm and to damage human health. In the case of coal generation, these adverse impacts might result from the effects of climate change caused by CO_2 emissions from combustion.

3.1 Waste similarities and differences

Waste quantities

- Globally, generation of electricity from coal produces about 0.6 Gt/a (90 kt/TWh) of ash and 10.5 Gt/a (1 600 kt/TWh) of CO_2. Nuclear power generation produces < 0.0005 Gt/a (< 0.2 kt/TWh) of solid[1] waste (including accounting for decommissioning wastes that will eventually arise from the currently operating facilities, but excluding mining and milling wastes which are addressed below), ranging from HLW/SF to VLLW.
- Both coal and nuclear power generation produce additional wastes from fuel mining and primary production processes. For nuclear this is < 0.025 Gt/a (< 8 kt/TWh) of lightly radioactive milling wastes and a similar quantity of non-active mining wastes (in total 0.045Gt/a and 15 kt/TWh) and for coal 20 Gt/a (3 000 kt/TWh) of wastes from mining and primary production.

1. In addition, nuclear power stations produce gaseous and liquid wastes that are typically filtered and, in the case of liquids, subject to ion exchange treatment before being discharged under authorisations granted by regulatory bodies. The annual production rates of these wastes are small.

- Coal energy generation produces waste (including mining and CO_2) at a rate per unit energy that is about 300 times higher than does nuclear (including mining and milling); however, in most countries coal generation wastes are not classified as hazardous.

Waste properties and disposal

- Some of the waste products from coal energy generation are disposed to the environment and some are recycled. There is significant global concern about the climate change effects of CO_2 emissions from fossil fired electricity generation, which is the largest single contributor by far to anthropogenic releases to the atmosphere.
- However, other releases also have significant detrimental effects. Air pollution from coal-fired electricity production includes a mixture of pollutants, including fine particulate matter, carbon monoxide, nitrogen dioxide, sulphur dioxide, ozone and volatile organic compounds and inorganic species.
- Air pollution control systems in modern coal fired power stations may include a scrubber system where most residues of sulphur and nitrogen oxides are removed, together with hydrochloric acid. Volatile species like mercury and cadmium are released, to some extent, into the atmosphere along with fluorine, chlorine and bromine. Estimates of global release rates from coal-fired generation include: mercury 210 t/a, bromine 22 000 t/a, fluorine 320 000 t/a and chlorine 990 000 t/a.
- A European Environmental Agency study shows that 30% of the total PM_{10} (particles less than 10 microns in diameter) emissions in Europe result from energy production. It states that coal is a significant emitter of PM_{10} during electricity production, and should therefore be considered a significant source of health damage worldwide, even in advanced economies. The OECD Environmental Outlook estimates that PM_{10} emissions caused 960 000 premature deaths in 2000, with 9.6 million years of life lost worldwide.
- Heavy metal concentrations in coal ash average 120 ppm with highest values up to 375 ppm. Coal ash also contains small amounts of carcinogenic organic compounds such as polycyclic aromatic hydrocarbons and dioxin.
- Coal ash has average radioactivity concentrations ranging from 157 Bq/kg in the United Kingdom to 500 Bq/kg in Poland. Maximum radioactivity concentrations of 2 900 Bq/kg have been reported. In some countries it is possible that coal ash could have a specific activity that exceeded national radioactivity *de minimis* levels if the ash had not been exempted. Solid residues from coal-fired electricity generation that are not recycled (see below) are generally sent for landfill. In the United States this amounts to about 85 Mt/a and in Europe about 7 Mt/a, excluding mining waste.
- With respect to radioactive wastes, about 0.3 Mt/a is LILW-SL and < 0.1Mt/a is LILW-LL, from current nuclear power production. Most countries with nuclear capability have disposal facilities for short lived waste of this type but not currently for long lived ILW. About 10 000 t/a is HLW/SF for which no disposal facilities are currently available.
- Current generation of radioactive waste from decommissioning nuclear power production facilities is quite small but accounting for the eventual wastes over assumed 40 year lives gives figures of committed waste of 0.05Mt/a of LILW-SL and 0.01MT/a of LILW-LL. It is possible that some of this material could be recycled and a considerable quantity of this waste will be VLLW.

Recycling waste to extract economic value

- In the United States, about 35% of the solid residues from coal-fired electricity generation are recycled (46 Mt/a) whilst in the former EU15 the figure was about 88% (53 Mt/a). Because so much coal ash is reused, to replace significant volumes of virgin raw materials, the distinction between a waste and a product is not as clear-cut as it is for radioactive waste.

- CCS, when available, may be capable of recycling some CO_2 as a means of increasing oil extraction.

- Some spent nuclear fuel is recycled to extract uranium and plutonium for future fuel manufacture. Some radioactively contaminated waste, mainly from decommissioning, is decontaminated and recycled.

3.2 Climate change considerations

Nuclear power and coal generation supplemented by CCS are both generally seen as means to reduce the impact of climate change. The following paragraphs are intended to paint a general comparison between the two technologies in a number of areas. References to the CCS matters discussed here are contained in Appendix A3.3 and are not repeated in this chapter. In line with current practice in the carbon capture and storage community, the word "storage" is used here. It is interesting to note the contrast with the terminology used in radioactive waste management where "storage" always implies an intention to retrieve and where, if there is no intention to retrieve, the word "disposal" is used.

Impact on climate change

- The Intergovernmental Panel on Climate Change considers that both carbon capture and storage and nuclear power have the capability to reduce annual greenhouse gas emissions. IPCC estimate that CCS applied to coal generation would reduce emissions by 0.49 Gt CO_2 eq by 2030; nuclear energy could reduce emissions by a further 1.9 Gt CO_2 eq beyond the 1.4 Gt CO_2 eq anticipated in the International Energy Agency's (IEA) *World Energy Outlook 2009*.
- CO_2 has been injected into oil reservoirs for almost 40 years to enhance oil recovery without detectable losses of CO_2 over these timescales. However, measurement accuracy is insufficient to provide confidence for CO_2 retention in the longer term. If there were to be long-term leakage, the impact on climate change would simply be deferred rather than eliminated. A key issue for investors will be the extent of their liability for long-term monitoring and potential remediation.
- A power plant equipped with CCS would need 10-40% more energy than an equivalent plant operating without CCS. The additional energy requirement will itself produce CO_2 so a power plant with CCS should reduce CO_2 emissions to the atmosphere by approximately 80-90% compared to a plant without CCS.

Economic issues

- Like nuclear power, CCS requires a significant up-front investment so the technology may only be suitable for large producers of CO_2.

- IPCC estimates show that CCS could increase the cost of electricity by between 22 and 60%. Generation III and III+ nuclear reactors (including the costs of waste management and decommissioning) that are currently being built are designed for being broadly competitive with coal-fired generation that includes a modest carbon constraint, however such a constraint would be unable to off-set the full cost of CCS.
- Experts agree that the cost of radioactive waste disposal, which is technically achievable, would add little to the economic cost faced by investors (mainly because the costs will accrue only decades after the building of the plants) or electricity consumers. The costs for CCS, most of which accrue already at the moment of construction, will instead be a substantial part of the total costs of generating electricity.

Development status

- Storage of natural gas in underground formations has been practised for around 100 years while CO_2 injection for the purpose of enhanced oil recovery has been performed for almost 40 years.
- Only one operational project is currently attempting to demonstrate both carbon capture and storage. This is a 30 MWe coal-fired plant near Spremberg in Germany where CO_2 is collected, compressed and trucked 350 km to an empty gas field for injection.
- The EU Zero Emission Fossil Fuel Power Plants programme aims to have up to 12 large-scale CCS projects operational by 2015 to demonstrate commercial viability by 2020.
- Large numbers of commercial nuclear power stations are in operation and others, including modern Gen III/III+ nuclear plants, are under construction. The commercial viability of nuclear power has been demonstrated. While there is no operating geological repository for SNF or HLW, the feasibility of the technology has been demonstrated with other facilities (the WIPP repository, for example) and is supported by extensive, decades-long research programmes, including a number of underground research laboratories.

Safety

- Both coal with CCS and nuclear power rely on deep geological disposal as their waste management solution, in the case of nuclear power as a stabilised solidified product and for CCS as a supercritical fluid. Coal using CCS technology would produce about 40 000 times more waste per unit of electricity produced that required geological disposal than does nuclear power, even assuming that all LILW-LL will need to go to deep disposal.
- In developing risk assessments, CCS has used safety assessment methodologies developed for radioactive waste disposal. Risk assessments for CO_2 injection for enhanced oil recovery are currently used in the oil industry.
- Radioactive waste disposal combines engineered and natural barriers to contain the radionuclides encapsulated into a solid matrix; CCS uses only natural barriers to contain the supercritical fluid, except for the seal to the injection well.
- Long-term impacts of radioactive waste disposal are assessed against well-defined numerical limits and constraints imposed by regulators. There are no generally adopted measures of health detriment for CCS risk assessments. See Appendix A3.3 and its references for more information on this matter.

Regulation

- CCS is a new technology and regulation is evolving. Although the regulation of HLW/SF disposal has been under consideration for many years, it also is still continuing to evolve (e.g. on the issue of retrievability).
- US regulations cover well siting, well construction, well operation, and well closure; over 800 000 regulated wells have injected fluids over the past 30 years. This experience could help inform the basis of regulations for CCS.
- For radioactive waste management, international conventions outline common principles. National programmes in NEA countries pursuing geological disposal provide a clear regulatory authority and framework for disposal, and comprehensive safety criteria have been established in many countries.

Stakeholder issues

- Experience from both the nuclear and hazardous waste industries suggests that public acceptance will be crucial if CCS is to progress. However, the largest current CO_2 storage projects do not yet have public acceptability as part of their remit.
- As examples of the views of non-governmental organisations (NGOs) on CCS technology, Friends of the Earth International classes both CCS and nuclear energy as "unsustainable technologies" and Greenpeace International opposes the application of CCS to coal-fired power stations as a means to combat climate change. References to these NGO views are contained in Appendix 3.
- The Environment Agency in England and Wales states "new and replacement coal-fired power stations should only be permitted where they are capable of capture and storage of carbon dioxide".
- Experience in national radioactive waste disposal programmes has shown that confidence by the technical community in the safety of geological disposal is not, by itself, enough to gain public confidence and acceptance. Furthermore, the search and selection of disposal sites has proved to be politically and socially challenging. Recent successes show the benefit of open and transparent processes that allow sufficient time for meaningful involvement of stakeholders.

Chapter 4

RISK, PERCEIVED RISK AND PUBLIC ATTITUDES

Any perspective on the management of radioactive and hazardous wastes (the first theme of this study) or comparisons between wastes arising from different forms of electricity generation (the second theme of this study) cannot be complete without consideration of public attitudes and perceptions of risk. This matter is summarised in this chapter and considered in detail in Appendix 4.

For almost all activities in society risk, and how risk is perceived, are important considerations for decision making by governments as well as by industries and consumers. Societal acceptance of risk depends not only on scientific evaluations, but also on perceptions of risk and benefit.

Radioactive wastes are clearly a danger to human health and the environment if not properly managed. Public perception is that these wastes are also a danger when they are properly managed, or there is low public confidence that they will always remain properly managed. Today, the siting of radioactive waste disposal facilities does not depend only on resolving technical matters, but also requires public values and concerns to be addressed, because the public (at the local or national level, or sometimes both) may have a low acceptance of such facilities. However, there are many examples of hazardous wastes (including wastes with toxic and biohazard characteristics) being safely disposed over many decades. This demonstrates, at least in principle, that safe disposal of inherently dangerous substances can be achieved. In fact, a number of countries safely operate disposal facilities for radioactive waste (and, as with hazardous waste, have done over decades), though they are so far limited to LILW waste.

Nonetheless, there is ongoing debate all over the world regarding the disposal of hazardous and radioactive wastes. Inherent to achieving safe disposal is gaining public acceptance to support the construction of properly designed disposal facilities. Public acceptance of waste disposal facilities plays an increasing role in the decision-making procedure. The successful siting of hazardous waste disposal facilities and the inability to do so for high-level radioactive waste raises questions about differences in public perceptions of the risks of these facilities, and perception of the need for or value of the industries that produce each of these waste types. This factor depends heavily on whether the public believes that they or their environment will or may be harmed by the proposed new disposal facility. The public perceives and judges the degree and acceptability of risk differently from experts in the field. Effectively addressing public concerns about the potential risks of a waste disposal facility – whether those concerns appear to be well founded or not – has become a critical practical need in siting new waste treatment or disposal facilities.

4.1 Risk and perceived risk

Risk is assessed in an objective manner in scientific and engineering calculations, often resulting in a probabilistic evaluation of death to those exposed. However, this approach does not represent the degree of risk that affected individuals might feel. This is known as *perceived risk*. Perceived risk is subjective, and depends on both, information about the scientifically evaluated risk and a number of individual and societal risk perception factors, such as those shown in Table 4.1. The decision-making process for any proposed infrastructural project, whether it is a new road, airport, nuclear power plant or waste disposal facility, will (consciously or not) involve a judgement about risk (and benefit) by all the stakeholders involved. In general, for a range of reasons, stakeholder judgements are made based on perceived rather than scientifically evaluated risk. This in turn directly influences their acceptance level for the proposal. How stakeholders' perceptions of risk are acknowledged affects the level of trust they place in their elected representatives and in the project developers. An additional problem with nuclear facilities is that stakeholders do not necessarily have sufficient personal experience to form a judgement on whether safety criteria are acceptable, especially when they are presented as numerical risk.

As an everyday example, the risk perception factors shown in Table 4.1 indicate that an activity like driving a car is likely to have a lower perceived risk or is in any case an acceptable risk, because it is voluntary, under the driver's control, familiar, has clear benefits and the process is well understood. It is also a risk that is distributed somewhat evenly over most of the population; that is, many people do some driving to meet their transportation needs. The reverse is, in general, true for a proposal to site a radioactive waste disposal facility close to someone's home: the perceived risk is higher, or is less likely to be acceptable because the facility and degree of risk is not under the person's control, is not familiar, may not be seen as necessary and, importantly, the person sees that he is being involuntarily and disproportionately exposed to what he regards as a hazard. Hazardous waste and radioactive waste share many factors that tend to elevate the perception of risk and, indeed, both are viewed as high risks in comparison to most other activities in society. Of course, on a statistical basis, driving has a higher risk than does living close to a radioactive waste disposal facility. However, this is not what is perceived and does not correspond with the level of acceptance.

Table 4.1: Examples of risk perception and acceptance factors

Risk perception factor	Perceived risk of an activity will be greater, or acceptance of the risk lower when the activity is seen as:
Volition	Involuntary or imposed
Controllability	Under the control of others
Familiarity	Unfamiliar
Equity	Unevenly and inequitably distributed
Benefits	Having unclear or questionable benefits
Understanding	Poorly understood
Uncertainty	Relatively unknown or having high uncertainty
Dread	Evoking fear, terror, or anxiety
Reversibility	Having potentially irreversible adverse effects
Trust in institutions	Requiring credible institutional response
Personal stake	Placing people personally and directly at risk
Ethical/moral nature	Ethically objectionable or morally wrong

Other studies have compared the perceived risk from different societal activities by analysing responses from a range of different groups in the United States. This led to the concept of *dread risk*, synonymous with perceived lack of control, catastrophic potential, fatal consequences or the inequitable distribution of risks and benefits (see Appendix A4.3). Nuclear power and radioactive waste are regarded very unfavourably by the public in this context, perhaps because these complex technologies are unfamiliar and incomprehensible to most citizens and they do not see the benefit derived as being necessary (there are other sources of electricity generation). This study also noted that making a set of hazards more or less specific (for example partitioning nuclear power into uranium mining, power plants, radioactive waste disposal, etc.) had little effect on the risk perception of either the parts or the whole; the public tends to judge it all as one.

The public perception of risk is closely related to dread risk. The higher the dread risk, the more the public wants to see risks reduced and strict regulation imposed to achieve this reduction. In contrast, experts' numerical evaluation of risk is not related to dread; they see riskiness as synonymous with expected annual mortality. As a result, conflicts over risk result from experts and the public having different definitions of the concept. Appendix 4 provides a broad perspective on the difference between risk and the public's perception of risk by comparing the consequences of severe accidents in the energy sector with public attitudes and risk perceptions. Severe accidents (defined as having ≥ 5 fatalities) are the most controversial in terms of public perception and energy politics. Table A4.2 summarises the consequences of the severe accidents that occurred in the fossil, hydro and nuclear energy chains in the period 1969-2000. The largest number of immediate fatalities in the fossil energy chains was in coal and oil (for OECD and non-OECD countries combined, 20 276 and 20 218 respectively). The energy chain responsible for the largest number of immediate deaths was hydroelectricity (for OECD and non-OECD countries combined, 29 938), mainly because of the Banqiao/Shimantan dam failure in China in 1975.

The public's perception of risk in the energy-related industries, and particularly of the risks from nuclear power, does not appear to be impacted by the consequences of severe accidents that have actually occurred. Appendix 4 (Table A4.3) shows the consequence of accidents associated with different energy chains to allow comparison with the public's perception of risk, as judged by attitudes to different energy sources. These statistics show that nuclear power is actually one of the safest energy technologies, but this is certainly not the public perception. In considering the consequences of severe energy-related accidents, in terms of the numbers of immediate fatalities, injuries and evacuations, nuclear power only appears in the top ten accidents with the highest evacuations – for Three Mile Island and for Chernobyl.

As noted above, partitioning nuclear power risks (e.g. into uranium mining, power plant operation, radioactive waste disposal, etc.) has little effect on the risk perception of either the parts or the whole; the public tends to judge it all as one. Hence the public's perception of the risk associated with radioactive waste management affects their views on nuclear power risks overall: conversely, views on the safety of nuclear power can affect views on the safety and acceptability of related waste management solutions. The public tends to view nuclear power as risky, even though the consequences of severe energy related accidents demonstrate otherwise.

4.2 Public attitudes to radioactive waste management

Many public opinion polls have demonstrated the public's concern over management of radioactive waste. For example, in June 2007, a poll by the Ministry of Industry in France asked, "Which are the two most important disadvantages with nuclear power?" 37% of respondents said the production and disposal of radioactive waste. An annual opinion survey among young Slovenians found that around 36% of the respondents consistently saw the disposal of spent fuel as the most important disadvantage of nuclear power, more than those who cited the risk of a major accident. The issue of radioactive waste is of significant concern to Canadians: a large majority (82%) agree that new nuclear power plants should not be constructed until the problem of radioactive waste disposal is solved.

More evidence of the depth of concern on radioactive waste disposal comes from responses to further questions in a Eurobarometer poll carried out in 2005 where:

- 92% agree that a solution for highly radioactive waste should be developed now and not left for future generations;
- 81% believe that it is politically unpopular to take decisions about the handling of any dangerous waste;
- 79% think that the delay in making decisions in most countries means there is no safe way of disposing of highly radioactive waste.[1]

Further detail on opinion polls is provided in Appendix A4.

Data from the Eurobarometer survey show that the risks of nuclear power are judged to outweigh its advantages by 53% of respondents. Only 33% judged the reverse to be true. Respondents believe the biggest risks associated with nuclear power include disposal of radioactive waste, with only 39% believing that it can be done safely. The poll first asked, "Are you totally in favour, fairly in favour, fairly opposed or totally opposed to energy produced by nuclear power stations?" This showed 55% of people to be opposed to nuclear and 37% to be in favour.[2] Opponents of nuclear energy were then asked to what extent they would be in favour of nuclear energy **if** the problem of radioactive waste were resolved.

Responses to this question show that 38% of those opposed to nuclear energy would support it, if the issue of radioactive waste disposal were to be resolved. Just over a half (57%) of people opposed to nuclear would continue to be opposed if the issue of waste were resolved. Responses are shown in Figure 4.1 on a country by country basis.

These data clearly show the importance of the perceived risks of radioactive waste management and the impact of this perception on both the progress of implementing HLW/SF disposal facilities and on the acceptability of continuing or further expanding nuclear power generation.

The outcomes of these various opinion polls show that the future of nuclear power is dependent on managing radioactive waste, including its disposal, in a way that is acceptable to the public. Currently, the perceived risk from managing radioactive waste is high, but if the public sees that waste can be disposed safely (for example by a number of successfully implemented schemes internationally), it is possible (but clearly by no means certain) that perceived risk might eventually reduce as has been seen in the case of some hazardous waste management facilities. Resolution of the

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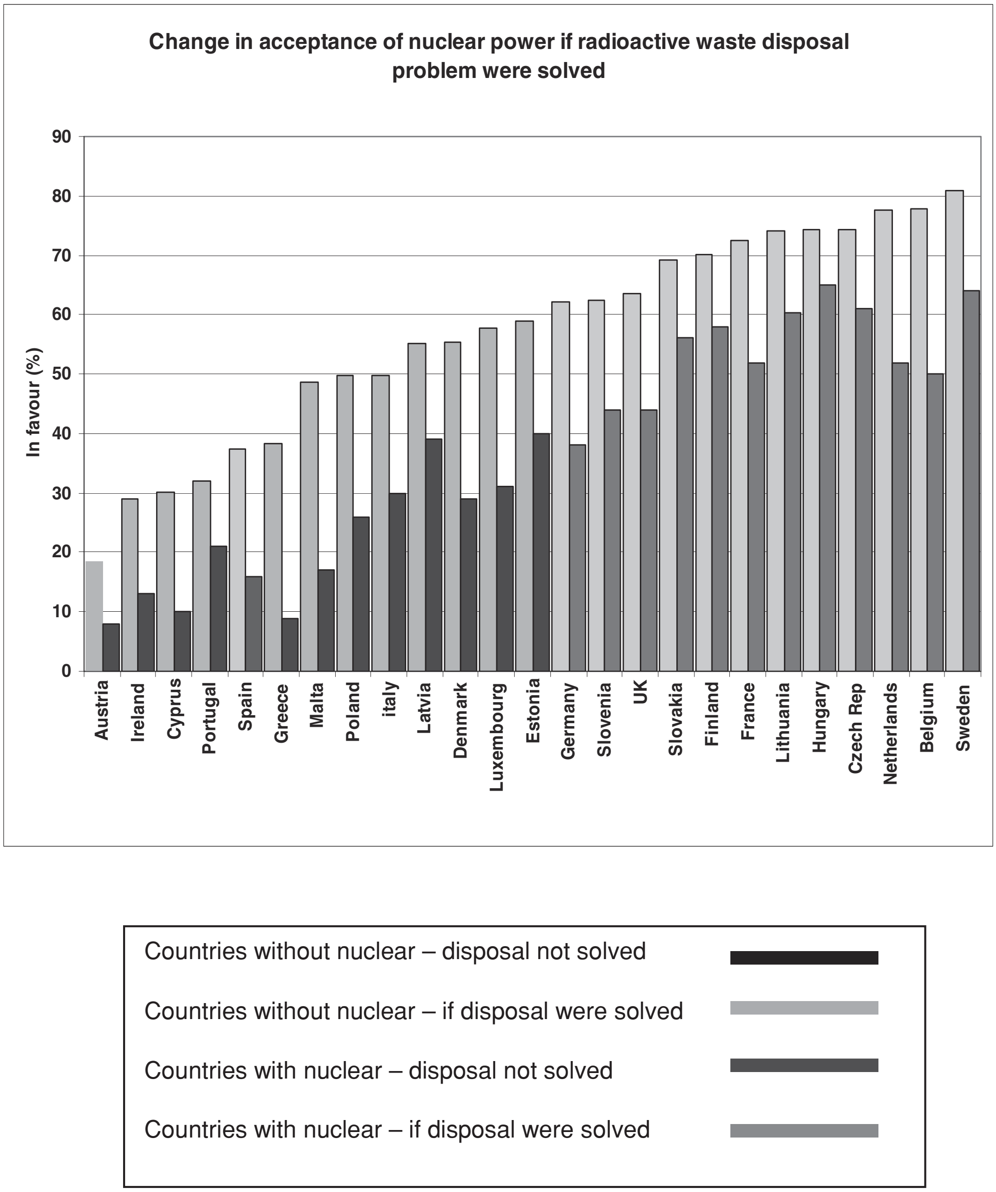
1. Since the text of this document was produced, a further Eurobarometer poll has been conducted, see http://ec.europa.eu/public_opinion/archives/ebs/ebs_297_en.pdf. This shows that, over the three years between which the data was collected, support for nuclear power has generally increased a few percentage points. However, the messages derived from the 2005 poll with respect to opinions on radioactive waste still remain valid.

2. A more recent poll (2008) even showed that support for nuclear power had grown from 37% to 44% and opposition reduced from 55% to 44%. Of those opposed, 39% would change their mind if the radwaste issue was resolved, 48% would not and 8% considered there was no safe solution to radwaste disposal.

waste issue in one country might be expected to have a positive impact on the public's perception of radioactive waste disposal elsewhere.

Figure 4.1: Europeans' change in acceptance of nuclear power if the radioactive waste disposal problem were to be solved



Chapter 5

CONCLUDING DISCUSSION AND LESSONS LEARNT

As explained in Chapter 1, the purpose of this NEA study is to offer policy makers a perspective on the management of radioactive waste. The study has two themes:

- comparison of radioactive and hazardous wastes and their management strategies;
- comparison of wastes that arise from electricity generation from coal and from nuclear power.

The purpose of this chapter is to present a concluding discussion from this study, together with some lessons learnt.

Sections 5.1 draws conclusions related to the first theme of the NEA study, by considering similarities and differences in the management of radioactive and hazardous wastes and their management strategies. Section 5.2 sets out conclusions from the study's second theme: comparison of wastes arising from coal and from nuclear power generation. Section 5.3 attempts to identify lessons learnt.

5.1 Theme 1 – Similarities and differences in the management of radioactive and hazardous waste

Similarities

In OECD countries competent authorities and stringent regulations are in place for both types of wastes and it is clear that both are generally well managed. There are many important similarities in the management of radioactive and hazardous waste.

Some of the similarities among OECD countries identified in this study are that both radioactive and hazardous wastes:

…at the international level

- Have agreed classification systems and definitions.
- Have a high degree of harmonisation and guidance on practices and management.

…at the national level

- Are subject to extensive legislation and standards.

- Have compliance monitoring carried out by dedicated administrative bodies or regulators.

- Require a proposed site to be fully characterised before development and that waste be characterised before treatment and/or disposal.

- Have a treatment and disposal facility licensing process that requires and checks safety assessments so that protection of the public and the environment can be reasonably guaranteed.

…regarding waste management and disposal

- Present risks over long time periods that cannot be totally avoided; some radionuclides have very long half-lives and some toxic materials in hazardous wastes last for an infinitely long time.

- Regard protection of human health and the environment and consideration for future generations as key components of their management principles.

- Use the same basic principles when developing national management policies.

- Have a primary aim of avoiding or reducing waste generation at source.

- Routinely dispose wastes at less hazardous levels in landfills or near-surface facilities that depend mainly on engineered barriers to reduce adverse impacts on human health and the environment.

- Have the siting procedure for treatment and disposal facilities performed in a stepwise manner that involves all concerned stakeholders in the decision-making process.

Differences

There are several important differences between the management of radioactive and hazardous waste. Some differences identified in this study are:

…characteristics

- Waste characteristics, and therefore management strategies, are fundamentally different between hazardous waste (which may have a range of hazardous characteristics making it flammable, oxidising, corrosive, reactive, explosive, toxic or ecotoxic) and radioactive waste (which, in the main, has only radioactivity and its potential to cause cancers as a hazard).

- Radioactivity decays over time, so the hazard associated with radioactive waste continuously and predictably reduces (although over a very significant time period for some isotopes); while many hazardous wastes can be treated to effectively reduce hazards to near zero, the intrinsic hazards in some hazardous waste (such as those containing heavy metals) remain for all time.

...quantities and sources

- The global generation rate of hazardous waste is of the order of 1 000 times that of current radioactive waste from nuclear electricity generation.

- Taking the United States as an example, there are in the order of 100 times more large hazardous waste generators than radioactive waste generators.

- Almost all industries, as well as households, generate some hazardous waste; most radioactive waste comes from a very few sources – primarily electricity generation.

…management processes (treatment)

- Whilst prevention, reuse and recycling are top priorities for hazardous waste, only a minority of countries reprocess spent nuclear fuel to recycle uranium and plutonium; although some countries recycle and reuse material previously contaminated with radioactivity others do not, primarily because of public concerns.

- In both cases the first objective is to avoid the creation of waste if at all possible. Once waste has been created, the *concentrate and contain* and the *delay and decay* options are used for radioactive waste, while the *eliminate or reduce the hazard* option (incineration, chemical treatment, etc.) is the primary strategy for hazardous waste. Containment is employed where the primary strategy is not practicable.

- For hazardous waste, a range of treatment options (such as incineration) is available, often to significantly reduce hazard before disposal; the intrinsic hazard from radioactivity cannot be removed or reduced by treatment before disposal, although interim storage can be used to allow the decay of short-lived radioactive components.

- Transboundary (transfrontier) shipments of hazardous waste occur regularly in the OECD and, on a smaller scale, worldwide to allow specialised treatment and disposal facilities to be used to manage specific waste streams; there is very little transboundary shipment of radioactive waste except, in a small number of cases, for spent fuel reprocessing.

…costs

- The unit costs of managing hazardous waste are considerably lower than for managing radioactive waste.

- Hazardous waste management is generally carried out on a commercial basis with immediate payment for services received; for radioactive waste, funds are generally built up from electricity generation revenues to pay for future disposal.

…factors influencing the progress in implementing disposal

- Many disposal facilities for hazardous waste, including a few geological repositories, have been successfully implemented and licensed worldwide; a few underground disposal facilities are also in operation for low- and intermediate- level radioactive waste (LILW) but no deep geological disposal facility is currently available for HLW/SF.

- The consensus in the scientific community is that disposal in stable geological formations is the best way to achieve the long-term management of long lived radioactive waste; geological disposal is not, in general, the primary management option for solid hazardous waste. However, in contrast to radioactive waste, deep geological disposal of some significant toxic waste streams has been successfully implemented and is being used in some countries.

- In most cases market forces drive early implementation of hazardous waste management facilities in a way that is not seen for radioactive waste.

- The siting and implementation of hazardous waste disposal facilities are generally being dealt with at the regional or local level whilst the disposal of radioactive waste (especially HLW/SF) is generally addressed at the national level, and there are even discussions at the international level.

- Typically, there is a high level of state involvement in radioactive waste management whilst a diversity of organisational frameworks deal with hazardous waste management, which is basically market oriented.

- The safety of radioactive waste disposal sites is generally quantitatively assessed against defined risk limits or targets, with assessments typically performed for time periods of up to one million years for HLW/SF and LILW-LL disposal sites; underground hazardous waste disposal facilities in rock salt are generally assessed for shorter periods of 10 000 to 50 000 years as in the case of Germany facilities.

- The implementation time for hazardous waste management facilities is generally considerably shorter than for implementation of radioactive waste facilities where the demonstration of the safety and feasibility of deep geological disposal for HLW/SF has involved costly R&D efforts (sometimes involving the construction of underground research facilities) that have typically stretched over two or three decades; corresponding R&D efforts for management of the most hazardous non-radioactive waste have been less costly and time consuming.

- In some countries, the concept of retrievability has been introduced for deep geological disposal of radioactive waste to provide a capability to manage unforeseen occurrences; for hazardous waste, retrievability is mainly aimed at recovering valuable resources from surface and near surface disposal facilities.

- Although gaining socio-political acceptance for hazardous waste disposal is difficult, it appears to be less complicated than achieving acceptance for geological disposal of radioactive waste.

5.2 Theme 2 – Similarities and differences in the management of wastes that arise from electricity generation from coal and nuclear power

In 2005, about 40% of the world's electricity came from coal and 15% from nuclear generation. The wide availability of coal means that it will continue to be used and projections suggest that its use will increase significantly as world energy demand continues to grow; globally, coal and nuclear are expected to be two of the primary sources of base load electricity in the future.

The main similarities and differences identified in this study between management of waste from coal and nuclear power generation are set out below. Unlike the previous section, where a considerable number of similarities were noted between radioactive and hazardous waste, there are few similarities between management of waste from coal and nuclear power generation.

…waste quantities

- Globally, coal generation produces waste at a rate per unit energy that is about 300 times higher than does nuclear.

- Waste from coal generation:

– Ash	0.6 Gt/a	(90 kt/TWh)
– CO_2	10.5 Gt/a	(1 600 kt/TWh)
– Mining	20.0 Gt/a	(3 000 kt/TWh)

- Waste from nuclear generation

– All solid radioactive waste (excluding mining & milling)	< 0.005Gt/a	(0.2 kt/TWh)
– HLW/SF	0.000010 Gt/a	(0.004 kt/TWh)
– Mining	< 0.05 Gt/a	(< 15 kt/TWh)

…waste properties and disposal

- In most countries coal generation wastes are not classified as hazardous whilst wastes from nuclear power generation are.

- Unlike nuclear power, most of the "wastes" stemming from coal-fired power generation are released directly into the environment. In particular, there is global concern about the climate change effects of CO_2 emissions from fossil fired electricity generation, and air pollution from coal-fired electricity production including a mixture of pollutants damaging to health and the environment.

- In the vast majority of countries, all solid waste from coal-fired generation can be disposed to landfill. In general, about half of nuclear power solid wastes can be considered for disposal at relatively simple landfill sites. About 2% of nuclear power waste is HLW/SF for which no disposal facilities are currently available.

…recycling waste to extract economic value

- Large fractions of the solid residues from coal-fired electricity generation are recycled. Some spent nuclear fuel is recycled to extract uranium and plutonium for future fuel manufacture. Because so much coal ash is reused, the distinction between a waste and a product is not as clear-cut as it is for radioactive waste.

…impact on climate change

- The Intergovernmental Panel on Climate Change considers that both carbon capture and storage and nuclear power have the capability to reduce annual greenhouse gas emissions, CCS applied to coal generation by 0.49 Gt CO_2 eq by 2030 and nuclear energy by a further 1.9 Gt CO_2 eq beyond the 1.7 Gt CO_2 eq already anticipated by the International Energy Agency. The IPCC analysis shows that both CCS and nuclear power will be needed in significant quantities to meet the necessary climate change targets.

- Because of energy requirements to operate the CCS equipment, a power plant with CCS should reduce CO_2 emissions to the atmosphere by approximately 80-90% compared to a plant without CCS.

…economic issues

- Like nuclear power, coal-fired generation equipped with CCS requires a significant economic investment.

- Estimates show that CCS would increase the cost of electricity by between 22 and 60%; Gen III/III+ nuclear reactors are broadly competitive with coal-fired generation that includes a modest carbon constraint that does not fully account for the use of CCS.

…development status

- The commercial viability of both nuclear power and coal-fired power without CCS has been demonstrated. While geological disposal of radioactive wastes from nuclear power production has been internationally endorsed as technically and economically feasible, the verdict on CCS – which has never even been demonstrated on an industrial scale – is still out.

- Only one operational project, a 30 MWe coal-fired plant, is currently attempting to demonstrate both carbon capture and storage. Large numbers of commercial nuclear power stations are in operation and others, including modern Gen III/III+ plants, are under construction. OECD countries such as Sweden and Finland are also in the process of building the geological repositories for high-level radioactive waste disposal.

…safety

- CO_2 is not considered to be a hazardous waste but both coal with CCS and nuclear power rely on deep geological disposal as their waste management solution. However, coal with CCS would produce about 40 000 times more waste per unit of electricity that required geological disposal than would nuclear power.

- Waste from CCS would be disposed over a very much larger geological volume as a supercritical fluid contained only by natural barriers whilst waste from nuclear power would be disposed as a solidified and encapsulated product contained by both engineered and natural barriers.

…regulation

- Regulation is still evolving for CCS and, to a much lesser extent, for HLW/SF disposal. The latter already has a well established international framework and guiding principles and many OECD countries have established safety standards.

- A key issue for investors will be the extent of their liability for long-term monitoring and potential remediation.

…stakeholder issues

- The largest current CO_2 storage projects do not yet have public acceptability as part of their remit whereas this is of prime importance to both the nuclear and hazardous waste industries.

- The largest of the international non-governmental environmental organisations are broadly opposed to both CCS and nuclear power.

5.3 Lessons learnt

Many of the differences between management of hazardous and radioactive waste have their origins in the significant variations between the nature and properties of the wastes. The ability to transfer experience from the hazardous waste world to the radioactive waste world is therefore somewhat limited.

The fact that there are numerous hazardous waste disposal facilities worldwide suggests that there are effective economic and other driving forces in place for implementation of strategies for hazardous waste management.

Examples of such driving forces are:

- The huge amount of hazardous waste generated by our society means that timely decision making on the implementation of hazardous waste facilities was essential if countries' industrial capabilities were not to come to a halt. There were therefore clear national economic, and hence political, imperatives to implement hazardous waste management processes, albeit under strict regulation. Because volumes of radioactive waste are relatively small, and the nuclear industry has historically managed them safely using surface storage, the same imperatives have not applied: this may have impacted the much slower development of radioactive waste disposal facilities. The availability of other methods of power generation may also reduce the perception that nuclear power generating capacity is essential, thereby reducing pressure on solving the waste disposal issue. The growing concern with respect to climate change already seems to be having some impact in changing this view.

- Because of the widespread generation of hazardous wastes, by small companies as well as large ones, and because strict regulation exists for their management, there are market opportunities for the development of hazardous waste treatment and disposal. The same is not true for radioactive wastes, where the generators usually treat the waste in house and, in many cases, store it on their own sites for eventual disposal without further treatment.

- Some organic hazardous wastes can become significant fire or explosion hazards if not treated promptly. It is therefore in the generators' commercial interests to have these wastes treated and disposed. In some cases, it is possible to recycle the hazardous waste, or to recover the energy it contains. None of these considerations applies to radioactive waste where there is generally no commercial incentive (at least in today's economy) to retrieve and recycle stored waste and which is generally not a fire hazard.

A similar situation regarding economic driving forces appears to have arisen for CCS (see Appendix 3), although this technology is clearly still in its infancy. A methodology is available to assess the effect of CCS on greenhouse gas emissions, enabling countries to report emissions reductions due to CCS, and providing the basis for its inclusion in emissions trading schemes. The EU Greenhouse Gas Emission Trading Scheme started to allow trading in CCS emission reductions in 2008. It seems that an essential precondition for development of CCS is the ability to profit from reduced CO_2 emissions.

One important factor, which appears to make timely decision making less difficult for hazardous, compared with radioactive, waste disposal is that the public perceives a lower level of risk for hazardous waste management. This study has identified this factor but has not evaluated the reasons behind it. One significant reason may be the difference in familiarity between radioactive and non-radioactive waste types. Many common household items such as constituents of refrigerators, fluorescent tubes and batteries are generally classified as hazardous wastes when they are disposed, and potentially toxic chemicals like wood preservatives and pesticides are in common household use. Thus, the public is broadly familiar with many types of hazardous wastes. Such familiarity does not generally exist for the small volumes of radioactive waste that are managed on relatively few sites.

Another factor may be that the public recognises that management of large volumes of hazardous waste is a by-product of the economic activities that are necessary to maintain a modern industrial society. In general, the public wants to maintain the lifestyle that an industrial society provides and is therefore inclined to accept the risks associated with hazardous waste. There are alternatives to nuclear electricity generation, so the public is less willing to accept the risks associated with radioactive waste.

For many people nuclear power represents complex technology that seems to them inherently hazardous and is difficult to understand. A 2005 Eurobarometer poll showed that disposal of radioactive waste was seen by many Europeans as a significant reason to oppose nuclear power. A majority of citizens in 16 of the (then) 25 EU countries said they would support nuclear power if the waste problem were solved, whilst a majority in only 8 countries would support nuclear with the waste issue unresolved. In addition, 92% of Europeans agree that a solution for highly radioactive waste should be developed now and not left for future generations and 79% think that the delay in making decisions in most countries means there is no safe way of disposing of highly radioactive waste.

These data clearly show the importance of the perceived risks of radioactive waste management and the impact of this perception on both the progress of implementing HLW/SF disposal facilities and on the acceptability of continuing or further expanding nuclear power generation. Support for nuclear energy will therefore be expected to increase when radioactive waste disposal facilities become available for HLW/SF.

Appendix 1

STRATEGIC ISSUES FOR RADIOACTIVE WASTE

The purpose of this appendix is to provide an overview of the quantities, principles, practices and experience in radioactive waste management. It is primarily aimed at decision makers who have some familiarity with the topic.

It starts by describing the main types and amounts of radioactive waste. Section A1.2 summarises the principles involved in managing the waste, including a description of their historical evolution while Section A1.3 looks at the characteristics of the hazards and risks to human health and the environment posed by these materials. Section A1.4 looks at solutions to radioactive waste disposal that have been planned or adopted; there is a two-fold approach: technologies for disposal and means of financing its implementation. To be certain that disposal options are safe and feasible, an adequate legal and institutional framework is needed. Section A1.5 describes some generally agreed institutional schemes, setting out the role and responsibilities of the main actors.

Safety is paramount to radioactive waste management and demands specific consideration. The philosophy and methodology underlying the assessment of the safety of disposal facilities is addressed in Section A1.6. Section A1.7 deals with the various stages and considerations in step-wise development and implementation of disposal solutions whilst Section A1.8 is devoted to currently perceived challenges in the future development of disposal facilities.

Cultural, societal and geographical similarities and differences have resulted in a variety of paths towards implementing national disposal solutions, but a common safety and security objective underlies all these paths. In addition, there is a common international framework that guides national regulatory oversight and implementation of disposal. This appendix refers primarily to this international framework established through active international fora (e.g. NEA, IAEA). References to specific countries and their facilities are provided to illustrate some important aspects of radioactive waste management.

Appendix 1 does not address the issues of public perception of radioactive waste management or the role of public participation and stakeholder involvement in decision making. These topics are pivotal to waste management and have been extensively studied by the NEA; they are discussed in Chapter 4 and Appendix 4 of this report.

A1.1 Radioactive waste definition, classification and quantities

Definition

Radioactive waste is defined by IAEA as "any material that contains or is contaminated by radionuclides at concentrations or radioactivity levels greater than the exempted quantities established by the competent authorities and for which no use is foreseen". Most civil radioactive waste arises from nuclear power production but a wide variety of industries, including medicine, agriculture, research, industry and education, use radioisotopes and produce radioactive waste.

Classification

Several classifications are possible when describing radioactive waste. These include physical state (since radioactive waste can be solid, liquid or gaseous) as well as isotopic content and concentration. The types of radiation (alpha, beta and gamma) emitted by the prevailing radioisotopes in the waste is another basis for classification that defines the necessary degree of shielding. Another form of classification relates to the half-life[1] of the predominant radionuclides of a given waste.

The system adopted by IAEA, which is the most internationally accepted, combines the type of radiation emitted, the activity of the waste and its half-life to present an easy method of classification based on the main following categories: (IAEA, 1994)[2]

- Exempt waste (EW): excluded from regulatory controls because radiological hazards are negligible.
- Low- and intermediate-level waste (LILW): radioactivity levels are above those for exempt waste and thermal power below about 2 kW/m^3; IAEA recognises two sub-categories of LILW:
  1. short-lived waste (LILW-SL): primarily contains short-lived radionuclides, with long-lived radionuclide (including long-lived alpha emitter) concentrations restricted to an overall average of 400 Bq/g per waste package;[3]
  2. long-lived waste (LILW-LL): contains long-lived radionuclide concentrations that exceed limits for short-lived waste.
- High-level waste (HLW): contains sufficient concentration of radionuclides to produce heat generation greater than 2 kW/m^3; the typical activity levels are in the range of $5x10^4$ to $5x\ 10^5$ TBq/m^3.

There are three exceptions to some radioactive waste classification schemes that correspond to the following materials:

- mining and milling wastes: residues left from mining and extraction of uranium and other raw materials that contain naturally occurring radionuclides;
- environmental contamination: radioactively contaminated environmental media, such as soil and groundwater;
- spent nuclear fuel (fuel that is removed from a reactor when its irradiation and energy output has reached its designed level) is considered as either a resource (as it still contains unused uranium and usable plutonium) or a waste depending on which management strategy a country is using.[4]

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1. Each radioactive element has its characteristic half-life ($t_{1/2}$), which is the time taken for half of its atoms to decay. In the classification scheme of IAEA two kinds of radioactive waste are distinguished: short-lived waste, whose predominant activity is defined by radionuclides with $t_{1/2} < 30$ years and long-lived one, where $t_{1/2} > 30$ years.
2. In late November 2008, after the text of this document had been prepared, the IAEA published a new Draft Safety Guide (DS390), in which it proposes 6 classes of radioactive waste.
3. Although not yet considered by IAEA, very-low-level waste (VLLW), is a new category of waste inside LILW-SL that is currently being applied in several countries (France, Spain and Sweden) for those short-lived wastes with very low specific activity of alpha emitters, generally less than 10 Bq/g.
4. Two different management strategies are used for spent nuclear fuel. In the closed-cycle strategy, the fuel is reprocessed to extract usable material (uranium and plutonium) for the fabrication of new fuel. In the

Even if these materials are not always part of a classification scheme for radioactive waste, they are still normally subject to regulatory control and requirements for management and, if applicable, for disposal.

Waste quantities

Cumulative generation

IAEA has developed and launched the Net Enabled Waste Management Database (NEWMDB).[5] (IAEA, 2007b) IAEA has produced an estimate of the cumulative worldwide inventory of radioactive waste in 2005 using NEWMDB and publicly available data sources for countries that were not reporting into NEWMDB in 2005.

These data on radioactive waste amounts and classes cover the 43 main waste-producing countries (listed in Table A1.1) and are considered appropriate, for the purposes of this study, to show the order of magnitude of cumulative worldwide radioactive waste generation.

Table A1.1: Countries contributing data to NEWMDB for 2005

Argentina	France** (data for 2004)	Norway
Belgium, Kingdom of	Germany	Philippines, Republic of the
Brazil, Federative Republic of	Hungary, Republic of	Romania
Bulgaria, Republic of	Indonesia, Republic of	Slovakia
Canada	Iran, Islamic Republic of	Slovenia, Republic of
Chile, Republic of	Ireland	Spain, Kingdom of
China** (preliminary data 2006)	Italy	Sweden, Kingdom of
Croatia, Republic of	Japan	Switzerland
Cuba, Republic of	Kuwait, State of	Thailand, Kingdom of
Czech Republic	Lithuania, Republic of	Turkey, Republic of
Ecuador, Republic of	Malaysia	Ukraine
Estonia, Republic of	Mexico	United States of America
Finland	Netherlands, Kingdom of the	United Kingdom** (data for 2006)
Australia*	Russian Federation*	Republic of Korea*
South Africa*		

Sources:

* for those countries: Commonwealth of Australia, 2005; Denmark National Board of Health, 2005; Korean Ministry of Science and Technology, 2006; Russian Federation, 2006.

** Reporting date is different from 2005.

The total global radioactive waste inventory that has been generated up to 2005 is presented in Table A1.2, which does not include wastes from uranium milling. This table presents the inventory

open-cycle strategy, spent fuel is considered a waste and is stored pending disposal. As of 2009, China, France, India, Japan, Netherlands, the Russian Federation and the United Kingdom reprocess most of their spent fuel, while Belgium, Canada, Finland, Germany, Sweden and the United States have currently opted for direct disposal (but, as of 2009, the US will be evaluating alternative approaches for its waste management programme). Some other countries have not yet decided which strategy to adopt. They are currently storing spent fuel and keeping abreast of developments associated with both alternatives.

5. The NEWMDB contains information on national radioactive waste management programmes, radioactive waste inventories, radioactive waste disposal, relevant laws and regulations, waste management policies, and plans and activities. The first NEWMDB data collection cycle was conducted in March 2002 (for year 2000 data). Subsequent collections have been performed annually from 2003 onwards.

divided into waste class and origin and shows the cumulative quantities that are in storage and that have been disposed. This table is based on data contained in the IAEA NEWMDB database.

Table A1.2: Global cumulative radioactive waste inventories for all countries, as of 2005[6]

Waste class and origin	Waste in storage (m^3 x 1 000)	Waste that has been disposed (m^3 x 1 000)
LILW_SL	2 288	19 704
Decommissioning/remediation	1 349	14 820
Defence	90	2 545
Fuel fabrication/enrichment	127	327
Not determined/unknown	55	32
Nuclear applications	171	427
Reactor operation	357	1 290
Reprocessing	138	262
LILW_LL	3 103	98
Decommissioning/remediation	2 326	35
Defence	76	48
Fuel fabrication/enrichment	21	0.09
Not determined/unknown	28	1.4
Nuclear applications	56	2.8
Reactor operation	550	11
Reprocessing	44	–
HLW	366	0.01
Decommissioning/remediation	6	–
Defence	356	–
Fuel fabrication/enrichment	0.02	–
Not determined/unknown	0.01	–
Nuclear applications	0.3	–
Reactor operation	0.7	0.01
Reprocessing	3	–
Total	**5 757**	**19 802**

Source: IAEA, 2007b.

Table A1.2 shows that about 26 million m^3 of radioactive waste (excluding milling wastes) had been generated worldwide up to 2005. Of this cumulative total, 20 million m^3 had been disposed and 6 million m^3 had been placed in storage. Note that these figures include wastes from military sources and other non-power production activities. This report is not intended to deal with military and other applications, but the data is included here for completeness. Note also that the category Decommissioning/Remediation in NEWMDB does not distinguish between military and civilian wastes. A closer look at the NEWMDB shows that most of this waste is reported by the US, where there have been very large clean up programmes on the military sites, which probably accounts for most of this waste. Also notable is that the HLW from military applications totally dominates the quantity of HLW in storage.

6. Table A1.2 includes in the global cumulative inventories radioactive waste that originates from defence sources. The scope of this NEA study does not include defence related waste; however, volumes are included here for completeness and comparison.

Annual generation rates from nuclear power production

Table A1.2 shows the cumulative generation of LILW over many decades. The IAEA provides data on the quantity of LILW that is generated annually from nuclear power plants, in this case in 2000. (IAEA, 2007a) These data are shown in Table A1.3.

Table A1.3: Global LILW generation from nuclear power plants in 2000

Reactor type	Number of reactors	LILW generated (m^3/a)
ABWR	2	1 300
AGR	14	5 450
BWR	89	38 400
FBR	3	520
GCR	20	17 000
RBMK	18	20 270
PHWR	31	3 180
PWR	206	49 100
WWER	49	18 560
TOTAL	**432**	**153 780**

Table A1.3 shows that about 0.15 million m3 of LILW is generated each year from nuclear power plants worldwide.

The NEWMDB allows an alternative method of calculating these values to also include wastes from the civil fuel cycle facilities servicing the reactors. NEWMDB data for 2005 shows 22 x 10^6 m^3 of accumulated ILW-SL, of which about 10% is from power generation, i.e. about 2.2 x 10^6 m^3. The figures for ILW-LL are 3.2 x 10^6 m^3 of which some 20% is attributed to power generation, i.e. about 0.64 x 10^6 m^3. IAEA (2007a) quotes Nucleonics Week data for total nuclear generation up to March 2005 of 5 402GWe-years. Hence average LILW-SL production per year is 407 m^3/GWe and ILW-LL production per year is 118m^3/GWe. The total annual LILW generation is therefore around 530m^3/GWe.

In 2005, Power Reactor Information System (PRIS) (IAEA, 2008) shows the energy availability factor was 83% and NEA 2008a shows the installed nuclear capacity was about 360GWe. On this basis the energy produced was about 300GWe-years and LILW-SL annual waste production some 120 x 103 m^3/GWe-year. Similarly, annual LILW-LL is approximately 36x103m^3/GWe-year, and total LILW annual production 160 x 103 m^3/GWe-year.

Note that these are quite conservative values in terms of today's waste generation rates, given that waste quantities produced have been significantly reduced over the last few decades of operation, as noted in IAEA and in many other references. (IAEA, 2007a) These values may be compared with the 400 million m^3 of hazardous waste generated yearly (see Appendix 2).

In principle, account must also be taken of wastes from the extraction of uranium from ores (milling wastes) which present a low level of radioactive content but at large volumes and which are managed separately, normally being disposed close to the site of the uranium mine. (NEA, 2002a) From the data in NEA, 2008b, typical uranium ores have grades of 0.14% with exception of Canada, where there are some very rich ores. Using the data on quantities of and contributions to the global uranium supply in NEA, 2008b shows that these wastes dominate the volume of annual production at around

14 million tonnes per year. Other mineral extraction industries also produce considerable quantities of extraction wastes and, like these other industries, uranium mining also produces mining wastes.

At the end of life, nuclear power plants and the fuel cycle facilities that serve them must also be decommissioned, generating more radioactive (and non-radioactive) wastes. NEA gives values of the quantities of radioactive wastes from decommissioning different sorts of reactors per GWe capacity (NEA, 2003a) and IAEA allows the numbers of different reactors in the world fleet and their powers to be identified. (IAEA, 2008) IAEA indicates that a reprocessing plant will generate a similar amount of waste to a power plant, but with a higher portion of LILW-LL. (IAEA, 2007a) Given that many reactors are serviced by an individual fuel cycle facility, the decommission volumes from power plants alone provide a reasonable indication of decommission volumes.

These various calculations lead to Table A1.5, the total quantity of wastes being produced or committed to being produced per year by nuclear power plants and the facilities needed to service them.

Table A1.5: Approximate quantities of radioactive wastes produced per year (base date 2005)

LILW[i]-SL	125 000 m^3/a or 300 000 t/a
LILW[i]-LL	35 000 m^3/a or 85 000 t/a
Committed[ii] decommissioning waste	25 000 m^3/a or 60 000 t/a
Spent nuclear fuel	10 000 tHM/a
Committed vitrified HLW[iii]	1 500 m^3/a
Milling[iv, v] waste	15 million m^3/a
Totals	~195 000 m^3/a (or 455 000 t/a) plus $15x10^6 m^3$ of low-level milling wastes

i) These values are likely to be an overestimate as they average the quantity of waste generated over the history of nuclear power plants over the total power produced. As indicated in IAEA, 2007a and elsewhere, better management practices have greatly reduced the quantities of waste produced as time has progressed. Approximate conversion factor of 2.4 t/m^3.

ii) Committed decommissioning waste: the quantity of decommissioning waste that will be generated at the end of life of the world fleet is accounted for by allocating equal quantities over each of the assumed 40y lives of the power plants. Value indicates quantities for all power and fuel cycle plant wastes, not just reactors. These wastes will include significant quantities of VLLW and LILW-SL, smaller quantities of LILW-LL and very small quantities of HLW.

iii) Committed HLW: the quantity of HLW that would be generated if all of the fuel generated in a year in those countries with a policy of reprocessing is eventually reprocessed; conversion factor of 400 l/tHM from IAEA, 2007a. Note that this waste quantity has already been included as part of spent nuclear fuel.

iv) Milling wastes are generally of low radioactivity and, as mentioned earlier, are not always included in radioactive waste classification systems. In 2005 only some 60% of uranium consumed was produced from freshly mined uranium ore, a percentage that this is fairly representative of current practice (NEA, 2008a) the rest coming from secondary sources (recycled weapons material, stock rundown etc). If secondary sources were not available, milling waste would rise to 25 million t/a, assuming the production mix remained constant.

v) Around 25% of produced uranium is from *in situ* leaching (ISL) (NEA, 2008a) which produces no milling or mining wastes. Of that uranium which is mined, some comes from open pit mining and some from underground mining. Essentially non-radioactive mining wastes are therefore also produced, open pit mining generally producing larger quantities. IAEA suggests that mining wastes can be estimated as equivalent to milling wastes, but points out that real data is scarce and values are highly variable from mine to mine and very uncertain. (IAEA, 2007a)

Table A1.6: Approximate quantities of radioactive waste produced per GWe-year (2005 base data)

LILW-SL[i]	410 m^3/a or 980 t/a
LILW-LL[i]	120 m^3/a or 290 t/a
Committed[ii] decommissioning waste	210 t/a or 90 m^3/a
Spent fuel	30 tHM/a
Committed[iii] vitrified HLW	12 m^3/a
Milling[iv, v] waste	45 000 m^3/a
Totals	~630 m^3/a (or 1 500 t/a) plus 45 000 m^3/a of low level milling wastes

i) As in Table A1.5, these values are likely to be an overestimate.

ii) As in Table A1.5, committed decommissioning waste is the quantity of radioactive waste that would be generated at the end of life of the whole plant and its supporting fuel cycle facilities, accounted for by allocating it evenly over an assumed 40y life.

iii) This is the quantity of HLW that would be generated if the **whole of the spent fuel** were eventually to be reprocessed. Note that this waste has already been included as part of spent nuclear fuel.

iv) As in Table A1.5, if secondary sources were not available, this value would rise to 80 000 t/a.

v) As in Table A1.5, essentially non-radioactive mining wastes of a similar quantity as milling wastes would also be produced.

A1.2 Ethics and principles for final disposal

Radioactive wastes are a potential risk to health and the environment due to their radiological and chemical properties. Although there are different categories and types of radioactive waste and accordingly different kinds of risks, there is a common basic principle for their management: radioactive waste shall be managed in a manner that protects human health and the environment, now and in the future without imposing undue burdens on future generations. (IAEA, 2006) Due to the long timescales involved, the implementation of this principle is especially relevant when considering HLW. The description of the ethics and principles in this section is focused on the final disposal of high-level radioactive waste; the principles for managing low-level waste can be stated in a very similar way.

Geological repositories as a method to isolate and dispose of high-level radioactive wastes (HLW) were proposed in several research papers as early as the 1950s. In the United States, the high-level waste that originated from defence-related activities had been stored in tanks. A discussion on how to manage and stabilise high-level waste was initiated in 1955 by the National Academy of Sciences/National Research Council (NAS/NRC) under a contract with the Atomic Energy Commission (AEC). Based on results of this discussion and others, the NAS/NRC compiled and published a report entitled *The Disposal of Radioactive Waste on Land* in 1957. (NAS/NRC, 1957) The report mentioned that safe disposal meant, "the waste shall not come in contact with any living thing". Accordingly, the principle was to be understood in the sense that safe disposal is the isolation of radioactive waste from the living environment. The report envisaged that the most promising method to dispose of high-level radioactive waste in the future would be the emplacement of the waste in a rock salt formation. Further, the next most promising alternative seemed to be the stabilisation of waste in a slag or ceramic material forming a relatively insoluble product.

In the 1960s, research and development activities (R&D) for the management of HLW made a sound start in several countries. For example, *in situ* tests commenced at the Asse salt mine in Germany. In the 1970s, R&D of geological disposal made great progress by means of intensified multilateral collaborative actions or international R&D. The OECD Nuclear Energy Agency was inaugurated in 1975; the international joint R&D sponsored by NEA at the Stripa iron ore mine in Sweden (1977-1992), represented a typical example of collaborative projects in that era. In the 1960s and 1970s, R&D and data collection provided the necessary data to show the feasibility of disposal and to allow the safety assessments required for the design and operation of geological repositories.

In 1977, soon after the establishment of the NEA, a report was published on the objectives, concepts and strategies for the management of radioactive waste. (NEA, 1977) The document set out basic aspects of radioactive waste disposal that lead to commonly accepted principles. Some of these are:

> "For long-lived wastes the objective of radioactive waste management is to ensure the required degree of isolation from man over a time scale which precludes completely any form of reliance on long-term surveillance.
>
> "Taking into account the relative uncertainties about the ultimate cost of disposal (at least for some categories of waste), the possible delays between waste production and the implementation of disposals schemes, and the need to foresee satisfactory financing of future waste management operations resulting from current activities, it appears desirable to make specific financial provisions. Such provisions might take the form of funds; contributions could be levied according to the 'polluter pays' principle, for example, on the basis of nuclear electricity production."

In the 1980s, comprehensive studies were launched to evaluate the feasibility of geological disposal and to clarify future issues on its implementation. A report published by NEA in 1982, entitled *Disposal of Radioactive Waste, An Overview of Principles Involved* discussed those aspects that had not been well clarified up to then. (NEA, 1982) The report concentrated on a review of the social and ethical aspects underlying the technical approach adopted for the disposal of radioactive waste. In this document, the goal of waste disposal was stated as follows:

> "The objective of waste disposal is to ensure that wastes are dealt with in a manner which protects human health and the environment, and minimizes any burdens placed on future generations while, at the same time, taking into account social and economic factors."

Thus, protection of human health and the environment and consideration for future generations were selected as the key components of principles for the management of radioactive waste.

From the late 1980s onward, and following the progress of R&D by individual countries or within an international framework, progress towards the implementation of disposal operations was made in several countries: Germany, Sweden, United States, and others. Meanwhile, the International Atomic Energy Agency (IAEA) began to develop safety principles, regulatory policies and standards required for the implementation of geological disposal. The IAEA Safety Series No. 99 (1989) provides internationally agreed principles and standards for a deep geological repository for HLW. (IAEA, 1989)

A significant milestone was achieved in 1995 when IAEA produced *The principles of radioactive waste management*, Safety Series No. 111-F, which defines a set of internationally agreed fundamental principles. (IAEA, 1995) In this document, IAEA formulated nine principles for the safe management of radioactive waste:

- Principle 1: Protection of human health

- Principle 2: Protection of the environment
- Principle 3: Protection beyond national borders
- Principle 4: Protection of future generations
- Principle 5: Burdens on future generations
- Principle 6: National legal framework
- Principle 7: Control of radioactive waste generation
- Principle 8: Radioactive waste generation and management interdependencies
- Principle 9: Safety of facilities

Despite the definition of a clear international framework, there was no clear progress towards implementation of geological disposal and several countries failed to achieve milestones such as siting decisions. In response to these developments, NEA published a 1995 collective opinion entitled *The Environmental and Ethical Basis of Geological Disposal*. (NEA, 1995) In the document, the Radioactive Waste Management Committee (RWMC) particularly addressed fairness and equity considerations between and within generations:

> "between generations (intergenerational equity), concerning the responsibilities of current generations who might be leaving potential risks and burdens to future generations; and
>
> "within contemporary generations (intra-generational equity), concerning the balance of resource allocation and the involvement of various sections of contemporary society in a fair and open decision-making process related to the waste management solutions to be implemented."

RWMC set out the ethical considerations for radioactive waste management strategy as follows:

> "the liabilities for waste management should be considered when undertaking new projects;
>
> "those who generate the wastes should take responsibility, and provide the resources, for the management of these materials in a way which will not impose undue burdens on future generations;
>
> "wastes should be managed in a way that secures an acceptable level of protection for human health and the environment, and affords to future generations at least the level of safety which is acceptable today; there seems to be no ethical basis for discounting future health and environmental damage risks;
>
> "a waste management strategy should not be based on a presumption of a stable societal structure for the indefinite future, nor of technological advance; rather it should aim at bequeathing a passively safe situation which places no reliance on active institutional controls."

A diplomatic conference supported by the IAEA also produced a set of basic ethics and principles in The Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management that was signed in 1997 and came into force in 2001. (IAEA, 1997c) The principles of the Joint Convention are similar to those produced by the NEA and provide guidance for the large number of countries integrated into the IAEA.

A1.3 Hazards and risks from radioactive waste management

All radioactive wastes present a potential hazard to human beings and the environment if not properly managed. However, a radioactive substance will result in an actual radiation dose to persons only if there is a chain of events (a scenario) that allows the radioactive isotopes in the waste to be transported to man. The risk associated with each scenario depends not only on the potential hazard but also on the likelihood of events occurring that may result in exposure to radiation. (Chapman, N.A. and C. McCombie, 2003)

Nature of hazard

The hazard from radioactive wastes is primarily due to the energy and type of radiation emitted by the radioisotopes in them. Chemical toxicity of these elements is also a source of hazard but usually to a much lesser extent than that associated with radiological characteristics. Radiation may produce effects on living cells resulting in three outcomes: a) injured or damaged cells repair themselves, resulting in no residual damage; b) cells die being replaced through normal biological processes; c) cells incorrectly repair themselves resulting in a biophysical change. In this third case, there is the possibility of inducing cancers or altering the genetic code (DNA) of irradiated cells. It is generally assumed that high radiation doses tend to kill cells, while low doses tend to damage or alter the DNA of irradiated cells.

Although radiation may cause cancers at high doses and high dose rates, currently there are no data to establish unequivocally the occurrence of cancer following exposure to low doses and dose rates – below about 100 mSv (10 000 mrem). Even so, the radiation protection community conservatively assumes that any amount of radiation will pose some risk of causing cancer and hereditary effects, and that the risk is higher for higher radiation exposures. A linear, no-threshold (LNT) dose response relationship is used to describe the relationship between radiation dose and the occurrence of cancer. This dose-response model suggests that any increase in dose, no matter how small, results in an incremental increase in risk.

The LNT hypothesis is accepted by the whole scientific and regulatory community as a conservative model for determining radiation dose standards recognising that the model may over estimate radiation risk. (NCRP, 1987)

Radioactive waste requires safe long-term management because of:

- the potential dose from external irradiation that would be received by humans in close proximity to the waste and in the absence of isolation or adequate shielding;
- the potential dose due to the ingestion or inhalation of radionuclides if, for example, radionuclides in the waste were to be released to the environment; and
- the potential effects of chemically toxic materials in the waste itself or its packaging, which may make the highest contribution to toxicity in the case of some low-level wastes in case that they were disposed of deep underground (which is not the default option for low-level wastes).

The risk associated with radioactive waste can be described in terms of the probability of exposure (that is, the potential accessibility of radioactivity from the waste to humans) and radiotoxicity (that is, the intrinsic hazard that depends on waste type and quantity). Risk – in this context – is a product of impact, level and probability of exposure. Since radiotoxicity varies with time, the requirement to limit access to the waste changes with time during the various waste handling stages. (Hedin, 1997) The public sometimes perceives risk differently; this matter is addressed in Appendix 4.

Accessibility

The probability of exposure is limited by keeping radioactive elements or nuclides isolated from man and the environment, in other words by keeping their accessibility low. This is achieved in different ways depending on the type of radioactive material. Radioactive waste is managed in a series of steps. In the case of spent nuclear fuel, for example, after having been discharged from nuclear reactors, accessibility is limited by special casks during transport and by keeping the fuel submerged in water during an interim storage period. The planned disposal in bedrock greatly reduces accessibility by means of a series of engineered and natural barriers such that the return to the biosphere via underground water transport (if any) at depth is minimised. Inherent properties of the fuel, such as its very low solubility, further limit accessibility by reducing the potential for dissolution in ground water and subsequent return to humans. Additional confinement is achieved, in the case of release from the waste form, by the bedrock properties and its capability to retain radionuclides either because they have very low solubilities in a reducing environment or they get adsorbed on rock minerals, thus avoiding or limiting migration. Encapsulation can provide a further barrier between wastes and the environment.

Radiotoxicity

Each radioisotope has a different radiotoxicity, so the radiotoxic inventory of a given radioactive waste (e.g. spent fuel) is calculated by weighting the radiotoxicity of each isotope according to the quantity present. This measure of the potential for harm from the waste assumes that humans have been exposed to the radioisotopes, for example by ingestion or inhalation, or because shielding was not adequate. To convert the activity (in Bq) of the inhaled or ingested radionuclide into a human dose (in Sv), it must be multiplied by a dose factor specific to that isotope and the means of exposure (or DPUI, dose per unit intake or Sv/Bq). (CEA, 2002)

Evolution of the hazard

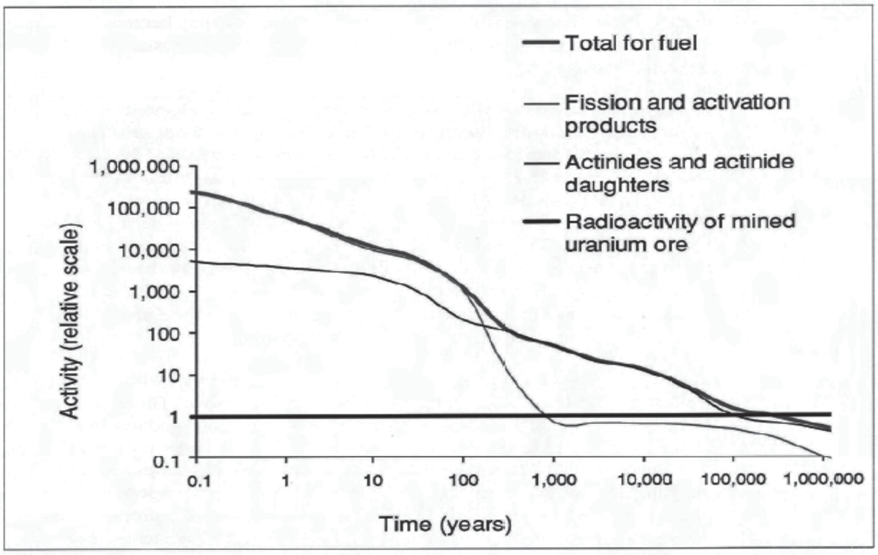
The radioactivity of waste decays significantly over time. Radioactive decay reduces the potential doses due to external irradiation and to ingestion or inhalation of radionuclides if isolation and containment are compromised at some future time. Thus, the greatest demands on a disposal system in terms of the need for protection arise at early times when the level of radioactivity of the waste is at its highest. In the case of spent fuel and vitrified high-level waste (HLW), for example, this may provide motivation for an initial period (several hundred years or more) of substantially complete containment of the waste within specially designed containers.

The half-lives of the isotopes in radioactive waste, however, vary widely. Although many (such as Strontium-90 and Caesium-137) decay substantially early in the evolution of a repository, others, such as Technetium-99, which decays with a half-life of 211 000 years, will persist for much longer. Thus, even though the hazard potential of spent fuel and some long-lived wastes decreases markedly over time, these wastes can never be said to be intrinsically harmless. Figure A1.1 below shows the progressive reduction in radioactivity of spent fuel compared to that of an equivalent amount of natural uranium ore used to manufacture the fuel. The activity is dominated during the first hundreds of years by fission products, thereafter actinides.

Radiation levels and the probability of exposure are the main indicators of safety currently used in assessment of radioactive waste disposal. Quantification of the dose-response relationship for radiation is needed for risk assessment.

Various authorities cite slightly different values for the "dose-to-risk" conversion factor for fatal cancers. Repositories are typically designed to dose constraints of up to 0.3 mSv/a, or a risk constraint of the order of 10^{-6}/a.

Figure A1.1: Relative activity of spent nuclear fuel with a burn-up of 38 MWd/kg U



Source: IAEA, 2006.

Approach to regulation

In the case of radiation, limits on exposure or dose are set and strictly observed, with legal repercussions for exceeding them. The limits themselves have been set in national frameworks, using arguments based on scientific observations on exposed persons (atomic bomb survivors, patients medically exposed, registered radiation workers) and comparisons with background radiation and with other societal risks.

The key issues to be considered when formulating regulatory standards include public health protection, the problems of extrapolating to low doses as well as long-term effects. In radiation protection for operating nuclear facilities (such as power plants), there is an international consensus for a "top down" approach based on overall dose risk limits together with a requirement of to reduce exposures below the limits, if this can be achieved taking technological, societal and economic factors into consideration to achieve an exposure "as low as reasonable achievable" (ALARA).[7] The application of ALARA to long-term disposal is not so straightforward, however, as it requires evaluation of benefits and impacts that span many generations and operational protection and long-term safety considerations must be balanced. In addition, for geological disposal, the concept of "constrained optimisation" is more often applied. In practice, these approaches call for assurance that

7. This is a principle of radiological protection in operating nuclear facilities that seeks to reduce radiation exposure to the minimum achievable levels compatible with the development of the activity involved. Some countries adopt different terminology, such as ALARP (as low as reasonably practicable).

safety criteria will be met and that sound technical and management practices be applied without offering any kind of more specific benchmark. (NEA, 2009)

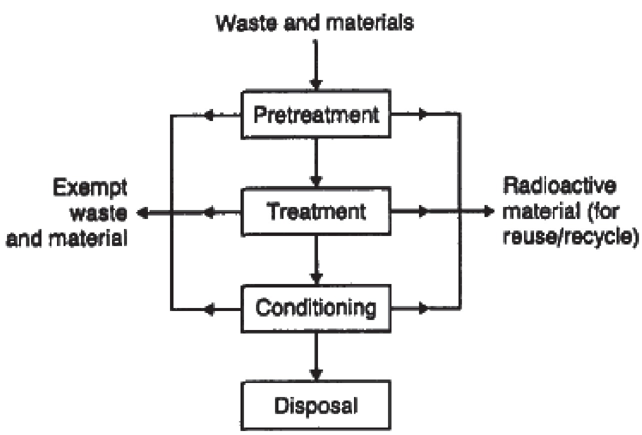
The doses from all sources of man-made radiation (with the exception of medical sources) are generally integrated. The established approach is to divide overall safety limits down into smaller levels (dose constraints) on the assumption that an individual could be exposed to more than one source. In radiation protection, the dose constraints set (typically, for a repository up to 0.3mSv/a (NEA, 1997) are much below the world average natural background radiation level of 2.4 mSv/a. (UNSCEAR, 2000) Thus, the concept of dose constraint to limit risk in order to protect future generations in respect of probabilistic events (and environmental changes) and potential exposures is also considered in the design of repositories. (ICRP, 1985)

A1.4 Overview of disposal and its implementation, including funding

Management before disposal

The first consideration is the avoidance of radioactive waste creation, if at all possible. Figure A1.2 shows the basic steps for the effective management of radioactive waste (once created) from generation through to disposal.

Figure A1.2: Basic steps in radioactive waste management



Source: IAEA, 1995.

Some IAEA definitions are: (IAEA, 2003)

- **Waste processing:** any operation that changes the characteristics of waste, including pre-treatment, treatment and conditioning.
- **Pre-treatment:** any or all of the operations prior to waste treatment, such as collection, segregation, chemical adjustment and decontamination.
- **Treatment**: operations intended to benefit safety and/or economy by changing the characteristics of the waste. Three basic treatment objectives are: a) volume reduction; b) removal of radionuclides from the waste; and c) change of composition of the waste.
- **Conditioning**: operations that produce a waste package suitable for handling, transport, storage and/or disposal. Conditioning may include the conversion of the waste to a solid waste form, enclosure of the waste in containers and, if necessary, providing an overpack.

For radioactive waste, storage is often used. Due to the decay of radionuclides, the radiation doses to operators of radioactive waste disposal facilities might be considerably lower after the waste has been subjected to interim storage for some decades. In some cases, storage is practised for primarily technical considerations, such as storage of radioactive waste containing mainly short-lived radionuclides for decay and subsequent (reduction of heat generation) prior to geological disposal. In other cases, storage is practised for reasons of economics or policy.

Options for radioactive waste disposal

Most radioactive wastes generated worldwide are LILW-SL (that is they consist mainly of short-lived radionuclides) and have been disposed of in engineered surface or near-surface engineered facilities. Concentration and containment is currently the only option used for disposal of solid LILW radioactive waste in OECD countries. Concentrating radioactive wastes into one location means that they can be contained more easily and many radionuclides will decay *in situ* to de facto insignificant levels without mobilisation into the environment.

The choice of disposal option depends primarily on a range of safety related issues, but other factors such as national and international guidance, local socio-economic factors and resource availability may also apply. It is generally accepted that, for short-lived wastes, the safety features of the repository can basically be achieved with man-made barriers (engineered barriers). On the other hand, disposal of long-lived wastes requires additional reliance on geology, in addition to engineered barriers, to prevent the return of radionuclides to the environment while natural decay of activity is taking place. Thus, for LLW, deep geological disposal is envisaged by only a few organisations. In many countries, LLW is disposed in near-surface repositories (for example France, Spain, United Kingdom). In the case of ILW, the relevant time frame for appropriate decay is in the order of ten thousand to one hundred thousand years. For SF/HLW, the relevant time frame is in the order of 10 000 to one million years.

Disposal in near-surface facilities

Radioactive wastes that decay to harmless levels within time spans ranging from some decades to a few centuries[8] are typically disposed of in engineered near-surface structures that can be designed to remain stable and intact as long as the wastes remain a hazard. (IAEA, 1999; IAEA, 2002c)

Trenches

Near surface disposal of wastes in trenches is generally applied to wastes that contain mainly short-lived radioisotopes and, potentially, low concentrations of long-lived radioisotopes. The use of trenches may be especially cost effective when disposing of large volumes of low activity wastes and/or large items of decommissioning waste. Long-term safety may be provided largely by a combination of natural site conditions, the engineered disposal system and the waste form. Designs to minimise plant and animal intrusion may also be employed. Typically, trenches are located above the groundwater level, although they may occasionally be located within the saturated zone utilising low permeability materials. Ensuring the safety of such facilities typically requires that the post-closure

8. There is no internationally agreed criterion for deciding when engineered near-surface structures are suitable for specific waste types. This issue mostly relies on the regulations of each country. It is generally considered however that those wastes principally containing radioisotopes with a half-life < 30 years (this means that their activity will decay by a factor of 1 000 in a 300 year period) are suitable for disposal in this type of facility.

institutional control period is sufficiently long (typically 60 to 100 years) so the potential risk from inadvertent intrusion is reduced to acceptable levels. The cost of this disposal option is generally lower than other approaches, though this is case specific.

Disposal in trenches has been used for many years in a wide variety of countries. Examples of trench disposal are the facilities for very low-level waste in each NPP site in Sweden (see Figure A1.3), in Morvilliers in France and in El Cabril in Spain.

Engineered disposal in near-surface facilities

For disposal of LILW with higher levels of radioactivity and/or longer-lived radionuclides, more heavily engineered disposal facilities are required, such as engineered near-surface facilities. In near-surface facilities the waste package, the disposal unit and the man-made cover, as the main engineered barriers, allow for isolation times in the order of 300 to 500 years. This can be interpreted as the time period of regulatory concern during which the barriers serve to enhance the disposal facility's isolation function. Infiltrating water is collected in a drainage system and released to the environment after being checked for possible radiological contamination. Vaults are a common type of near surface disposal facility employing engineered barriers. Vaults may be either above-ground or below-ground reinforced concrete structures, typically containing an array of storage chambers for emplacing one or more waste packages. Following emplacement of the waste, the space between the packages is generally backfilled with soil, clay or concrete grout. A low permeability capping system is placed over the backfilled disposal units to minimise the ingress of surface water and to prevent intrusion by plants and animals. The integrity of these covers is maintained during the institutional control period.

Figure A1.3: Disposal facility for VLLW at the Oskarshamn nuclear power plant (Sweden)

As in the case of trenches, there is extensive experience with this type of technology. Examples of near-surface facilities are e.g. the Centre de la Manche in France, the Centre de l'Aube also in France (in operation since 1992), Drigg in the United Kingdom (in operation since 1959), El Cabril in Spain (in operation since 1992, see Figure A1.4) and Rokkasho-mura in Japan (in operation since 1992). The capacity of these facilities varies between several 100 000 m^3 up to 1 000 000 m^3. (IAEA, 2005a)

Figure A1.4: Aerial view of El Cabril LILW disposal facility (Spain)

Disposal at intermediate depth

Specially excavated cavities or disused mine caverns at depths typically of tens of meters are examples of this option. The primary distinguishing feature of this option compared to near surface concepts is that the distance below the ground surface is usually adequate to eliminate potential intrusion by plants, animals and humans during periods of time beyond 300 years. The disposal caverns may be unlined or lined with concrete, and may incorporate a number of engineered barriers to limit or delay radionuclide migration from the disposal facility based on site-specific geological conditions and the waste characteristics. Such facilities may accept a broader spectrum of radioactive wastes including higher proportions of long-lived waste. These facilities are generally more secure against intrusion but may require more extensive barrier systems to prevent water ingress if located below the water table. In comparison with typical near surface disposal facilities, less reliance may be placed on institutional controls. In Sweden, a repository at intermediate depth (60 m below the seabed) for the disposal of low and medium waste has been operating at the Forsmark nuclear site since 1988. In Finland, two other facilities for the disposal of low and intermediate level waste were opened in 1992 and in 1998 at the Olkiluoto and Loviisa nuclear sites. Both of them are caverns excavated in granitic bedrock at depths of around 100 m below ground. Hungary decided to build its national repository for LILW following this type of concept.

Borehole disposal facility

The borehole disposal concept entails emplacement of radioactive waste in an engineered facility of relatively narrow diameter, bored and operated directly from the surface. It aims to achieve safety by a combination of natural and engineered barriers together with institutional control. Borehole disposal facilities cover a range of design concepts with depths ranging from a few meters up to several hundred meters. Borehole diameters may vary from a few tens of centimetres up to a few meters. The borehole may have a casing and the packaged waste would typically be surrounded by backfill material. A common characteristic of borehole facilities is the small relative size of the footprint at the surface, which may reduce the likelihood of human intrusion. Siting a borehole facility requires the same safety data and analysis as cavern-type facilities, but construction and operational costs may be significantly reduced, which is a consideration when disposing of small waste volumes. (IAEA, 2003) Boreholes have been implemented in several countries mostly for the long-term store of spent sealed sources.

Deep disposal in geological formations

The geological disposal systems under investigation in many national programs involve the excavation of a repository at a depth of several hundred meters in an appropriate host rock in a suitable geological environment. In the most common approach, vertical shafts or an access tunnel, or a combination of these, are first excavated to the planned depth. At this depth, horizontal disposal galleries are excavated where the waste packages are emplaced so as to be surrounded and protected by the combined engineered components and the natural barriers provided by the host rock. Geological disposal is a clear example of the "concentrate and contain" approach where containment could be achieved with reasonable expenditure of resources, in such a way as to have insignificant effect on the biosphere for many thousands of years. (NEA, 1999)

Deep geological disposal of radioactive waste (at depths of several hundred metres) is generally considered the most appropriate approach for high-level waste and spent nuclear fuel where it is necessary to isolate them from the biosphere for many thousands years. The overall objective of deep disposal is thus to isolate the wastes from the biosphere until such time as natural processes of decay and dilution prevent any radionuclide from returning in concentrations sufficient to pose an unacceptable hazard. Clearly, many processes of mobilisation, transport, retardation, retention, dilution, re-concentration, etc, need to be accounted for in evaluating whether this aim can be met, for a range of possible scenarios of future evolution of the disposal system. Geological disposal is based on the multi-barrier approach, whereby the engineered barriers and geological environment around the solid waste act together to provide a variety of "safety functions" that control any eventual releases of radioactivity from the repository and their movement through the rock.

The consensus in the scientific community is that disposal in stable geological formations is the best way to achieve the long-term management of long-lived radioactive waste. With a well designed and implemented geological disposal system, it is possible to achieve the required degree of isolation of radioactive waste from the biosphere, thus ensuring protection of human health and the environment without imposing undue burdens on future generations.

Currently, there is no geological repository in operation in the world for civilian spent fuel or HLW. A deep geological repository (WIPP) for long-lived defence-related transuranic waste with negligible heat generation is being operated near Carlsbad (New Mexico). Three sites have had a site designation for construction of a geological repository for HLW and spent fuel geological disposal: Yucca Mountain (Nevada, United States; license under review, but the United States will be evaluating alternative approaches), Olkiluoto (Finland) and Forsmark (Sweden). Several other countries have officially announced their intention of achieving this solution in the near future, including France, Switzerland and United Kingdom.

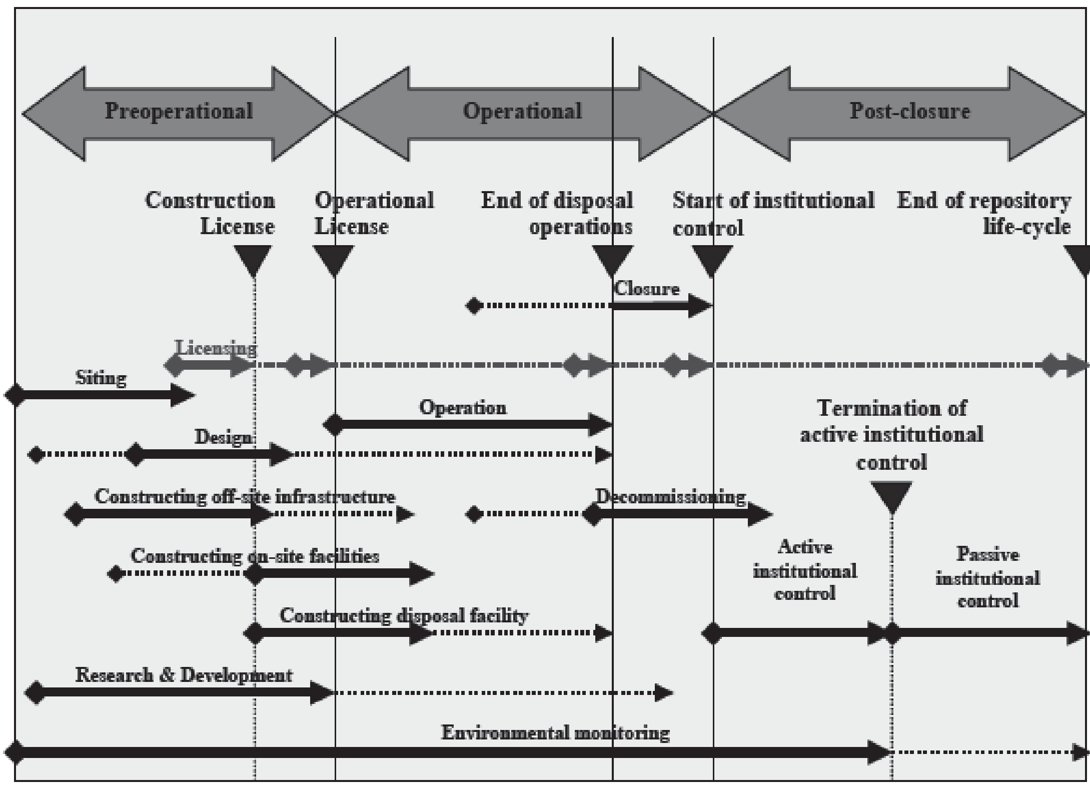
Costs and financing

Any chosen waste management strategy has to be economically viable. Achieving a cost effective solution is an important aspect of managing national liabilities and resources, but it must not preclude achieving an acceptable level of safety and an acceptable approach to the ethical issues. (IAEA, 2002d)

The life cycle of a disposal facility may be divided into the pre-operational phase, operational phase, closure and post-closure phase, as shown in Figure A1.5. The costs of a disposal facility are estimated for all its life cycle phases.

International studies typically show that for nuclear electricity, approximately 60% of the electricity generation cost represents capital costs of the power plants, 20% operation and maintenance and 20% fuel costs. (NEA, 2003b) The back-end costs (included in the fuel costs) are typically 5-10%, or up to about half the overall fuel costs, although the numerous estimates of future costs that have been made by different national programmes vary widely.

Figure A1.5: Life cycle of a disposal facility



Source: IAEA, 2007c.

These different programmes do not always include the same set of items in their cost lists. The most recent cost for the expected life cycle of the Yucca Mountain Programme (150 years, between 1983 and 2133) is projected to be 96 billion USD in 2007 money value for disposal of 100 000 tonnes of spent nuclear fuel. (US DOE, 2009) A recent study performed in the context of regional repositories compared cost estimates for several national programmes and found the disposal costs for one tonne of heavy metal to be in the order of 300 000 to 600 000. (EC, 2008) For Sweden's nuclear power programme, with a capacity of roughly 9 000 MWe, the cost of a repository to accommodate the spent fuel is estimated to be 4.3 billion. Equivalent repository related costs for HLW/SF from the 11 000 MWe United Kingdom nuclear programme are estimated at 6.3 billion.

The figures illustrate that the cost of spent fuel disposal corresponds to only a fraction of a euro cent per kWh – a minor part of the electricity prices paid by consumers.

Financing

The fundamental principle for financing the management of waste is that "the polluter pays". A second accepted principle is that no undue burdens should be imposed on future generations. Since many of the activities associated with management of radioactive waste (particularly disposal of HLW and/or SF) will take place several decades (or more) into the future, it has been generally accepted that the most prudent way is to collect the financial resources that will be needed for future operations while the waste generators are still in operation. Thus, the systems established in most countries for financing radioactive waste management are intended to provide sufficient means for funding necessary activities at the time required. (EC, 1999)

In the majority of OECD countries, the licensees that generate wastes are held responsible for paying money for building up assets for future waste management. They also have a duty to estimate the costs and update the cost estimates periodically. Generally, a review of the cost estimates is carried out by a body having economic or/and technical expertise and submitted to a competent authority, which in most countries is the Ministry of Industry and/or Energy. One of the basic differences between countries is whether the fund of assets is external or internal to the balance sheets of the producer, which naturally influences the responsibilities and roles of different bodies in the country. Both internal and external systems have been adopted in OECD countries. In countries where external management is the option, the funds are entrusted to a subsidiary of the operator or to a public organisation related to the nuclear branch, or to the state. In the internal management option, the funds are legally entrusted to the operator, but usually strictly ring-fenced in its accounts – in only a few cases the funds are fully internally managed by the operator, in the sole framework of general accountancy rules. It could be stated generally that as far as HLW and SF management is concerned, the majority of the established funds follow the external model, while a fifty/fifty proportion between internal and external funds is observed if LILW management or decommissioning funding is considered.

There is a long period between receipt of the revenue out of which waste management costs will need to be covered and the actual expenditure of those costs. This makes the accuracy of the final cost estimates that support the approved liabilities very important. Estimating means forecasting the future and the element of uncertainty always exists. Periodic updates of estimates are essential for that reason – to have the best possible forecast in use.

The philosophy of most countries is that the financial assets for radioactive waste management are collected gradually and must fully cover the liabilities by the end of the planned operational time of the nuclear facility or by some other fixed time point. In some countries, legislation requires a system where guarantees are submitted to cover any deficiencies in the funds, or the licensee has a few years to cover the liabilities that are lacking if such deficiency has been revealed in an assessment.

Fund management

In most countries that have established funds, the government itself, or a high-level organisation within the government, is designated as the financial resource management organisation. (McCombie and Tveiten, 2004) However, there are some exceptions. For example, in Spain the implementing organisation manages the funds, and in Japan a non-profit, third party body designated by the Minister performs this function. In every case, the government is responsible for developing criteria or guidelines for management of the funds. (EC, 1999 and 2000)

In the countries where the financial resources are retained internally by the waste generators, the waste generators are responsible for management of the resources. The annual amount deposited to such reserves is primarily determined by the waste generators themselves.

Usually, the funds are statutorily managed in a low risk manner (e.g. by depositing them in the national account or investing them in government bonds or according to a financing strategy established by the competent body). Finland has a unique system in which the waste generators (nuclear power plant operators) may borrow up to 75% of the accumulated funds.

In addition to collecting funds as waste is generated, any liability associated with management of waste generated prior to establishment of the financing system must also be covered. The fees for waste generated prior to establishment of the financing systems have been collected as one-time-fees upon establishment of the financing systems (in Finland and Sweden), through a series of payments over time (in Japan and Switzerland), or as a combination of both (in the United States).

Because the back-end is a relatively small part of total costs and because of the interest expected to be earned, the contributions required are relatively modest. For example, the United States levies 0.001 USD /kWh (0.0008) on nuclear electricity production, Sweden 0.01 SEK (0.001), Japan 0.13 JPY (0.001), Czech Republic 0.05 CZK (0.002), Bulgaria 3% of the electricity bill and Slovakia 6.8%. These differences reflect not only differences in national economics but also in the exact cost items covered (e.g. decommissioning is sometimes included and sometimes not). Some countries do not have an explicit levy per kWh of electricity, but they require the waste producers to set aside sufficient funding. This is the case in Switzerland where government controlled trust funds exist for both decommissioning and disposal.

A1.5 Legal and organisational infrastructure

All OECD countries have a well-defined, national legal framework to regulate the management of radioactive waste. Generally, the provisions applicable to this sector of activity are under those of the nuclear law or radiological protection regulations. In some cases governments have preferred to produce specific legislation to deal with waste management because some of its aspects, such as funding, R&D, public participation, siting and so on are unusual.

A common principle in the legislation of all countries is the acknowledgement of safety as the primary concern of all management activities and the necessity to direct all the legislative efforts to effectively achieving such an end. Accordingly, in all OECD countries the different activities involved in the management of radioactive waste require an administrative authorisation to be carried out and are permanently the subject of state supervision.

Principles such as "the polluters pays", "no burden on future generation", "minimisation of waste", etc., are generally integrated in legal texts.

Due to the high-level of international co-operation in the sector of radioactive waste management, a key document to harmonise and orient the legal approach to safety was developed in 1997 under the auspices of IAEA. The Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management, in force since 2001 (IAEA, 1997c), was drafted with the aim of being an international reference:

> "to achieve and maintain a high-level of safety worldwide in spent fuel and radioactive waste management, through the enhancement of national measures and international co-operation, including where appropriate, safety-related technical co-operation".

The Joint Convention provides framework for the safe management of radioactive waste. The Convention states that:

> "each Contracting Party shall establish and maintain a legislative and regulatory framework to govern the safety of spent fuel and radioactive waste management";

and lists six fundamental matters that are clearly well established in OECD countries:

i) the establishment of applicable national safety requirements and regulations for radiation safety;

ii) a system of licensing of spent fuel and radioactive waste management activities;

iii) a system of prohibition of the operation of a spent fuel or radioactive waste management facility without a licence;

iv) a system of appropriate institutional control, regulatory inspection and documentation and reporting;

v) the enforcement of applicable regulations and of the terms of the licences;

vi) a clear allocation of responsibilities of the bodies involved in the different steps of spent fuel and radioactive waste management.

The institutional framework in most countries has three main components:

i) the regulatory body, in charge of issuing the licences, establishing the safety requirements and supervising the different activities involved;

ii) the implementer, a specialised body responsible for discharging the duties of definitive management or disposal of radioactive waste (in most of the cases, it could also deal with predisposal activities);

iii) the producers of waste, that have to follow the rules of the regulatory body, co-ordinate with the implementer for the collection of waste and compliance with disposal requirements and provide the money to fund management activities.

The regulatory body may be the same as the regulator of nuclear safety and radiological protection and an environmental regulator may also have a role. In the majority of OECD countries, the implementer is a single body, with national coverage and usually a public organisation, solely devoted to radioactive waste management. Frequently, this agency is in charge of disposing both LILW and HLW/SF.

The development of a disposal facility requires the clear and systematic division of responsibilities between the national government, the appointed regulatory body and the operator of the facility.

The government is responsible for providing an appropriate national legal and organisational framework within which radioactive waste management could be safely undertaken and the facilities for so doing could be sited, designed, constructed, operated and closed. This latter includes the definition of the steps to be followed in the facility's development and licensing, the allocation of responsibilities, the way to secure financial and other resources, and the provision of independent regulatory functions.

The national legal and organisational framework for radioactive waste management includes:

- the definition of the national policy and strategy for the long-term management of radioactive waste of different types;
- the setting of clearly defined legal, technical and financial responsibilities for organisations to be involved in the development of disposal facilities;
- ensuring the adequacy and security of financial provisions, for example by establishing segregated funds;
- the definition of the overall process for the development, operation and closure of disposal facilities, including the legal and regulatory requirements at each step, and the processes for decision making and the involvement of stakeholders;
- ensuring necessary scientific and technical expertise is available to support site and facility development, regulatory review and other national review functions;
- the definition of legal, technical and financial responsibilities for any post-closure institutional arrangements, including any post-closure monitoring and any arrangements for ensuring the security of the disposed waste.

The function of the regulatory body is to establish the regulatory requirements for safe management of radioactive waste and for the development of disposal facilities, to set out the procedures for meeting the requirements for the various stages of the licensing process and to undertake the supervisory measures for doing so. The regulatory body sets conditions for the development, operation and closure of disposal facilities and carries out such activities as are necessary to ensure that the conditions are met.

Thus, it is the duty of the regulatory body to develop regulations, guidance and other regulatory criteria specific to disposal facilities, consistent with national policies and with due regard to the objectives and criteria. Regulations and guidance include:

- radiation and environmental protection criteria for operational and post-closure safety;
- requirements for the content of the safety case of a disposal facility;
- criteria and requirements for the siting, design, construction operation and closure of disposal facilities; and
- criteria and requirements for the waste, disposal canister, any filling material and other components of the waste package to be disposed.

The regulatory body should establish and document the procedures that it uses to evaluate the safety of disposal facilities and the procedures that operators are expected to follow in a licensing process and in demonstrating compliance with the safety requirements. The procedures and responsibilities may include:

- identification of the information to be supplied by the operator;

- review of the required submissions and assessment of the compliance with regulatory requirements;
- issuing approvals and licenses and setting conditions in conformity with legislation and regulations;
- inspection and audit of operator's data gathering, safety assessment, construction and operational activities to ensure quality and compliance with terms of approvals and licenses;
- periodical reviews of approvals, licenses and inspection procedures, to determine their continued suitability or need for amendments.

The implementer or operator of disposal facilities is responsible for its safe development and for demonstrating its safety. These functions comprise the following responsibilities:

- To carry out safety assessments and develop a safety case.
- To carry out all the necessary activities for siting, design, construction, operation and closure, in compliance with the regulatory requirements and within the national legal infrastructure.
- When designing the disposal facility and the safety case, the operator should take account of the characteristics and quantities of the radioactive waste to be disposed of, the available geological and hydro-geological conditions, available engineering and mining techniques and the national legal infrastructure and regulatory requirements.
- To conduct the research necessary to understand and support the basis on which the safety of the geological disposal facility depends. This would include all the necessary investigations of the site and materials, including packaging, assessment of their suitability and providing data for safety assessments.
  - To develop technical specifications to ensure that the disposal facility is constructed, operated and closed in accordance with the regulatory stipulations and the assumptions included within the safety case. This includes waste acceptance criteria and other controls and limits to be applied during construction, operation and closure.
  - To undertake operational and post-closure safety assessments and demonstrate the suitability of the disposal facility by the development of a safety case.
  - To keep all information relevant to the safety case and the supporting safety assessments of the disposal facility, and the records that demonstrate compliance with regulatory requirements. Such information and records should be retained until the records are transferred to another organisation that assumes responsibility for the facility.

A1.6 Safety

Safety is the highest priority in radioactive waste management. Acceptable levels of safety are usually stated in national legislation; however, a common international approach has also been agreed in the Joint Convention on the Safety of Spent Fuel Management and on Safety of Radioactive Waste Management. The Joint Convention was based on the principles of radioactive waste management established in IAEA Safety Fundamentals publications. (NEA, 1995; IAEA, 1997c) So far 46 states, parties to this Joint Convention, have agreed to take appropriate steps to ensure that at all stages of radioactive waste management individuals, society and the environment are adequately protected against radiological and other hazards. There is also agreement that before construction of a radioactive waste management facility starts, a systematic safety assessment and an environmental assessment covering both operating lifetime and the period following closure shall be carried out.

Another example of a consensus and harmonised approach to safety is the publication by IAEA of recommended safety requirements for geological disposal of high-level radioactive wastes. (IAEA, 2006) The safety requirements for near surface disposal of LILW had already been established in 1999. (IAEA, 1999)

i. Safety approach

Importance of safety in the development process

The development, operation and closure of radioactive waste repositories, especially of those intended for wastes containing long-lived radionuclides requires a significant national effort over several decades and a substantial amount of skilled human, economical and technical resources. Current plans for geological disposal in several states envisage that a disposal facility should be developed in a series of steps. Such a step-wise approach involves:

- the systematic accumulation and assessment of the necessary scientific and technical data;
- the evaluation of possible sites;
- the development of disposal concepts;
- iterative studies for design and safety assessment with progressively improving data;
- technical and regulatory reviews;
- public consultations; and
- political decisions.

During the operational period (i.e., the period when waste is being received and emplaced), the radiological protection requirements of a disposal facility and the related safety criteria are typically the same as for any licensed nuclear facility facility during its operational period. An international approach is established in IAEA's Basic Safety Standards. (IAEA, 1996) In radiological protection terms, the radiation source is under control during the operational period: releases can be verified, exposures can be controlled and actions can be taken if necessary. No release, or only very minor releases, of radionuclides and no significant doses to members of the public are expected under normal operating of radioactive waste disposal facilities. Even in the event of accidents involving the breach of a waste package, releases are unlikely to have an impact outside the facility. This will be confirmed by means of safety assessment of operational procedures, which must be sufficiently detailed and comprehensive to provide the necessary technical input for informing the regulatory body at each step.

The doses and risks associated with the transport of radioactive waste are required to be managed in the same way as the doses and risks associated with the transport of other radioactive material complying with the requirements of the IAEA Regulations for the Safe Transport of Radioactive Material. (IAEA, 2005b)

Containment, isolation, multiple barrier concepts and the concept of passive safety

The safety of a disposal facility after closure is ensured by passive means inherent in the characteristics of the site, the facility and waste packages so that no further actions are required to provide for the protection of human health and the environment in the future. Thus, safety depends on a combination of the site features, the quality of the facility design and the proper implementation of the design. Ensuring the required level of safety and quality entails developing the disposal facility in an integrated manner, on the basis of sound scientific understanding, good engineering, thorough and robust safety assessments, and with the application of quality assurance (QA) to all of these elements. The safety of disposal facilities is optimised taking into account social and economic factors.

Reasonable assurance must be provided that doses or risks to members of the public in the long term will not exceed the dose or risk level set by the national regulatory body. It is generally assumed that protection of people against the radiological hazards would also satisfy the principle of protecting the environment and therefore separate environmental limits are not often established.

As said before, the principal strategy adopted at present for achieving long-term safety of radioactive waste disposal is to concentrate and contain the waste and to isolate it from biosphere. By applying this strategy, the entry of radionuclides into the biosphere is limited and the corresponding hazards associated with the waste are considerably reduced. Safety of a disposal facility is achieved by developing a disposal system in which the various components work together to provide and to ensure the required level of protection. Thus, it is the performance of the natural and engineered barriers that provides safety in the post-closure period. The need for demonstrability requires that safety be provided by robust features whose performance is of low sensitivity to disturbing events and processes that can occur in the repository.

Accordingly, natural and engineered barriers are selected and designed to ensure that post-closure safety is provided by means of multiple safety functions. That is, safety is provided by means of multiple barriers whose performance is achieved by diverse physical and/or chemical processes. In this way, the overall performance of the repository is not unduly dependent on a single barrier or function. For example – and this is one of the main benefits of geological disposal – the geological system can be selected so that it is capable, by itself, of retaining or retarding radionuclides, such that it could provide safety at very long time frames even if, for example, the waste form or engineered barriers degrade. The presence of multiple barriers and safety functions provides assurance that, even if a barrier or safety feature does not perform fully as expected (e.g. owing to an unexpected process or an unlikely event), safety of the overall facility can still be achieved.

ii. Safety case and safety assessment

Preparation of safety cases and safety assessments

Safety assessment is the process of systematically analysing the hazards associated with a planned disposal facility and the ability of the site and designs to provide the safety functions and meet technical requirements. It includes quantification of the overall level of performance, analysis of the associated uncertainties and comparison with the relevant design requirements and safety standards. It also identifies any significant deficiencies in scientific understanding, data or analysis that might affect the results presented.

In the context of the long-term disposal of HLW and SF, safety assessments must consider periods lasting many thousands of years. Issues related to timescales are of interest in all countries considering the development of deep geological repositories. There is a consensus that there is no real justification to prescribe a specified time following which no arguments for safety need to be presented, but the nature of arguments for safety may change over time. (NEA, 2002b) Over the course of this period, changes due to natural processes and possible human action are anticipated in the repository and surrounding environment.

The safety assessment identifies possible sets of events and processes (scenarios) that could affect the performance of the disposal system and especially those that could lead to the release and transfer of radionuclides to the environment. (NEA, 1992) The behaviour of the disposal system is studied through the identification of possible future states of the repository and the use of models that simulate future repository behaviour in response to scenarios. Safety assessment studies generally utilise a central or base case scenario, which describes the normal evolution or expected performance of the disposal system and serves as a backbone to the scenario formation. The quantitative safety assessments are usually performed for periods of about 10 000 up to 1 million years. A special category of scenarios is related to future human activities that may disrupt the barrier system of a repository. This scenario is more relevant to near surface repositories, but intrusive actions by man at or close to the site are also considered in the assessment of deep geological repositories, often with separate calculations.

The safety case is an integration of arguments and evidence that describe, quantify and substantiate a claim that the repository will be safe after closure and beyond the time for which reliance can be placed on active control of the facility. (NEA, 2004) The main aim of a safety case is to establish that there is a high-level of confidence on the performance of repository barriers (both natural and engineered) so they are reliable over the required period for containment and isolation. The safety case and supporting safety assessments for review by the regulator and other interested parties are essential inputs to all the important decisions concerning the facility. It includes the output of safety assessments, together with additional information, including supporting evidence and reasoning on the robustness and reliability of the facility, its design, the design logic, and the quality of safety assessments and underlying assumptions.

A safety case evolves during repository development, providing different types of information and evidence – and at different levels of detail – suitable to support decisions at progressive stages in the development, operation and closure of a repository. At an early stage, it is used to determine the feasibility of major disposal concepts, to direct site investigations and to assist in initial decision making. In subsequent stages, it is developed to assist in system optimisation.

The safety case for a disposal facility describes all the safety relevant aspects of the site, the design of the facility, and the managerial and regulatory controls. It illustrates the level of protection and provides assurance that safety requirements will be met. With regard to post-closure safety, the possible events and processes that might affect the performance of disposal facility are considered in the safety case and supporting safety assessments, by presenting evidence that the disposal system, its possible evolutions and relevant events that might affect it are sufficiently well understood.

A1.7 Stepwise design and development for disposal facilities

1) Site characterisation and facility design

Introduction to characterisation

Characterisation of a site where a disposal facility is to be built refers to all the investigations, tests and explorations to be carried out in the existing environmental, physical and geological media in order to understand its properties and evaluate its adequacy as a host for waste isolation. The process of characterisation requires specific information on a site to establish its characteristics and the ranges of parameters relevant to disposal system performance. The site is characterised at a level of detail sufficient to support both a general understanding of the site, its past evolution and likely future natural evolution over the period of interest for safety, and a specific understanding of the impact on safety of features, events and processes associated with the site and the disposal facility. (IAEA, 1997b)

To reduce the uncertainty and risk in geological disposal, the geological properties of the site have to be well identified, and the future behaviour of the system (consisting of both geological and engineered barriers) well understood. An understanding of whether natural geological processes (e.g., new faulting, volcanic eruptions and climatic changes) can significantly jeopardise the behaviour of the system and make it unsafe must be obtained. (NAS, 2001) Site characterisation also provides the basis to reduce or compensate for uncertainties, as much as possible, by the facility configuration and design of engineered barriers. Thus, the information gathered during the characterisation stage will be used iteratively in the development of repository licensing: detailed design, safety analysis, environmental impact analysis and licensing.

R&D: the role of underground laboratories

R&D supports the demonstration of safety and feasibility of a given disposal project for HLW and spent fuel. Such R&D has historically involved extensive periods of time (15 to 20 years, sometimes more) and often demands the construction of underground research facilities; thus, it is a complex and costly process. Although not all the URLs are devoted to site characterisation, those located at potential repository sites are devoted foremost to this purpose, assessing the site through comprehensive underground experimentation, testing and validation. A particular aim of URLs is to validate different models used in assessing the performance, safety and design of the repository system: R&D work provides the means to develop and refine methods and data for testing the scientific and mathematical models used in safety assessment. It also provides practical demonstrations that can boost confidence in the disposal solutions. Underground research laboratories (URLs) play an important role in the development of geological disposal systems. Several OECD countries have run extended experimental programmes in underground research facilities over two decades. Moreover, Finland and the United States have URLs with extensive testing programmes at Olkiluoto and Yucca Mountain; in Finland the URL is at the site of the intended repository, while in the US scientific and design work have been halted until alternative approaches for the waste management programme have been evaluated.

Design

Repository design takes into account all lifecycle stages of the repository (construction, commissioning, operation, decommissioning and closure) to demonstrate that the requirements, established by the national authorities for the protection of workers, public and the environment, are met both during normal operating conditions and in the event of accidents; and for safety in the long-term, without relying on continued institutional controls, maintenance or intervention.

Deep geological disposal aims to contain and isolate waste from the biosphere. Since the components of the repository system act together to provide safety functions, all components are selected and designed to meet requirements that are established for the overall system. (IAEA, 1990) The components forming the near field[9] are generally engineered barriers and their design maximises

9. Near field refers to barriers in the immediate vicinity of the emplaced waste.

the overall performance of the natural barriers.[10] Four elements are considered as potential components of many disposal concepts for the near field:

- Waste forms are conceived to be inert and have low solubility so the release of the nuclides is constrained by virtue of the slow degradation of the waste matrixes.

- The waste container provides physical isolation of the waste form for the time it maintains its integrity (analyses for some deep geological disposal facilities show container lifetimes of more than 1 million years).

- The emplacement environment includes materials placed around the container (buffer material). This buffer material can serve various functions, including to restore the host rock integrity, to limit the rate of migration of groundwater to the surface of the waste container, to provide a chemically stable environment that supports the function of other repository components and to limit the rate of migration of radionuclides from a breached container.

- The repository sealing systems main function is to restore the host rock hydraulic properties and prevent releases of radionuclides from the repository. The seals, especially those between the disposal areas and the surface, should also be designed to resist inadvertent intrusion.

2) Waste acceptance criteria (including decommissioning waste)

Waste acceptance criteria (WAC) ensure that waste packages and their contents are compatible with the requirements for long-term management at a specific disposal facility. WAC will therefore define the properties and characteristics of waste packages that are consistent with ensuring that the waste is managed safely.

WAC are derived by identifying:

- What the method of waste management needs to achieve, and the role of the waste package within that method?

- The conditions under which the waste package will need to perform.

- The period of time for which the waste package will need to achieve its function.

- The nature and quantity of wastes that will be the subject of long-term management.

- A range of standard waste packages and the containers from which they are manufactured.

- The waste, waste form, waste container and waste package properties and characteristics that may affect the ability of the waste package to perform adequately throughout all the stages of long-term management.

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10. Near surface disposal facilities for disposal of LILW do not usually rely on geology as an isolation barrier as the long-term safety objectives of radionuclide retention can be fully achieved by means of engineered barriers. However, geology may additionally be taken into account, as was the case, for example, in the site selection process for the Centre de stockage de l'Aube in France.

3) *The phases in repository lifetime (construction, operation and closure)*

Construction

The construction period covers the time up to the commissioning of the repository and start of the operation period. The aim of the construction work is to provide the required facilities and repository capacity. The techniques used for repository construction are selected to limit deterioration of the site performance resulting from construction. In addition to the requirements for the construction work, methods for verification of the design and the construction techniques are included in the construction programme. There is also a separate programme of confirmation covering the site investigation activities that continue concurrently with construction, as it is necessary to identify changes in the natural conditions of the site caused by certain construction stages like excavation of the tunnels and caverns. Based upon this investigation programme, the predicted changes in the geomechanical, hydrogeological and geochemical conditions of the site can be checked throughout the construction period. The goal is to demonstrate that the actual conditions and any deviation from those assumed for the preliminary safety assessment will be identified and considered in an updated safety evaluation of the site (e.g. for a license to begin waste emplacement operations).

Operation

The main objective of the operation of a repository is to transfer waste packages to their final emplacement in a safe and efficient manner. Operation of a repository includes all the activities necessary to achieve the waste emplacement goals including receipt of the waste, temporary storage, waste package preparation, emplacement of waste and partial backfilling and sealing.

In the case of geological repositories for SF and HLW, it is important to note that during the lengthy period of operation of the repository, there are likely to be continuing improvements in technology; combined with on site experience, this may lead to modification or improvements to structural design features such as construction of underground openings, and backfill and sealing techniques.

Throughout the repository operation stage, a programme of ongoing testing and monitoring is expected to be carried out. Such a programme should include plans for radiation monitoring of the repository environment as well as a programme for continuing the testing and monitoring that was initiated during earlier repository stages. These operations are intended to continue after the emplacement of the last waste package and up to the time of the closure, sometimes beyond. (IAEA, 1991)

Closure

Retrievability/reversibility

Before closure commences there must be agreement between the national authorities and the repository operator that there is a sufficient level of confidence that the repository system will satisfactorily perform its function of long-term isolation of the waste. (IAEA, 2001)

The main closure activities in a geological repository could be categorised into two separated sets:

- The backfilling and sealing processes designed to limit the flow of groundwater and transport of radionuclides to the biosphere and provide structural stability, among other functions.

- The decommissioning of surface facilities to bring the site as close as possible to its original condition. These tasks include decontamination of buildings, plant and equipment.

The activities related to closure of LILW repository are well known and have a short time span (a few years). They may take longer when carried out in underground repositories, depending on the extent of backfilling and sealing that had occurred in the operation stage.

Motivated in part by the desire to bolster public confidence, the concept of retrievability has been introduced as a special feature of the geological repositories concept.[11] Retrievability may consist of an option where the engineered barriers foreseen by the disposal system are emplaced as promptly as feasible, but their emplacement is designed to be reversible. (IAEA, 2002a) Reversibility implies a disposal programme implemented in stages with options and choices open at each stage. Thus, the capacity to manage the repository with flexibility to make strategic changes over time is maintained. (NEA, 2008c) Retrievability should be undertaken in a way that does not compromise either operational safety or long-term safety. The notion of retrievability is included in many national programmes for geological disposal (e.g. Canada, Finland, France, Switzerland, the United States, etc.).

4) Monitoring programmes and post-closure and institutional controls including nuclear safeguards

Monitoring and institutional controls are crucial elements in a strategy that protects human health and the environment from the risks associated with radioactive waste. They serve, in particular, to reduce the probability of human intrusion and to bolster public confidence. According to the USDOE, besides engineered barriers, natural barriers and physical controls,

> "administrative controls are the administrative set of policies, procedures and laws that help ensure that activities or uses do not disturb physical controls, engineered barriers, or the residual contamination. Physical and administrative controls are commonly referred to collectively as institutional controls." (US DOE, 2003)

IAEA states that:

> "monitoring is the continuous or periodic set of observations and measurements of engineering, environmental and radiological parameters, to help evaluate the behaviour of components of the repository system, or the impacts of the repository and its operation on the environment".

Monitoring plays a pivotal role in the development and execution of geological disposal programs as it brings essential information for the satisfactory completion of the various phases of the repository program and thus strengthens the confidence in its long term safety. (IAEA, 2001)

The primary objective of monitoring is to provide information to assist in decision making. In this context, the key purpose of monitoring deep disposal systems is:

a. to provide information for making management decisions in a stepwise programme;

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11. "Whenever radioactive waste disposal is discussed by the public at large, the potential for making irreversible decisions always come to fore and usually broadens into discussions on ethics and decision-making, whilst exploring the unknown wishes of future generations." C. Odhnoff *in*: *Retrievability – a too simple answer to a difficult question?*, (IAEA, 2002a)

b. to strengthen understanding of some aspects of system behaviour used in developing the safety case for the repository and to allow further testing of models predicting those aspects;
c. to provide information to the society that the repository is not having undesirable impacts on humans and the environment;
d. to accumulate an environmental database for use of future decision makers;
e. to address the requirement to maintain nuclear safeguards.

Actions to be taken for the purposes of monitoring could be also classified into the categories of observation, control and protection. In the first case, monitoring is oriented to data and knowledge acquisition, modelling phenomena and making predictive calculations. Monitoring as a control tool is destined to follow observed phenomena and to take the necessary corrective actions should these parameters be out of the authorised functioning domain. Finally, monitoring for protection is used as a warning against the evolution or transition from a safe to an unsafe situation. Bearing this classification in mind it is possible to deploy the strategy shown in Table A1-6 that combines the monitoring purposes, the parts of the repository to be monitored and the lifecycle phase of this kind of facility.

The role of institutional control is to reduce the probability of intrusion into disposed waste, to reduce the magnitude of the consequences if intrusion does occur, to expedite intervention activities after intrusion has taken place and to help achieve societal confidence. Monitoring and inspection are particular forms of institutional control and are very important parts of generating societal confidence.

Table A1-7: Strategy for monitoring

	Overall Disposal system	Waste packages	Engineered barriers	Host rock	Environment geosphere-biosphere
Before construction	O(URL)	O C P	O(URL)	O	O
Construction	–	O C P	O C P	O C P	O
Operation and waste emplacement	O C –	O C P	O C P	O – P	O
Before closure	O – P	–	O	O	O
After closure	O C P	–	–	–	O

O: Observation; C: Control; P: protection; URL: Underground Research laboratory.

Source: IAEA, 2002b.

Institutional controls in geological disposal facilities are not necessarily required to ensure long-term safety but are complementary to other barriers and could help to build societal confidence. In this context, radiological monitoring is undertaken to facilitate societal confidence as there are no consequences expected to be observed for very long times. Accordingly, it is society that must decide on the period over which this monitoring might continue. Any post-closure monitoring decided by future generations should be designed in such way that no negative impacts on the performance of the containment barriers and therefore the long-term safety of the repository would occur. Markers and passive land use controls may be appropriate and passing of records and other design and decision-making information should be carried out.

A particular form of institutional control that applies to spent nuclear fuel is that of nuclear safeguards. These apply to spent fuel where the amount of fissile material is above the level considered to be practically recoverable under the Non Proliferation Treaty. It would also apply to weapons grade plutonium if it were considered a waste and placed into a repository. The key issue for safeguarding waste is to ensure that any measures taken to verify the materials do not significantly compromise the overall safety of the repository. Conversely, it would also be important to ensure that any retrievability measures for geological disposal do not violate safeguards requirements to limit access. In the long-term it is generally viewed that the same measures that guard against inadvertent human intrusion would, at least in some degree, also address safeguards.

A1.8 Challenges in the near future

Some key challenges in the next 10 years are discussed below.

Public acceptance is judged the primary challenge especially for geological disposal of HLW and SF. The NEA has already noted: (NEA, 2008c)

> "…confidence by the technical community in the safety of geological disposal is not, by itself, enough to gain public confidence and acceptance. There is consensus that a broadly accepted national strategy is required. This strategy should address not only the technical means to construct the facility but also a framework and roadmap allowing decision makers and concerned public the time and means to understand and evaluate the basis for various proposed decisions and, ultimately, to gauge whether they have confidence in the level of protection that is being indicated by the implementing organisation and evaluated by the regulator through its independent review."

Other near term challenges fall into three categories: technology, legislation and regulation. In the area of technology, there is clear international consensus that geological disposal is technologically feasible and can be safely implemented. Nevertheless, ongoing R&D can further support the implementation of waste management solutions by technological innovations and by improving understanding and reducing uncertainties. Knowledge retention will be an important challenge.

In the area of legislation, there is a consensus that radioactive waste management is an issue that is being adequately addressed in OECD countries. Legislation requires progressive adaptation to new societal situations and technical developments, basically arising from the expected implementation of national policies on HLW and SF disposal. In this context, a key issue will be the legislative and regulatory definition of the concepts of reversibility and retrievability of a repository. Again in the words of NEA: (NEA, 2008c)

> "…reversibility and retrievability are considered by some countries as being important parts of the waste management strategy… There is general recognition that it is important to clarify the meaning and role of reversibility and retrievability for each country, and that provision of reversibility and retrievability must not jeopardise long-term safety."

There is a clear framework for legal and regulatory issues. Radioactive waste management – as with decisions on investment and priorities in the overall energy mix – would benefit from more continuity and stability on the part of decision makers and greater independence from day-to-day political concerns. This would be expected to allow better use of allocated resources and reduced implementation timescales.

Funding must be provided at adequate levels. Past funding deficits originating from times when the principles of "polluter pays" and "no undue burden to future generations" were not in force should be provisioned as soon as possible.

Regulatory challenges may arise as a consequence of having to address successive applications for licensing disposal facilities or repositories. Disposal of LILW is an internationally tested practice either in near-surface facilities or in deep repositories. There is considerable regulatory experience in this area that has been shared and contrasted in international organisations like NEA and IAEA and that is helping countries that are new to LILW repositories. However, no underground repository for HLW/SF has yet been licensed and although, globally, the first application was submitted in June 2008 by the US DOE for Yucca Mountain, scientific and design work have been halted and there are plans to evaluate alternative approaches for the waste management programme. The complexity of the documentation involved in the submissions for this type of facility is considerable.

The challenge for policymakers is to align the emerging consensus of the scientific and technical community concerning the feasibility and safety of underground repositories for high-level waste and spent fuel with both the continuing high level of public anxiety concerning such installations and the very stringent regulatory requirements with regards to both performance (extending up to one million years) and procedure.

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Appendix 2

STRATEGIC ISSUES FOR HAZARDOUS WASTE

The intention of this chapter is to provide an overview of the main strategic issues associated with managing non-radioactive hazardous waste. Hazardous waste covers a far broader spectrum of materials and objects than does radioactive waste. Whilst Appendix 1 looks at strategic issues for radioactive waste primarily in an international context, this appendix considers hazardous waste primarily using national examples, mainly from Germany and the United States. These countries were chosen because of the availability of expertise in the expert group that produced this report.

Section 1 of this appendix sets out some hazardous waste definitions, classifications, and then outlines global production rates. Section 2 explains the generally accepted ethics and principles for disposal and Section 3 describes the options for managing hazardous waste. The hazards and risks associated with hazardous waste management are discussed in Section 4. An overview of landfill and underground waste management facilities and their implementation is presented in Section 5. Matters associated with the legal and organisational infrastructure are described in Section 6, whilst Section 7 considers the crucial matter of safety in managing hazardous waste streams. Finally, Section 8 describes the development of landfill and geological disposal facilities.

A2.1 Waste and hazardous waste definitions, classification schemes and quantities

Waste includes all items that people no longer have use for, which they either intend to get rid of or have already discarded. Additionally, wastes are items which people are required to discard by law because of their hazardous properties. Many items can be considered as waste e.g. household waste, sewage sludge, wastes from manufacturing activities, packaging items, discarded cars, old TV sets, garden waste, old paint containers, etc. Thus, all our daily activities give rise to a wide variety of different wastes arising from different sources.

There are a number of slightly differing waste definitions. The Basel Convention[1] (Basel, 1989) on the control of transboundary movements of hazardous wastes and their disposal, the OECD (OECD, 2001), the EU and individual countries each have their own definitions.

The Joint Questionnaire OECD/Eurostat sent biennially to all European countries provides the following broad definition of waste:

> *Waste refers here to materials that are not prime products (i.e. products produced for the market) for which the generator has no further use for his own purpose of production,*

1. The Basel Convention on the Control of transboundary movements of hazardous wastes and their disposal is the most comprehensive global environmental treaty on hazardous and other wastes. It has 170 member countries (Parties) and aims to protect human health and the environment against the adverse effects resulting from the generation, management, transboundary movement and disposal of hazardous and other wastes.

transformation or consumption, and which he discards, or intends or is required to discard. Wastes may be generated during the extraction of raw materials, during the processing of raw materials to intermediate and final products, during the consumption of final products and during any other human activity.

The definition of hazardous waste in the Basel Convention, and in OECD and EU documents, is based on categories,[2] made up of waste streams and constituents. Different types of hazardous wastes exhibit one or more hazardous characteristics (these are described in Section A2.4). (BC, 1989; HWD, 1991; OECD, 1998; EWL, 2000; OECD, 2001)

Some examples of hazardous waste streams are:

- clinical wastes from medical care in hospitals, medical centres and clinics;
- wastes from the production, formulation and use of biocides;
- wastes from the manufacture, formulation and use of wood preserving chemicals;
- wastes from the production, formulation and use of organic solvents;
- waste mineral oils unfit for their originally intended use;
- waste substances and articles containing or contaminated with polychlorinated biphenyls (PCBs);
- wastes of an explosive nature not subject to other legislation;
- residues arising from industrial waste disposal operations.

Some examples of hazardous waste constituents are:

- metal carbonyls;
- hexavalent chromium compounds;
- arsenic; arsenic compounds;
- cadmium; cadmium compounds;
- mercury; mercury compounds;
- inorganic cyanides;
- acidic solutions or acids in solid form;
- asbestos (dust and fibres);
- organic phosphorus compounds;
- phenols;
- halogenated organic solvents.

2. Complete lists of waste categories and waste constituents as set out in the OECD (1998) Waste Definitions can be seen at www.oecd.org/dataoecd/57/l/42262259.pdf.

A2.1.1 Waste classification schemes

Some countries have their own national waste classification schemes; some are using the Basel classification scheme, whilst others have implemented the European Waste List (EWL).

Situation in Europe

The creation of the EWL represents the most significant move to date towards harmonisation of information on waste generation and management in Europe and the development of a common Europe-wide waste classification system for hazardous and non-hazardous waste.

The European waste classification system was established in December 1993 by Council Decision 94/3/EC and revised in 2000 and 2001. The EWL of 2001 comprises 849 entries of which 404 are considered to be hazardous waste. In general, the EWL is a process-based and source listing of wastes. The EWL has three levels that describe the waste source, the process generating the waste and the substances in the waste. Not all Member States of European Union have fully implemented the EWL into national legislation and data registration systems.

Situation in the United States

In the United States, hazardous wastes can be liquids, solids, contained gases, or sludges and can be by-products of manufacturing processes or simply discarded commercial products, like cleaning fluids or pesticides. Hazardous waste is defined by the Resource Conservation and Recovery Act (RCRA) as one that appears on one of four hazardous wastes lists (the F, K, P and U-lists), or as a waste that exhibits at least one of four characteristics: ignitability, corrosivity, reactivity or toxicity. (EPA, 2006a)

The F-list identifies wastes from non-specific sources in common manufacturing and industrial processes, such as solvents that have been used in cleaning or degreasing operations. The K-list includes certain wastes from specific industries, such as petroleum refining and pesticide manufacturing. The P-list and the U-list include specific commercial chemical products in an unused form (discarded commercial chemical products), and can include some pesticides and some pharmaceutical products.

The US hazardous waste characteristics are:

Ignitability: wastes that can create fires under certain conditions, are spontaneously combustible, are oxidisers, are compressed gases that are flammable under certain conditions or are liquids that have a flash point less than 60°C (for example waste oils and used solvents).

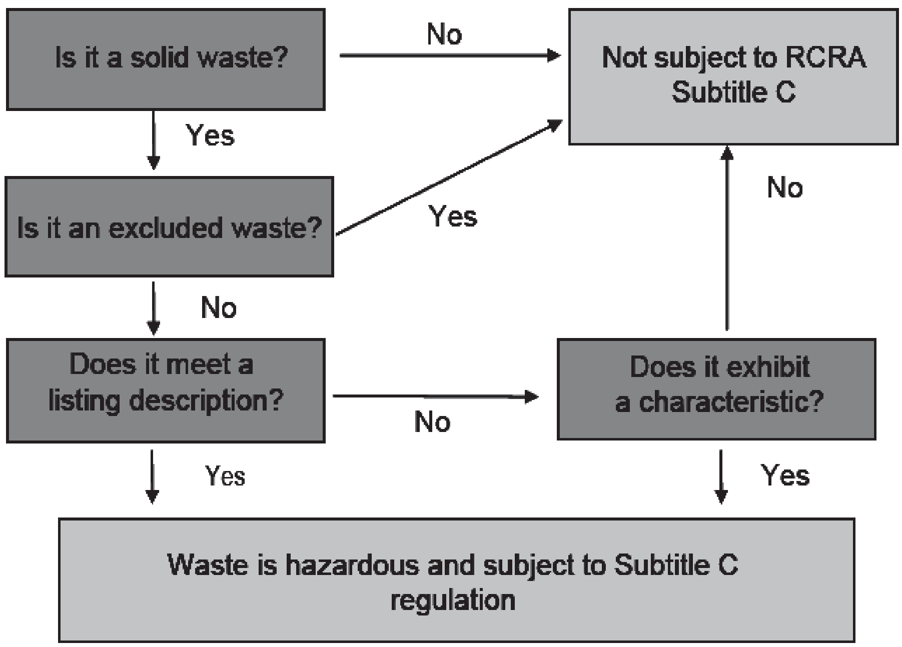
Corrosivity: liquid waste that are strong acids or bases (pH less than or equal to 2, or greater than or equal to 12.5) and/or are capable of corroding metal containers, storage tanks, drums, and barrels (for example battery acid).

Reactivity: wastes that are unstable under "normal" conditions and can cause explosions, toxic fumes, gases, or vapours when heated, or mixed with water (for example lithium-sulphur-dioxide batteries that have not been discharged, or explosives).

Toxicity: wastes that contain constituents that are harmful or fatal when ingested or absorbed (for example mercury or lead) and that can potentially pollute ground water if they leach out of the waste. Toxicity is defined through a laboratory procedure called the Toxicity Characteristic Leaching Procedure (TCLP). (EPA, 1992)

The Resource Conservation and Recovery Act (RCRA) Subtitle C, establishes a federal programme to manage hazardous wastes from cradle to grave [40 CFR, 42 USC]. Hazardous waste is defined as a subset of solid waste. Generators of waste are responsible for determining if a waste is hazardous, with their responsibility beginning at the point the waste is generated. A generator may use test results or process knowledge in making the determination. The overall process for hazardous waste identification in the United States is presented in Figure A2.1.

Figure A2.1: The US Hazardous Waste Identification Process



The objective of the Subtitle C programme is to ensure that hazardous waste is managed in a way that protects human health and the environment, and to this end, Subtitle C includes regulations for the generation, transportation, and treatment, storage, or disposal of hazardous wastes. In practical terms, this means regulating a large number of hazardous waste handlers. As of 2003, EPA had on record approximately 600 treatment, storage and disposal facilities (TSDFs), 18 000 transporters, and 16 000 large quantity generators (LQGs).[3] The Subtitle C programme is a comprehensive set of environmental regulations that cover the treatment and management of hazardous wastes. The regulations first identify the criteria to determine which solid wastes are hazardous and then establish various requirements for the three categories of hazardous waste managers: generators, transporters and TSDFs. In addition, the Subtitle C regulations set technical standards for the design and safe operation of TSDFs. Almost all hazardous wastes produced within the United States are treated and disposed of within the country.

3. Large quantity generators, under RCRA in the United States, are defined as those facilities that generate: 1 000 kg or more of hazardous waste per calendar month (approximately 2 200 lbs) or 1 kg or more of acutely hazardous waste per calendar month (approximately 2.2 lbs). (EPA, 2006a)

A2.1.2 Annual production rates of different types of waste

Worldwide, 8-10 billion tonnes of wastes are currently generated annually (this figure excludes mining and milling wastes, which are not normally counted). Of this, over 400 million tonnes is hazardous wastes. Within the OECD area, around 4.5 billon tonnes of wastes are generated annually, of which 150-200 million tonnes is hazardous.

Over 2 billion tonnes of waste – including hazardous waste – is generated each year in the European Union. This is equivalent to 3.8 tonnes per person. Most of this waste comes from households, commercial activities (e.g., shops, restaurants and hospitals), industry (e.g., pharmaceutical production and clothes manufacturing), agriculture (e.g., slurry), construction and demolition projects, mining and quarrying activities and from the generation of energy.

With such vast quantities of waste being produced, it is vitally important that it be managed in a way that minimises harm to human health and the environment. Although hazardous waste represents only 3% of waste generated in Europe, it is subject to special legislation and requires special management arrangements to ensure that it is kept separate and treated differently from non-hazardous waste.

In the United States, industrial wastes account for about 0.5 billion tonnes of which about 35 million tonnes are classified as hazardous. (OECD, 2008)

Table A2.1: Generation of hazardous waste in selected OECD member countries (tonnes per year)

Country	Waste definition*	2000	2001	2002	2003	2004
Austria	N	1 035 000	1 026 000	920 000	n.a.	1 014 000
Czech Republic	N	263 000	2 817 000	1 311 000	1 219 000	1 447 000
Denmark	N	183 000	200 000	248 000	328 000	342 000
Finland	N	963 000	827 000	1 188 000	n.a.	2 349 000
France	N	9 150 000	n.a.	n.a.	n.a.	n.a.
Germany	N	14 937 000	15 830 000	19 636 000	19 515 000	18 401 000
Greece	N	391 000	326 000	353 000	354 000	n.a.
Hungary	B	951 000	893 000	543 000	n.a.	n.a.
Italy	N	3 911 000	4 279 000	5 025 000	5 440 000	5 365 000
Korea	N	2 779 000	2 858 000	2 915 000	2 913 000	n.a.
Poland	N	1 601 000	1 308 000	1 029 000	1 339 000	1 349 000
Slovak Republic	N	1 627 000	1 663 000	1 441 000	1 258 000	1 021 000
Spain	N	3 063 000	3 223 000	3 223 000	3 223 000	3 534 000
Sweden	B/N (2004)	1 100 000	n.a.	n.a.	n.a.	1 354 000
United Kingdom	N	5 419 000	5 526 000	5 370 000	4 991 000	5 285 000
United States	N	n.a.	37 033 000	n.a.	27 376 000	n.a.

Notes: * Waste definition: N – National or other definition including the EWL; B – Basel Convention.

Source: OECD, 2007a.

The OECD publishes data on hazardous waste generation, as shown in Table A2.1. However, data are scarce and are generally based on national classifications and definitions that make it difficult to draw valid comparisons between different countries. OECD hazardous waste statistics do not generally provide information on the composition of hazardous waste generated.

Because of different national definitions, data on hazardous waste from different countries are not directly comparable. It is currently not possible to say to what extent the variations found in the reported statistics can be explained by different:

- classifications of hazardous waste;

- systems and obligations for collecting hazardous waste;
- reporting systems on hazardous waste data;
- industrial structures;
- levels of application of cleaner technology and other waste reduction methods.

Therefore, comparison of data on hazardous waste from one country to another must be made with caution. (EEA, 1999; EEA, 2002)

A2.2 Ethics and principles for final disposal

Policies and practices for hazardous waste management have evolved over a long time and differ between the OECD member countries. The following principles have been used to varying extents by many countries in developing their waste management strategies. (BC, 1995)

a) *The Source Reduction Principle* – the generation of waste should be minimised in terms of its quantity and its potential to cause pollution. This may be achieved by using appropriate plant and process designs; e.g., efficient processes in manufacturing, reduction of disposable material in consumer goods or increase in product durability.

b) *The Integrated Life-cycle Principle* – substances and products should be designed and managed such that minimum environmental impact is caused during their generation, use, recovery and disposal.

c) *The Precautionary Principle* – preventive measures should be taken, taking account of the costs and benefits of action and inaction, when there is a scientific basis, even if limited, to believe that release to the environment of substances, waste or energy is likely to cause harm to human health or the environment.

d) *The Integrated Pollution Control Principle* – the management of hazardous waste should be based on a strategy which takes into account the potential for cross media and multi-media synergistic effects.

e) *The Standardization Principle* – standards should be provided for the environmentally sound management of hazardous wastes at all stages of their processing, treatment, disposal and recovery.

f) *The Self-sufficiency Principle* – countries should ensure that the disposal of the waste generated within their territory is undertaken there by means which are compatible with environmentally sound management, recognising that economically sound management of some wastes outside of national territories may also be environmentally sound.

g) *The Proximity Principle* – the disposal of hazardous wastes should take place as close as possible to their point of generation, recognising that economically and environmentally sound management of some wastes may be achieved at specialised facilities located at greater distances from the point of generation.

h) *The Least Transboundary Movement Principle* – transboundary movements of hazardous wastes should be reduced to a minimum consistent with efficient and environmentally sound management.

i) *The Polluter Pays Principle* – the potential polluter must act to prevent pollution and those who cause pollution pay for remedying the consequences of that pollution.

j) *The Principle of Sovereignty* – countries should take into account political, social and economic conditions in establishing a national waste management structure. A country may, for example, ban the importation of hazardous wastes into its territory in accord with its national environmental legislation.

k) *The Principle of Public Participation* – countries should ensure that in all stages, waste management options are considered in consultation with the public as appropriate, and that the public has access to information concerning the management of hazardous wastes.

Principles f), g) and h) are clearly related. It should be recognised that considerations for waste disposal might be different from those for recovery, which, if soundly managed, provides environmental and economic benefits that should be encouraged.

Economic, social, technical and institutional issues affect how a particular region or country chooses specific policies with respect to waste management. Industrial activity inevitably generates by-products or wastes in addition to the goods and services that are directly produced. Since industrial growth is a goal of most countries, the question of how to deal with these wastes will eventually arise. Many countries have experienced adverse consequences resulting from improper management of certain hazardous wastes; there is an abundance of data concerning many sites where such wastes were deposited inappropriately. Costs of remedial action are often extremely high, and the threat of adverse health and environmental effects may never be completely removed.

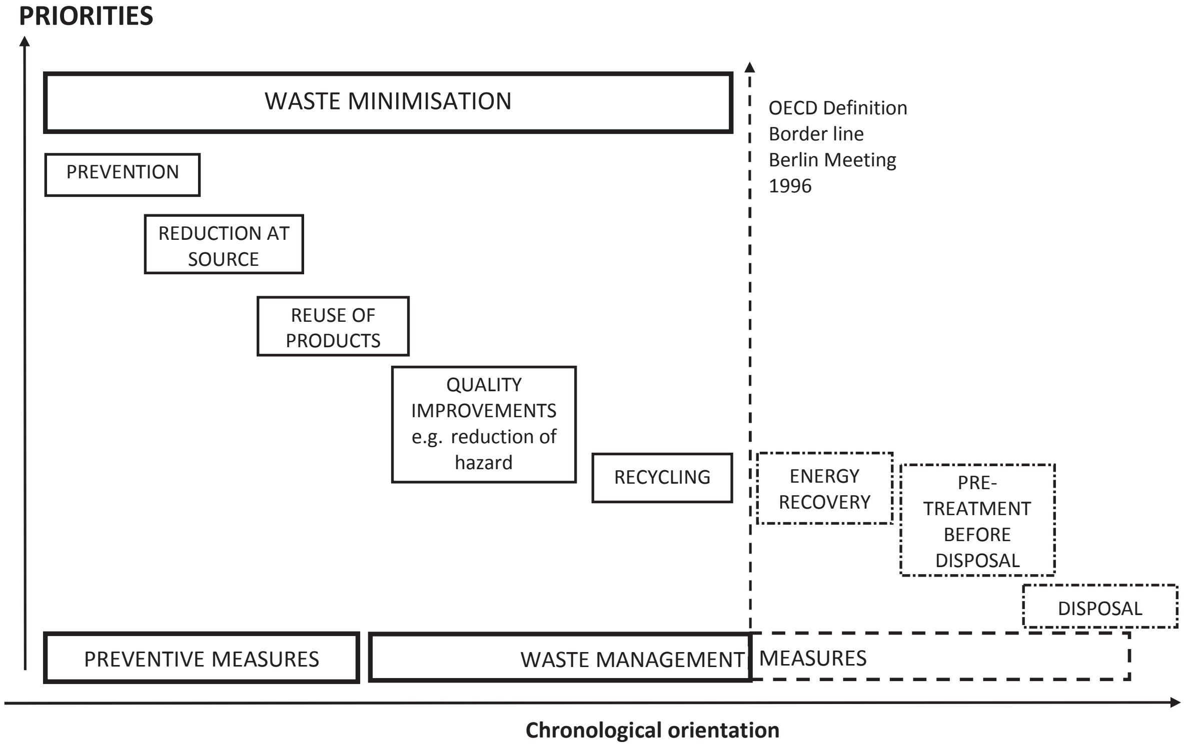
In one form or another, and with certain national variations, these principles form the foundation for all active systems of hazardous waste management. A number of factors bear on how individual countries choose to emphasise particular aspects, including cost, geography, industrial mix, public awareness and legislative mandate.

It has been said (Kummer, 1995) that "a future waste management system should be primarily global, holistic and integrated, and should focus on the preventive approach. It should however, make allowance for the adoption of regional rules … and for the application of existing sectoral rules that are in line with its objectives and provisions."

A2.3 Hazardous waste management options

There are many options available for the treatment and management of hazardous waste, including avoidance, source reduction, minimisation, reuse, recycling, energy recovery and disposal. Stockpiling waste is not a viable solution. The best solution always is to prevent the generation of such waste, reintroducing it into the product cycle by recycling its materials or components where there are ecologically and economically viable methods of doing so. Many countries view disposal as the last resort, which should only be used when all the other options have been exhausted, i.e., only material that cannot be avoided, reduced, reused, recycled or otherwise treated (including by incineration) should be sent to landfill. Nearly all approaches to waste management are based on a waste hierarchy. An example of such a waste hierarchy is the OECD working definition of waste minimisation as shown in Figure A2.2.

Figure A2.2: OECD working definition of waste minimisation



Prevention: Strictly avoiding waste generation, both qualitatively (through virtual elimination of hazardous substances) and quantitatively (through reducing material or energy intensity in the production, consumption and distribution of commodities).

Reduction at source: Minimising use of toxic or harmful substances; minimising material or energy consumption.

Reuse: Multiple use of a product in its original form, for its originally intended purpose or an alternative purpose, with or without reconditioning.

Recycling: Using waste materials in manufacturing other products of an identical or similar nature.

Energy recovery: Utilising the energy content of waste materials with or without pre-processing.

Pre-treatment before disposal: Reducing volume, mass or toxicity before sending to landfill or final storage by mechanical, physical, chemical or biochemical processes.

Source: OECD, 1997.

Implementation and operation of facilities for hazardous waste are market-oriented processes. Major enterprises in which large quantities of waste are generated have their own disposal facilities such as incinerators, chemical and physical treatment plant and landfills, with transportation often done by the generator of the wastes. Small- and medium-sized enterprises, and large enterprises with small accumulations of waste, usually make use of third parties to collect and transport it. Small quantities are also delivered to collection sites/transfer stations operated by waste disposal companies or public bodies. There, they are compiled into larger batches in accordance with the requirements of further treatment and disposal.

Short-term storage of hazardous waste has several objectives. The main objective is to store the waste safely before it is introduced as feed into the waste treatment process. Another reason is to provide adequate accumulation time, e.g. to collect an economically viable amount of waste prior to treatment. Temporary storage can also be used for the purpose of control and inspections.

Treatment options are selected according to the composition and the hazard of the waste. Some hazards can be destroyed by treatment methods. For example the Stockholm Convention dealing with persistent organic pollutants (POPs) stipulates that POPs shall be:

> "…handled, collected, transported and stored in an environmentally sound manner and disposed of in such a way that the POP content is destroyed or irreversibly transformed … or otherwise disposed of in an environmentally sound manner when destruction or irreversibly transformation does not represent the environmentally preferable option or the persistent organic content is low." (SC, 2004)

The purpose of chemical, physical and biological treatment is to prepare the wastes so that they can be deposited or incinerated without harm to the environment, or perhaps be recycled. (BATWT, 2006)

Two main categories of hazardous wastes are treated by chemical and physical methods:

- *Wastes with mainly inorganic pollutants:* Examples are acid, solutions of heavy metals, cyanide, nitrite, and chromate. They originate mostly in the chemical and automotive industries.
- *Wastes with mainly organic pollutants:* Examples are oily wastewater, synthetic coolants and lubricants, rinsing and wash water with organic pollutants from the metalworking and automotive industries, from tank and vessel cleaning and related sources.

Incineration is used as a treatment for a wide range of wastes. It is available on an industrial scale, for which comprehensive knowledge and data is available. It allows the hazard from a large number of substances to be greatly reduced. (BATWI, 2006)

Incineration allows:

- minimisation of solid, liquid, and semi liquid wastes which cannot be sent to landfill or treated chemically or physically without harm to the environment;
- minimisation of the hazard potential of harmful substances in the wastes;
- substantial reductions in volume and weight;
- recovery of the energy released.

The incineration sector has undergone rapid technological development over the last 10 to 15 years. Much of this change has been driven by legislation specific to the industry aimed at reducing airborne emissions.[4] Continual process development is ongoing, with the sector now developing techniques that limit costs, while maintaining or improving environmental performance. Incinerators fulfilling the limit values of the European Waste Incineration Directive and operated according to best available techniques (BAT) do not significantly harm health. (Bachmann, 1993)

The use of BAT[5] ensures that waste is managed in an environmentally sound manner within a particular waste management facility. The use of BAT is a policy approach that a number of OECD

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4. The main problems caused by incineration are emissions of organic micro-pollutants such as polychlorinated dibenzo-p-dioxins and polychlorinated dibenzo-p-furans or the release of volatile metals such as mercury, cadmium and lead which can be transported over long distances.

5. Use of best available techniques implies the use of technology, processes, equipment and operations that are based on scientific knowledge, whose functional value has been successfully tested in operative comparable plants.

countries (primarily within the EU) are using through national or international regulations with the aim of bringing about environmental benefits while still achieving economic viability.

The general EU approach to BAT has been developed in the framework of the Integrated Pollution Prevention and Control (IPPC) policy in 1996.[6] The EC Directive on IPPC aims at preventing and controlling pollution[7] for 33 identified industrial sectors, including part of the waste sector. To achieve this goal, industrial installations have to apply, *inter alia*, general principles, including the application of BAT. Within the EU, BAT is a legal or regulatory requirement that is used as a criterion by competent authorities to grant licences or permits to industrial installations. The EU approach to BAT forms the basis for the setting of emission limit values and the operating conditions included in the permitting procedure for installations. It is defined as:

> "…the most effective and advanced stage in the development of activities and their methods of operation which indicate the practical suitability of particular techniques for providing in principle the basis for emission limit values designed to prevent, and where that is not practicable, generally to reduce emissions and the impact on the environment as a whole."

Public participation in hazardous waste management – example from the United States

The RCRA hazardous waste permitting programme actively involves the public in decision-making by providing equal access to information and an opportunity to participate in the hazardous waste permitting process. This integration is achieved through a public involvement policy, (EPA, 2003) where the term "public involvement" encompasses the full range of actions and processes that EPA uses to engage the public in its work. This policy applies to all EPA programmes and activities, including RCRA.

The emphasis on public participation comes from the recognition that the hazardous waste management process, particularly the aspects associated with siting of a waste management facility, is not simply a technical problem, it also has social, economic and political dimensions. (EPA, 1997)

One important aspect of the social dimension is environmental justice, which refers to the fair distribution of environmental risks across socio-economic and racial groups. EPA addresses environmental justice on a local level and on a site-specific basis, encouraging permitting agencies and facilities to use all reasonable means to ensure that all segments of the population have an equal opportunity to participate in the permitting process and have equal access to information in the process. Some states have also adopted environmental justice provisions.

A2.4 Hazards and risks from hazardous waste management

Hazardous waste is any waste with properties that make it dangerous or potentially harmful to human health and the environment. The universe of hazardous wastes is large and diverse. Hazardous

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6. See Directive 96/61/EC (24 September 1996), concerning Integrated Pollution Prevention and Control, as amended by Directives 2003/35/EC (26 May 2003) and 2003/87/EC (13 October 2003), and Regulation (EC) n° 1882/2003 (29 September 2003).

7. See Annex I of Directive 96/61/EC (24 September 1996), concerning Integrated Pollution Prevention and Control: "Categories of Industrial Activities Referred to in Article 1".

wastes can include liquids, solids, contained gases, or sludges. They can be by-products of manufacturing processes or simply discarded commercial products, like cleaning fluids or pesticides.

Nature of hazard

The characteristics that make a waste hazardous, according to the OECD are: (OECD, 2001)

- explosive;
- flammable liquids or solids;
- liable to spontaneous combustion;
- emit flammable gases in contact with water;
- oxidising;
- poisonous (acute);
- infectious;
- corrosive;
- liberate toxic gases in contact with air or water;
- toxic (delayed or chronic);
- ecotoxic;
- capable, by any means after disposal, of yielding another material that possesses any of the characteristics listed above.

As with radioactive waste, the risk associated with hazardous waste generally depends on the quantity and composition of the waste and on the length of exposure.

The risk associated with a waste is generally taken to be the product of the probability of exposure and the consequence of exposure to the toxic components of the waste. The probability of exposure to hazardous waste is minimised by reducing the accessibility; the consequence of exposure is dependent on the quantity and composition of the waste.

Accessibility

Exposure to hazardous waste is limited by keeping the accessibility low. This is done in different ways depending on the type of hazardous material in the waste. During collection, transport and handling of hazardous waste, measures comparable to those in the chemical industry (protective clothing, dedicated transport casks, etc.) must be taken to avoid health risks. Leakage or spills to the environment are carefully avoided.

After volume reduction and treatment, the residues are disposed of. Final disposal in geological formations (e.g. in rock salt as in Germany) greatly reduces accessibility by means of a series of engineered and natural barriers. The inherent properties of the waste after treatment, such as its low solubility, usually limit transport though the environment and eventual human exposure.

Evolution of the hazard

Hazards that can be destroyed by treatment methods are normally treated, for example by incineration, before disposal. However, some dangerous substances, like toxic heavy metals, do not change their toxicity over time. These wastes can be regarded as having an infinite half-life, so they require isolation from the biosphere over extremely long timescales.

Mitigating the adverse impacts of hazardous waste management

Public and political concerns over the environmental impacts of the increasing volume and toxicity of hazardous wastes have grown dramatically in the last three decades. Improper management of waste has caused numerous cases of contamination of soil and groundwater and threats to the health of the exposed population. (EEA, 2000) Environmental impacts of these increasing waste volumes and toxicities are strongly influenced by waste management methods and practices.

Historically, waste disposal practice has followed the path of least resistance and of lowest costs. Several factors have driven the development towards landfills and shallow land disposal methods; these include the relatively low cost of land and land disposal procedures, the low capacities of other disposal technologies and the economic consequences of environmental legislation at both regional (e.g. European Union) and national levels whose principal objective was the protection of water and air quality.

Problems related to the emission of gases from above ground landfill sites are mainly caused by biological degradation of organic materials. The EU Landfill Directive, when fully implemented, will result in a reduction in organic inputs to landfills so this problem is likely to decrease in the coming years in the EU Member States. Germany stopped sending biodegradable wastes to landfill in mid-2005. (MWLO, 2001)

Risks associated with landfill can be controlled by good operational practices, by exercising tight control over the type of wastes accepted into the landfill and by proper treatment and management of emissions to atmosphere and water. Although leachate from landfills has potentially high concentrations of heavy metals, organic substances and salts, the risks associated with this can be reduced by appropriate wastewater treatment prior to discharge.

In the United States, one of the major risks posed by waste in landfills is the threat of groundwater contamination. EPA employs a three-tiered groundwater protection strategy using land disposal restrictions (LDR), land disposal units (LDU) and groundwater monitoring (GWM). LDRs are the first line of defence since hazardous waste placed on the ground poses a potential contamination risk to groundwater. LDR requirements apply to the entire cradle-to-grave chain (i.e., generation to disposal) and LDR treatment standards reduce the toxicity and mobility of each hazardous constituent.

The LDR programme prohibits three activities:

- disposal of untreated hazardous waste;
- storage of hazardous waste for long periods of time to avoid proper treatment;
- dilution of hazardous waste to meet treatment standards, unless the treatment standard is specified as "deactivation".

Wastes must be treated to achieve LDR treatment standards prior to land disposal. Treatment standards are usually based on an evaluation of the best demonstrated available technologies (BDAT) for treatment of a hazardous waste. To be specified as BDAT, a treatment technology must have the demonstrated ability to treat the hazardous constituents present in the waste stream and it must be available for public use. The hazardous waste must be treated in one of two ways:

- by using any treatment technology (other than impermissible dilution) to meet constituent concentrations (e.g., 0.05 mg/L);
- by using specified BDATs for the hazardous waste (e.g., combustion).

EPA prohibits the storage of waste as a substitute for treatment. TSDFs cannot store waste unless the storage is to accumulate sufficient quantities of waste to facilitate proper recovery, treatment or disposal. EPA also prohibits hazardous waste dilution in lieu of adequate treatment. In general, dilution does not satisfy the statutory requirement of reducing the toxicity and mobility of hazardous constituents. In some situations, dilution is permissible, e.g. aggregating similar wastes to facilitate treatment, for example, when managing ignitable, corrosive or reactive characteristic wastes, or in Clean Water Act treatment systems.

Properly constructed LDU serve as the second line of groundwater defence and include:

- surface impoundments (natural or manmade depressions used for managing liquid wastes);
- waste piles (open piles used for storing or treating non-liquid waste);
- land treatment units (which utilise the biodegradation properties of soil);
- landfills, the final disposal unit for a significant portion of hazardous waste.

Groundwater monitoring is the final line of defence. LDU operators must monitor underlying aquifers for contamination to ensure that the unit is not leaking. If monitoring results indicate a release, facilities must begin corrective action. GWM programmes must consider a site's hydrology and must include sampling and analysis procedures that ensure consistent results.

A2.5 Overview of landfill and underground waste management facilities and their implementation

A landfill facility is an essential component of most waste management concepts. Despite using all possible ways of avoiding and recycling wastes, there will usually remain wastes that have to be placed in landfills. In practice, it is impossible to guarantee permanent pollution control at an above ground landfill site by man-made barriers. In many cases, the natural barriers are not uniformly structured and the prediction of long-term performance is a difficult task.

Control of environmental impacts in planning, design, operation, evaluation and maintenance of landfills is based on the multiple barrier concept. (Stief, 1987) Applying the multiple barrier concept for landfill sites is the basic means of leaving acceptable landfills for future generations.

The following elements perform the role of barriers:

- the natural properties of the site;
- the bottom lining system;
- the landfill body (the waste);
- the surface liner system (the cap);
- the controlled post-closure use of the landfill area;
- the long term monitoring and control of the landfill behaviour.

Generally, when designing landfills, a worst-case scenario is used regarding discharge of leachate into the ground. To meet the worst-case scenario requirements, the landfill bottom lining system and the surface sealing system, necessary at every type of landfill, often have a composite lining as a sealing element.

The landfill body normally contains as few organic wastes and as little soluble waste as possible. The waste is generally highly compacted to reduce settlement.

Position in Europe: the EU Landfill Directive

The EU Landfill Directive applies to all landfills, which are defined as waste disposal sites for the deposit of waste onto or into land. It defines three main classes:

- landfills for inert waste;
- landfills for non-hazardous waste;
- landfills for hazardous waste.

The Directive's objective is to reduce as far as possible negative effects on the environment and human health by introducing stringent technical requirements for both the waste and the landfills. The Directive sets targets for the reduction of biodegradable waste sent to landfill as 75% of the 1995 level by 2010, 50% of the 1995 level by 2013 and 35% of the 1995 level by 2020.

EU Landfill Regulations have provisions covering location of landfills, and technical and engineering requirements for aspects such as water control and leachate management, protection of soil and water and methane emissions control.

A standard waste acceptance procedure is laid down in the EU Landfill Directive to reduce risks:

- waste must be treated before being put in the landfill;
- hazardous waste within the meaning of the EU Directive must be assigned to a hazardous waste landfill;
- landfills for non-hazardous waste must be used for municipal waste and for non-hazardous waste;
- landfill sites for inert waste must be used only for inert waste;
- criteria must be set for the acceptance of waste at each landfill class.

The EC waste acceptance criteria (WAC) set out the standards that waste must meet to be accepted at one of the three classes of landfill prescribed by the Landfill Directive.

The WAC aim to obtain greater control on the nature of the waste disposed of at landfills, to minimise the impact of this form of disposal. Furthermore, the requirement to characterise all waste disposed of will make the producers more aware of the type of waste they produce, whilst improving the overall knowledge of the constitution of the waste being disposed to landfill.

In general, there are different WACs for the different landfill classes mentioned above. Each WAC might include:

- a list of acceptable wastes which do not have to be tested;
- leaching limit values for a number of contaminants;
- limit values for other parameters.

The following wastes may not be accepted in a landfill:

- liquid waste;
- flammable waste;
- explosive or oxidising waste;
- hospital and other clinical waste which is infectious;
- used tyres, with certain exceptions;
- any other type of waste which does not meet the acceptance criteria.

Sites for permanent underground storage are not subject to the generic hazardous WAC – they rely instead on specific acceptance criteria designed to suit the circumstances of the site.

Figure A2.3 shows the landfill options provided by the EU Landfill Directive. As the WAC criteria provided by the Directive have to be implemented into the national law of each EU Member State, the implementation may vary from country to country.

Landfill disposal in Germany

In Germany, the WAC are implemented through the Landfill Ordinance. The German Landfill Ordinance stipulates the following landfill classes. Each class has acceptance criteria that must be met.

- Class 0 above-ground landfill for inert waste (not contaminated construction waste and excavated soil) (EU class A);
- Class I above-ground landfill for other inert waste (EU classes B1a and B1b);
- Class II above-ground landfill for non-hazardous municipal waste (EU classes B2 and B3);
- Class III above-ground landfill for hazardous waste (EU class C);
- Class IV under-ground landfill other than in salt-rock for hazardous waste (EU class D).

Class IV underground landfills in salt rock must be constructed in accordance with specific requirements and the operator must observe instructions on the maintenance of long-term safety records.

Waste may only be deposited on landfills or landfill sections provided it complies with the acceptance criteria. If necessary, waste must be treated prior to disposal. Hazardous waste may only be deposited provided:

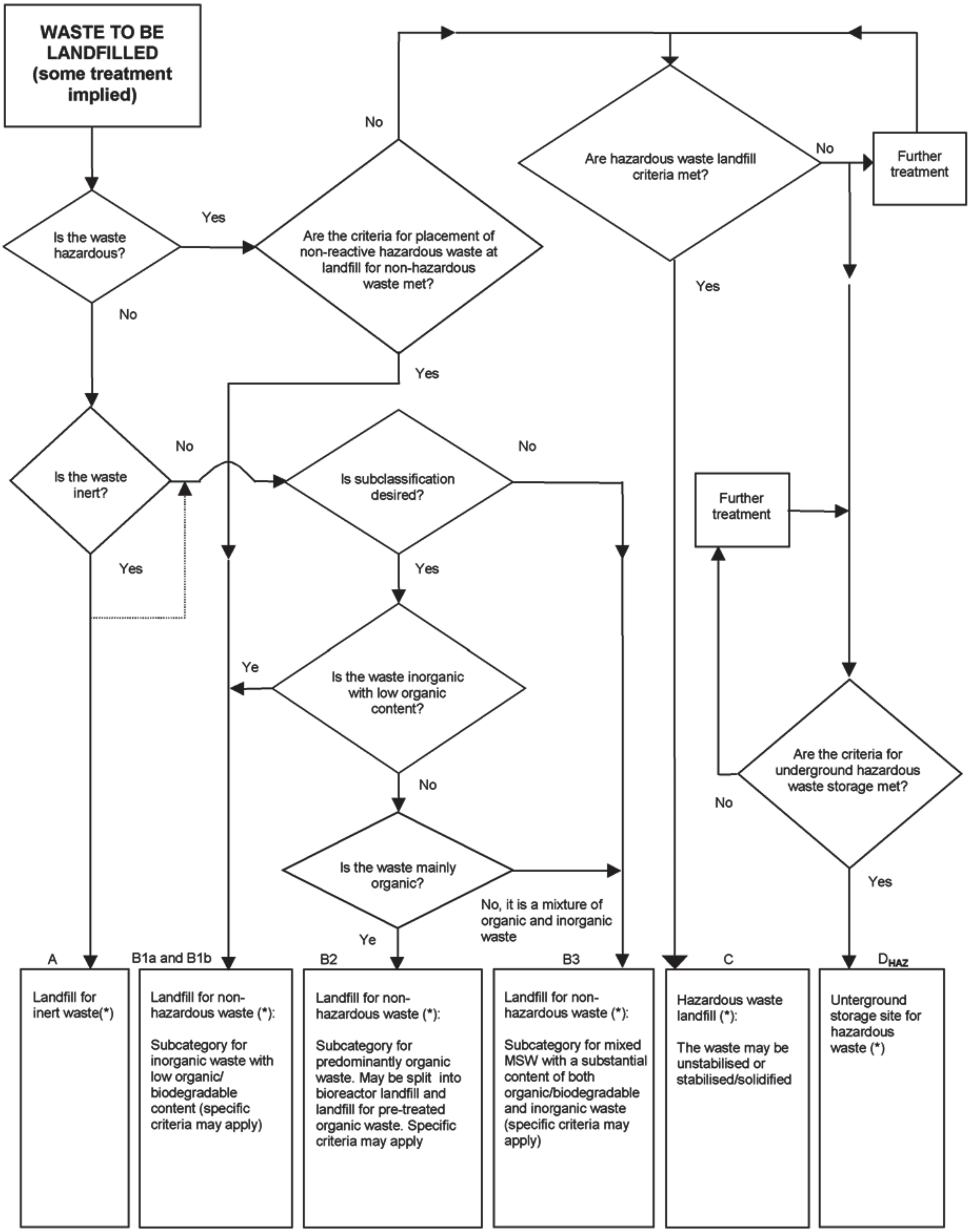
- the landfill or landfill section meets all the requirements for hazardous landfill class III, and the allocation criteria for landfill class III; or
- the landfill meets all the requirements for landfill class IV in salt rock.

The landfill must be secured in such a way as to prevent unauthorised access to the facility.

To ensure permanent protection of the soil and groundwater, above-ground landfills and landfill sections may only be constructed provided the geological barrier and base sealing system at least meet the requirements of Landfill Regulations.

Acceptance criteria for the various classes of landfill site used in Germany are shown in Table A2.2.

Figure A2.3: Landfilling options provided by the EU Landfill Directive



(*) In principle, underground storage is also possible for inert and non-hazardous waste.

Table A2.2: Acceptance criteria for landfill classes I, II, III and IV according to Landfill Ordinance (Germany)

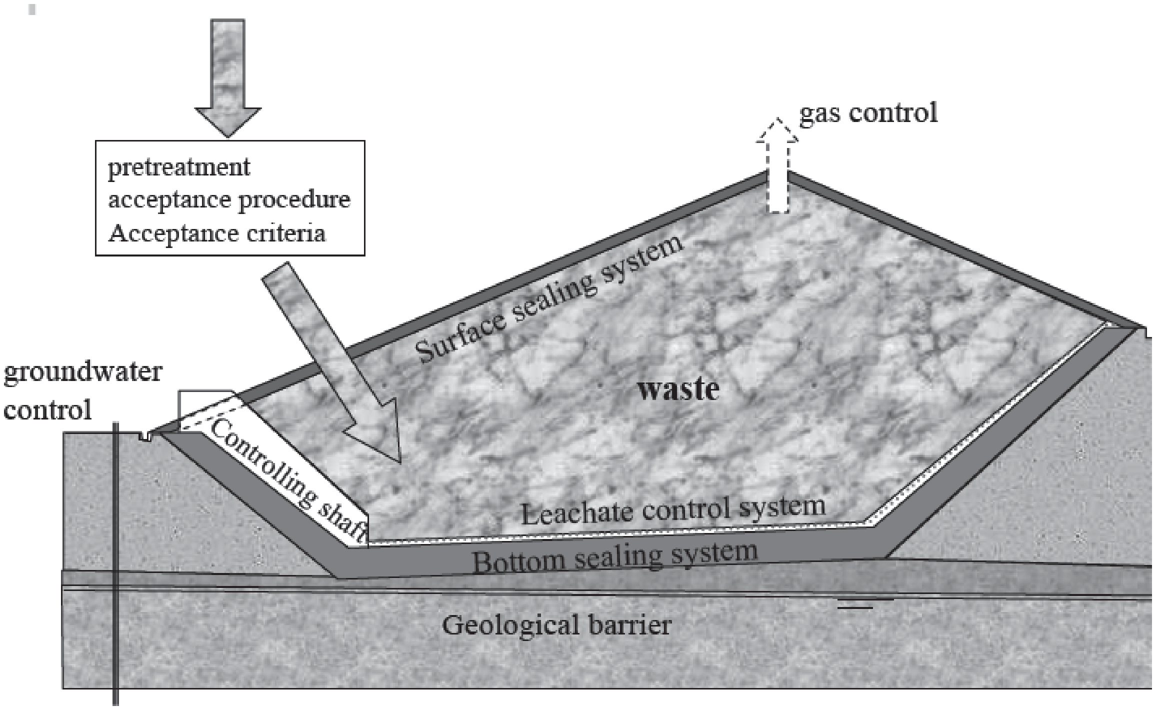
No.	Parameter		Landfill class I	Landfill class II	Landfill class III	Landfill class IV*
1.	**Strength**					
1.01	Vane shear strength	in kN/m²	≥ 25	≥ 25	≥ 25	
1.02	Axial deformation	in %	≤ 20	≤ 20	≤ 20	
1.03	Uniaxial compressive strength	in kN/m²	≥ 50	≥ 50	≥ 50	
2.	**Organic component of dry residue in original subst.**					
2.01	Determined as ignition loss	in % by weight	≤ 3	≤ 5	≤ 10	
2.02	Determined as TOC	in % by weight	≤ 1	≤ 3	≤ 6	
3.	**Other solid criteria**					
3.1	Extractable lipophile substances in original substance	in % by weight	≤ 0.4	≤ 0.8	≤ 4	
	…………					
3.6	Acid neutralisation capacity	in mmol/kg			to be calculated	
4	**Eluate criteria**					
4.01	pH value		5.5-13	5.5-13	4-13.0	5.5-13
4.02	Conductance	in µS/cm	≤ 10 000	≤ 50 000	≤ 100 000	≤ 1 000
4.03	DOC	in mg/l	≤ 50	≤ 80	≤ 100	≤ 5
4.04	Phenols	in mg/l	≤ 0.2	≤ 50	≤ 100	≤ 0.05
4.05	Arsenic	in mg/l	≤ 0.2	≤ 0.2	≤ 2.5	≤ 0.01
4.06	Lead	in mg/l	≤ 0.2	≤ 1	≤ 5	≤ 0.025
4.07	Cadmium	in mg/l	≤ 0.05	≤ 0.1	≤ 0.5	≤ 0.005
4.08	Chromium-VI	in mg/l	≤ 0.05	≤ 0.1	≤ 0.5	≤ 0.008
4.09	Copper	in mg/l	≤ 1	≤ 5	≤ 10	≤ 0.05
4.10	Nickel	in mg/l	≤0.2	≤ 1	≤ 4	≤ 0.05
4.11	Mercury	in mg/l	≤ 0.005	≤ 0.02	≤ 0.2	≤ 0.001
4.12	Zinc	in mg/l	≤ 2	≤ 5	≤ 20	≤ 0.05
4.13	Fluoride	in mg/l	≤ 5	≤ 15	≤ 50	≤ 0.05
4.14	Ammonium-N	in mg/l	≤ 4	≤ 200	≤ 1,000	≤ 1
4.15	Cyanide, easily released	in mg/l	≤ 0.1	≤ 0.5	≤ 1	≤ 0.01
4.16	AOX	in mg/l	≤ 0.3	≤ 1.5	≤ 3	≤ 0.05
4.17	Water-soluble component (evaporation residues)	in % by weight	≤ 3	≤ 6	≤ 10	≤ 1
4.18	Barium	in mg/l	≤ 5	≤ 10	≤ 30	≤ 2
4.19	Chromium, total	in mg/l	≤ 0.3	≤ 1	≤ 7	≤ 0.05
4.20	Molybdenum	in mg/l	≤ 0.3	≤ 1	≤ 3	≤ 0.05
4.21	Antimony	in mg/l	≤ 0.03	≤ 0.07	≤ 0.5	≤ 0.006
4.22	Selenium	in mg/l	≤ 0.03	≤ 0.05	≤ 0.7	≤ 0.01
4.23	Chloride	in mg/l	≤ 1 500	≤ 1 500	≤ 2 500	≤ 80
4.24	Sulphate	in mg/l	≤ 2 000	≤ 2 000	≤ 5 000	≤ 100
5.	**Gross calorific value (H0)**	in kJ/kg			6 000	

* Underground landfill in rock other than salt rock; in salt rock other requirements than limit values are set.

Sources: LO, 2002; WAC, 2006.

Figure A2.4 shows the construction principles for an aboveground landfill site. Different barrier and sealing systems are used for the different landfill classes.

Figure A2.4: Barrier system of an above ground landfill site

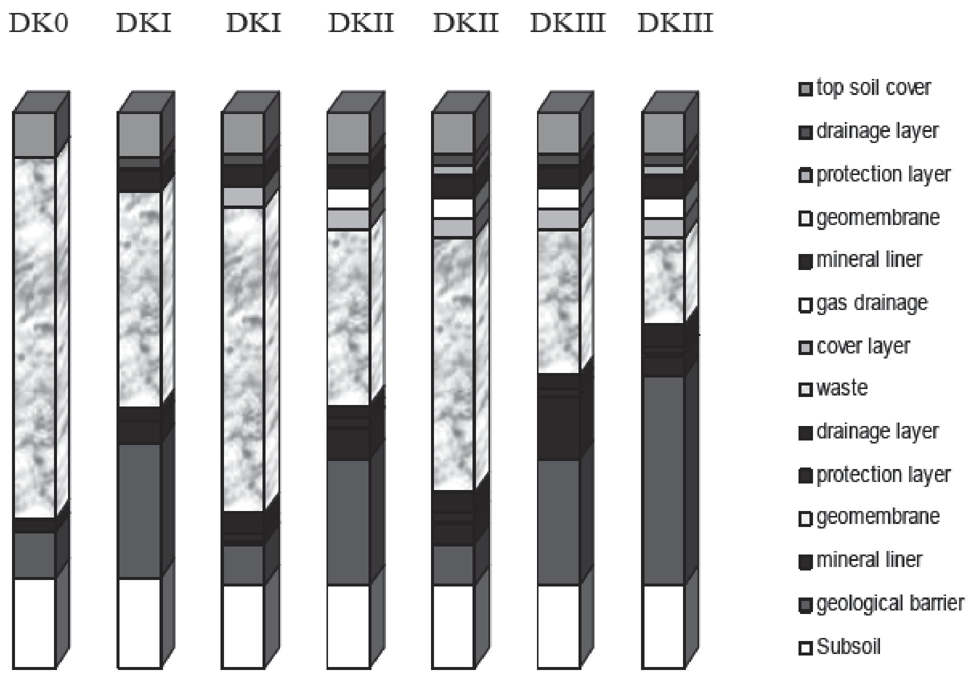


The geological barriers and the surface sealing systems used in Germany are shown in more detail in Figure A2.5. From the environmental protection point of view, each of the barriers should be permanently effective independently of the others. However, the engineered barriers are likely to have limited lifetimes. It is therefore necessary to know when their effectiveness is highest, and what the probable lifetime will be.

According to the EU regulations, every barrier is to be selected and constructed according to BATs.

The post-closure uses of a landfill are also controlled. Within the EU, the Declaration of Landfill Behaviour is published annually, based on regular measurements of emissions and barrier performance.

Figure A2.5: Barrier systems of all classes of landfills (Deponieklassen DK0 to DKIII) in Germany



Notes: DK 0 = landfill class 0, DK I = landfill class I, DK II = landfill class II, DK III = landfill class III; the pictures for the landfill classes I to III represent different possible geological barriers depending on the coefficient of permeability.

Landfill disposal in the United States

In the United States, to minimise the potential for leachate to leak from a landfill, EPA developed the following design standards that are embodied in the RCRA:

- double liner;
- double leachate collection and removal system;
- leak detection system;
- run-on, run-off, and wind dispersal controls;
- construction quality assurance.

To ensure that a landfill meets the design and technological requirements, EPA requires a construction quality assurance programme. The programme mandates a construction quality assurance plan that identifies how construction materials and their installation will be monitored and tested and how the results will be documented. The programme must be developed and implemented under the direction of a registered professional engineer, who must also certify that the construction quality assurance plan has been successfully carried out and that the unit meets all specifications before any waste is placed into the unit.

In the United States, closure of a landfill triggers post-closure care. These facilities must obtain a permit or enforceable document for post-closure care. Post-closure requirements last for at least 30 years, unless the permitting authority approves a shorter period. Post-closure care requires groundwater monitoring and facility maintenance. EPA requires the submission of specific information for post-closure permits.

TSDFs must demonstrate financial assurance for closure and if applicable, post closure of the facility. In addition, TSDFs must also demonstrate the financial assurance to provide for liability coverage for accidental occurrences. EPA can also require financial assurance for corrective action activities, where such activities must be performed.

Closure coverage includes funding needed to conduct closure and post-closure. Closure cost is facility-specific so the facility must prepare a cost estimate (but it must be based on cost to hire a third party to close the facility). TSDFs must update their cost estimates annually to adjust for inflation and revise them if the facility expands and increases the cost of closure. TSDFs must also maintain liability coverage until the permitting authority receives certification of final closure and notifies the facility that it is released from this obligation.

A2.5.1 Underground waste disposal

Hazardous wastes that need to be isolated from the biosphere can be disposed in underground disposal facilities. Detailed knowledge of the rock properties, the characteristics of the specific wastes and of any adjacent mining operation are needed to ensure the safety of the disposal facility operators. Waste pre-treatment and packaging are needed.

In contrast to above ground landfills, the natural barrier systems in underground disposal facilities play a more significant role in keeping hazardous substances from leaking into the biosphere. In addition, underground waste disposal leaves above ground space available for uses that are more appropriate.

Underground disposal in rock salt

There is experience in France, Germany, the United Kingdom and the United States of disposing hazardous wastes in salt rock formations.

Germany

In Germany, there is experience from placing toxic, water-soluble and environmentally hazardous wastes in intact and compact rock salt deposits. The wastes deposited are encapsulated in the rock salt mass and due to the favourable hydrological properties of the rock salt, the wastes are not subjected to the dissolution and subsequent transport processes that occur if disposed of in other types of rock media or above ground.

A practice for many years in Germany has been the use of suitable wastes in stabilising cavities during or after rock salt mining operation. To prevent environmental impacts and ensure long-term safety, wastes must fulfil certain requirements on mechanical stability.

At present, there are 14 aboveground landfill sites for hazardous wastes, four underground landfill sites in rock salt and more than 20 backfilling facilities in operation in Germany.

France

The former French underground disposal facility called Stocamine, located in Wittelsheim, Alsaçe was set up in abandoned parts of an old potassium mine at 600-700 m depth. The first waste was disposed of in February 1999 and waste disposal was planned to continue beyond the termination of mining activities in 2004. However, in September 2002 there was a severe fire in the facility at a depth of 600 m. The resultant gases and fumes contaminated the landfill as well as the mine and both the disposal site and the mine were rendered unusable.

United Kingdom

The British underground disposal site called Minosus, located near Winsford in Cheshire, has been in operation since 2005. The landfill is situated at 170 m depth and is able to accept 42 different categories of waste included in the European Waste List. A further 24 potential waste categories are permissible but are subject to Environment Agency improvement orders. PCB containing wastes are not accepted.

United States

The underground disposal of solid waste (hazardous or any other kind) is relatively uncommon in the United States. However, the deep well injection of liquid hazardous waste (which is not in the scope of this report), while conducted by only 3% of hazardous waste facilities, does account for almost 50% of all hazardous waste managed. (EPA, 2006b) The fact that most abandoned or closed mines are located comparatively far from population and industrial centres, and the economic advantages of using engineered landfills rather than excavating subsurface facilities, have worked against sub-surface disposal. There are however a few examples of the use of salt caverns for the disposal of special wastes, dating back over half a century.

The best-known example of salt disposal of hazardous waste is the Waste Isolation Pilot Plant (WIPP), the world's only deep geologic underground repository for the disposal of transuranic waste with negligible heat load that originated from military sources. WIPP is located in south-eastern New Mexico, 26 miles east of Carlsbad. Though constructed as a repository for radioactive waste, WIPP is required to meet all RCRA requirements as a hazardous waste landfill. This is necessary because the waste contains hazardous constituents subject to regulation under RCRA. Generally, the transuranic waste consists of clothing, tools, rags, residues, debris, soil and other items contaminated with radioactive elements.

The salt formations at the WIPP location were formed over 250 million years ago by evaporation of an ancient ocean. The salt formations begin at approximately 260 m beneath the surface and extend to over 870 m depth. Waste at WIPP is emplaced in disposal rooms at 655 m beneath the surface. The total volume of waste anticipated at WIPP is 175 570 m^3 and disposal operations are scheduled to continue until 2035.

Other than WIPP, salt caverns have been examined for the disposal of hazardous waste for some time. Although there are very few examples in operation the technology to construct them is well understood through development as hydrocarbon storage facilities. Salt caverns can be easily created by drilling into a salt formation, injecting water to dissolve the salt, and removing the brine. The storage of liquids and gases in solution-mined salt caverns was used in Canada during the Second World War. By the 1950s, storage of liquefied natural gas, petroleum, and light petroleum hydrocarbons was widespread in Europe and North America. Natural gas was first stored in an excavated salt cavern in Pennsylvania in 1961. Salt caverns have also been used for storage of compressed air, hydrogen, helium and anhydrous ammonia.

Costs

The unit cost for treatment of hazardous waste is highly dependent on the substances and materials involved and the many different ways of potential treatment. Examples from Sweden and Finland give some order of magnitude of the costs involved. Costs for incineration range between 80 and 500 /tonne for different types of hazardous waste, averaging 270 /tonne and 300 /tonne respectively. For management of highly toxic waste, like incineration of PCBs, the cost is in the order of 1 000 /tonne. Some rough data from Germany indicates that the unit costs for treatment of hazardous waste are:

- underground waste disposal in salt rock ~250 /tonne;
- hazardous waste incineration 250-1 000 /tonne;
- chemical physical treatment ~110 /tonne.

A2.6 Legal and organisational infrastructure

As described above, with the EU Landfill Directive (WLD, 1999), the European Union has laid down strict requirements for waste and landfills to prevent and reduce as far as possible any negative effects on the environment in particular on surface water, groundwater, soil, air and human health.

The EU Landfill Directive sets up a system of operating permits for landfill sites. Applications for permits must contain the following information:

- the identity of the applicant and, in some cases, of the operator;
- a description of the types and total quantity of waste to be deposited;
- the capacity of the disposal site;
- a description of the site;
- the proposed methods for pollution prevention and abatement;
- the proposed operation, monitoring and control plan;
- the plan for closure and aftercare procedures;
- the applicant’s financial security;
- an environmental impact assessment study, where required under Council Directive 85/337/EEC on the assessment of the effects of certain public and private projects on the environment.

Germany

In Germany, the Ordinance on Landfills and Long-Term Storage Facilities defines underground landfills as “Class IV landfill”. These underground facilities must be completely encased in rock, in a mine with disposal areas that are created separately from mineral extraction. Hazardous wastes may only be disposed provided the facility meets all the requirements for landfill class IV in salt rock.

The following wastes may not be disposed in a landfill of class IV constructed in salt rock:

- liquid wastes;
- infectious wastes, body parts and organs;
- unidentified or new chemical waste from research, development and education activities, the effects of which on humans and the environment are not known;
- whole or shredded used tyres;

- waste leading to significant olfactory nuisances for those employed at the landfill site and for the neighbourhood;
- wastes classified as explosive, highly flammable or readily flammable;
- wastes which, under disposal conditions, may lead to:
  - increases in volume;
  - the formation of self-igniting, toxic or explosive substances or gases; or
  - to other hazardous reactions by reacting with one another or with the rock, if this would cast doubt on the operational reliability and integrity of the barriers.

The acceptance procedures of the Landfill Ordinance include checks to verify that the waste delivered is consistent with the waste declared.

United Kingdom

The Acceptance procedures for Minosus in the United Kingdom include specific testing to be carried out during the waste characterisation phase. The tests determine the wastes ability to react under mine storage conditions and the risk of production of toxic and/or flammable gases. Due to the extended pre-acceptance testing of waste, verification testing is minimal. Currently, only alkaline waste has been deposited in the Minosus site. Waste has generally been derived from thermal processes, e.g. air pollution control residues from incinerators.

United States

In the United States, the legal structure for addressing hazardous waste derives primarily from RCRA, as discussed in earlier sections. RCRA includes a Congressional mandate directing EPA to develop and issue a comprehensive set of regulations that translate the general mandate of a statute into a set of requirements, addressing items such as standards, permitting, enforcement, public participation, etc. Implementation of the RCRA requirements may be performed by EPA, but implementation may be delegated to individual states provided that stringency and consistency with the current federal requirements are met.

A2.6.1 National, regional and local level responsibilities

Virtually every level of government administration and nearly every authority is involved in environmental protection and waste management in one way or another. In most OECD countries, shared responsibility exists between the national, regional and local levels.

Germany

In the case of Germany, the Constitution governs the distribution of these tasks among Federal, Regional and Local Government.[8] The German Federal Government exercised its power to implement

8. In the case of concurrent legislation, the Federal Government of Germany has the power to legislate, provided there is a need for legislative provisions at a national level. Should the Federal Government choose to exercise this power, Federal law will override the law of the Federal States. The Federal States are involved in Federal legislation via the Federal Council (Bundesrat). Waste management is likewise subject to concurrent legislation at Federal level.

EC Directives, and stated the basic obligations concerning waste management, with the Recycling Management and Waste Act (RMWA, 1996) and some subsequent ordinances.

The implementation of statutory and administrative provisions in the waste sector, i.e. the enforcement of these provisions, is the sole responsibility of the Federal States. For example, the Federal States are exclusively responsible for supervising waste management, licensing waste disposal facilities, organising the management of hazardous wastes, and preparing waste management plans.

The German RMWA primarily obligates the producers of waste to take responsibility for the avoidance, recycling or disposal of waste.

EU

The European Community stresses the particular significance of landfill-specific requirements. Within the context of Community environmental policy, great importance is attached to the provision of a high-quality waste management infrastructure with harmonised environmental requirements. The Directives adopted by the Council in the late Eighties and early Nineties regarding the incineration of municipal and hazardous wastes, together with the EU Landfill Directive which came into force in July 1999, form the cornerstones of European waste management provisions.

United States

In the United States, the EPA is obligated to delegate authority to operate many federal environmental programmes to the states who meet the qualifications. For states to receive authorisation from EPA to implement the RCRA hazardous waste programme in lieu of the Federal government, states must maintain standards that are equivalent to and at least as stringent as the federal programme. Implementation of the authorised programme usually includes activities such as permitting, corrective action, inspections, monitoring and enforcement. Currently 48 of the 50 States have authorised hazardous waste programmes with only Alaska and Iowa not authorised.

A2.6.2 Transboundary shipments of waste

Transboundary shipment of waste is regulated by the UN via the Basel Convention and implemented by the EU in the Waste Shipment Regulation. The Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal, which came into force in 1992 with the primary objective of restricting shipments of hazardous wastes to developing countries, contains the first outlines of a global "waste management convention". It includes the principle of waste disposal at the site of generation, giving priority to measures aimed at reducing the volumes of waste and the task of formulating general principles for an environmentally sound system of waste disposal that is applicable globally.

Within the context of the Basel Convention, a system for notifying, identifying and control of transfrontier shipments of wastes for recovery was established for the OECD countries. In the EU, transfrontier shipments of waste are regulated by EC Regulation on shipments of waste (EU-WSR), which implements the Basel Convention and the OECD control system for transboundary movements of waste.

Under the EU-WSR, a planned transboundary shipment of waste must either be accompanied by certain information or have prior written notification and consent depending on the intended recovery or disposal method, the country of destination and the classification of the waste. (WSR, 2006) These requirements are set out in Table A2.3.

Table A2.3: Simplified overview of permissible transfrontier waste shipments under the EU-WSR

	Between EU member states	Import into the EU	Transit through the EU	Export out of the EU
Waste for disposal	Consent required	Consent required	Consent required	Prohibited[1]
"Green wastes" for recovery that do not contain any hazardous components	Information requirements[2]	Information requirements	Information requirements	Information requirements or special provisions[3]
All other waste	Consent required	Consent required	Consent required	Prohibited[4]

1. Export to Iceland, Lichtenstein, Norway and Switzerland is permitted with prior written notification and consent.
2. Transitional arrangements still apply to some new EU Member States. Export to Bulgaria requires written consent until the end of 2014, to Latvia until the end of 2010, to Poland until the end of 2012, to Romania until the end of 2015 and to Slovakia until the end of 2011.
3. Further restrictions by the national law of the non-EU country in question may exist.
4. The export of hazardous wastes for recovery to countries to which the OECD Decision does not apply is prohibited.

The procedure for prior written notification and consent requires checks before the beginning of the waste shipment and verification of the waste's destination. The exporter must notify the shipment to the competent authorities of the exporting, importing and transit countries.

Transboundary shipments of waste are only allowed when the competent authority in the country of dispatch, the competent authority in the country of destination and the competent authorities of transit countries have all consented. The competent authorities of dispatch and destination have to consent in writing, whereas the competent authority of transit may choose tacit consent. All consents have to be in place together and are valid for one year.

Within the EU, hazardous wastes are crossing borders on a regularly basis, because specialised waste treatment and disposal facilities are not available in every Member State. Increasing amounts of waste are being shipped within the EU.

A2.7 Safety

Isolation of wastes from the biosphere is the ultimate objective for the final disposal of wastes in underground facilities. The wastes, the geological barrier and the cavities, including any engineered structures, constitute a system that together must meet the safety requirements.

Safety considerations for underground waste disposal include evaluation of the disposal location, as well as assessment of the waste to be disposed of. The properties of the wastes must be compatible with the properties of the underground facility to prevent any contact of the waste with the biosphere for extremely long periods.

The requirements for groundwater protection can be fulfilled only by demonstrating the long-term safety of the installation. Experience of storing hazardous wastes in underground disposal facilities only exists in a limited number of countries.

EU-Requirements on the location and the waste are described in the EU Landfill Directive (WLD, 1999) and the Council Decision establishing criteria and procedures for the acceptance of waste at landfills. (WAC, 2003) The Council Decision in its chapter "Safety Philosophy for Underground Storage" points out the importance of the geological barrier for the long-term isolation of the wastes from the biosphere being "the ultimate objective for the final disposal of wastes in underground storage".

The assessment of risk requires identification of:

- the hazard (the deposited wastes);
- the receptors (the biosphere including groundwater);
- the pathways by which substances from the wastes may reach the biosphere;
- the impact of substances that may reach the biosphere.

Acceptance criteria for underground storage must be derived from, *inter alia*, analysis of the host rock, to confirm that no adverse site-related conditions are present. The acceptance criteria for underground storage can be obtained only by referring to the local conditions. This requires a demonstration of the suitability of the strata for disposal, i.e. an assessment of the risks to containment, taking into account the overall system of the waste, engineered structures and cavities and the host rock body. The site specific risk assessment of the installation must be carried out for both the operational and post-operational phases. From these assessments, the necessary control and safety measures can be derived and the acceptance criteria can be developed.

An integrated performance assessment analysis must be prepared, including the following components:

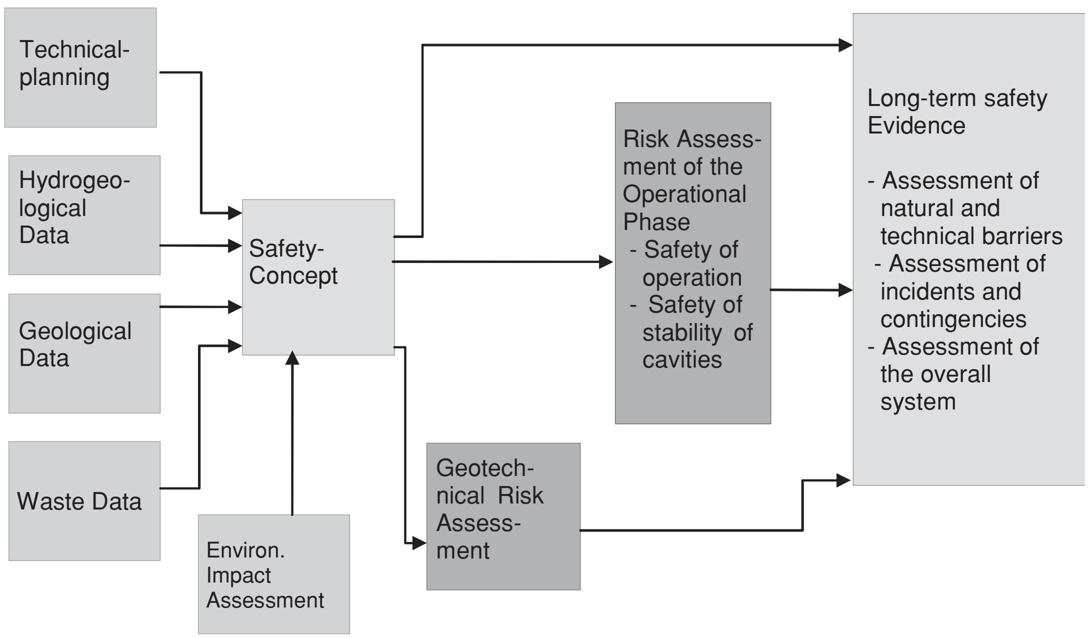
- geological assessment;
- geomechanical assessment;
- hydrogeological assessment;
- geochemical assessment;
- biosphere impact assessment;
- assessment of the operational phase;
- long-term assessment;
- assessment of the impact of all the surface facilities at the site.

Containers and cavity lining should not be taken into account when assessing the long-term risks of waste disposal because of their limited lifetime. Wastes that may undergo undesirable physical, chemical or biological transformation after they have been emplaced must not be disposed of in underground storage. The EU Council Decision on Waste Acceptance Criteria and the EU Landfill Directive both specifically exclude wastes from underground disposal that are biodegradable, explosive or auto-flammable and wastes that can generate a gas-air mixture that is toxic or explosive. However, limit concentrations for hazardous substances contained in the wastes are not required because of their almost complete and permanent isolation from the biosphere.

A2.7.1 Safety approach

In all OECD countries, the basis for issuing the necessary operating license for an underground disposal facility is the generation of a long-term safety analysis. Such an analysis normally includes a site-specific safety evaluation that demonstrates that the setting-up, operation and post-operational maintenance of the underground waste disposal plant does not lead to any harm to human health and the environment. The process is set out in Figure A2.6. In general, a geological assessment demonstrates the suitability of the site for underground storage. The location, frequency and structure of any faulting or fracturing in surrounding geological strata and the potential impact of seismic activity on these structures must be included. Alternative site locations should be considered.

Figure A2.6: Site specific safety assessment



In Germany, the stability of rock salt cavities must be demonstrated by appropriate investigations and assessments. The disposed waste must be part of this assessment. The processes must be analysed and documented in a systematic way.

The following should be demonstrated:

- that during and after the formation of the cavities, no major deformation is to be expected either in the cavity itself or at the earth surface which could impair the operability of the underground storage or provide a pathway to the biosphere;
- that the load-bearing capacity of the cavity is sufficient to prevent its collapse during operation;
- that the deposited material has the necessary stability compatible with the geo-mechanical properties of the host rock.

A thorough investigation of the hydraulic properties is required to assess the groundwater flow pattern in the surrounding strata based on information on the hydraulic conductivity of the rock mass, fractures and the hydraulic gradients. A thorough investigation of the rock and the groundwater composition is required to assess the current groundwater composition and its potential evolution over time, the nature and abundance of fracture filling minerals, and to provide a quantitative mineralogical description of the host rock. The impact of variability on the geochemical system should be assessed.

An investigation of the biosphere that could be impacted by underground disposal is required. Baseline studies must be performed to define local natural background levels of relevant substances.

For the operational phase, the analysis must demonstrate the following:

- the stability of the cavities;
- that no unacceptable risk of a pathway will develop between the wastes and the biosphere;
- that no unacceptable risks affect the operation of the facility.

When demonstrating operational safety, a systematic analysis of the operation of the facility must be made based on specific data on the waste inventory, facility management and the method of operation. It must be shown that the waste will not react with the rock in any chemical or physical way, which could impair the strength and tightness of the rock and endanger the disposal facility itself. For these reasons, wastes that are liable to spontaneous combustion under the storage conditions (temperature, humidity), gaseous products, volatile wastes and wastes that are collections of unidentified mixtures should not be accepted.

Particular incidents that might lead to the development of a pathway between the wastes and the biosphere in the operational phase should be identified. The different types of potential operational risks should be summarised in specific categories and their possible effects evaluated. It should be shown that there is no unacceptable risk that the disposal facility containment will be breached. Contingency measures must be provided.

A2.7.2 Containment, isolation and multiple barriers concept

Example from Germany

To comply with general objectives of sustainable landfilling, risk assessments must cover the long-term. In the case of underground facilities in Germany, long-term is taken to mean 10 000 to 50 000 years. It must be demonstrated that no pathways to the biosphere will be generated during the long-term post-operation of the underground storage, that the wastes are sealed by a multiple barrier system consisting of natural/geological as well as of artificial/technical barriers. The safety assessment normally comprises a description of the initial status at a specified time (e.g. the time of closure) followed by a scenario outlining important changes that are expected over geological time. Finally, the consequences of the release of relevant substances from the underground storage are assessed for different scenarios reflecting the possible long-term evolution of the biosphere, geosphere and the underground disposal facility. Some R&D is usually required.

The barriers of the underground disposal site (e.g. the waste quality, engineered structures, back filling and sealing of shafts and drillings), the performance of the host rock, the surrounding strata and the overburden are quantitatively assessed over the long-term and evaluated on the basis of site-specific data or using conservative assumptions. The geochemical and geohydrological conditions such as groundwater flow, barrier efficiency, natural attenuation as well as leaching of the deposited wastes are all taken into consideration.

In Germany the experience is that salt mine caverns offer the safest, as well as the environmentally most responsible, solution for the disposal of hazardous wastes. The surrounding rock salt mass is a perfect seal against liquids and gases. The layers surrounding the rock salt mass and the covering layers reliably seal the rock salt layer against any intruding moisture. The storage areas of an underground waste disposal plant are positioned lower than any groundwater reservoirs.

The geological conditions, which have remained stable for more than 200 million years, and which have guaranteed an intact rock salt layer, also guarantee reliable conditions for the future, particularly in reference to the protection of the biosphere. The rock salt as host rock simultaneously assumes the sole function of the barrier rock. For this reason, long-term safety records should be kept for the salt rock as barrier rock. Where available, further geological barriers could afford additional protection, but these are not compulsory. However, in addition to natural barriers, artificial barriers are used. For example, the entrances to the separate storage chambers are closed by dry brick walls or by rock salt fillings.

Should on-going mining operations and storage be conducted concurrently within a larger mining field, the disposal area is sealed from the extraction activity by a salt layer of an appropriate thickness. All connecting links and ducts between the waste disposal and the operating mine are sealed.

The artificial/technical barriers, such as packaging the wastes in containers, closing the storage chambers between each other and/or against the concurring mining operations primarily serve the safety of the operating phase of the underground waste disposal plant.

When underground disposal ceases, the shafts, as the sole connections between the storage chambers and the environment, will be sealed by appropriate solid materials, and a hydraulically secure closure of the mine will be undertaken. Filling the shafts is the final and most important barrier, as it blocks the only connection to the wastes underground, thereby ensuring that the stored waste is reliably sealed from the biosphere.

A2.7.3 Safety case and safety assessment

The aim of the safety case is to demonstrate that the development and operation of the underground waste disposal facility and, in particular, the phase after its closure do not cause an unacceptable level of harm to the biosphere. The term biosphere is far reaching; in particular, it includes groundwater.

Example from Germany

When disposing of wastes in an underground disposal facility in salt rock, the objective is the complete and permanent sealing of the waste from the biosphere. The requirements relating to the wastes, the mine chambers, the geotechnical barriers (sealing structures) and all other technical equipment and operational measures are based on this objective. Salt, as the host rock, must meet the requirements of being gas and liquid-impermeable, of gradually enclosing the waste by its convergence behaviour, and at the end of the deformation process, of encapsulating it completely.

The convergence behaviour of salt rock is consistent with the requirement that the caverns must be stable during the operational phase of the landfill, provided it causes only fracture-free deformations and does not open up any water migration pathways. The requirements relating to stability are intended, firstly, to ensure operational safety, and secondly, to preserve the integrity of the geological barrier so that the protective effect against the biosphere is maintained.

The salt barrier rock must have an adequate spatial spread and, in the selected emplacement area, an adequate thickness. The existing salt thickness must be sufficiently large that the barrier function is not impaired in the long term.

A methodology to determine long-term safety record, for normal and fault conditions including physical modelling and numeric simulation is available. (Lux, 2008)

A2.8 Development of landfill and geological disposal facilities

This section discusses landfill and geologic disposal mainly drawing on experience from Germany.

The following matters are among those that should be considered when setting up and operating an underground hazardous waste disposal plant:

- geological characteristics of host rock formations suitable for hazardous waste disposal;
- facility design and construction;
- waste acceptance criteria;
- operation;
- closure of the facility;
- monitoring programmes and need for post-closure and institutional controls;
- environmentally sound management.

In general, salt mines are used for hazardous waste disposal in Germany. Among the important factors for consideration when using an existing mine are:

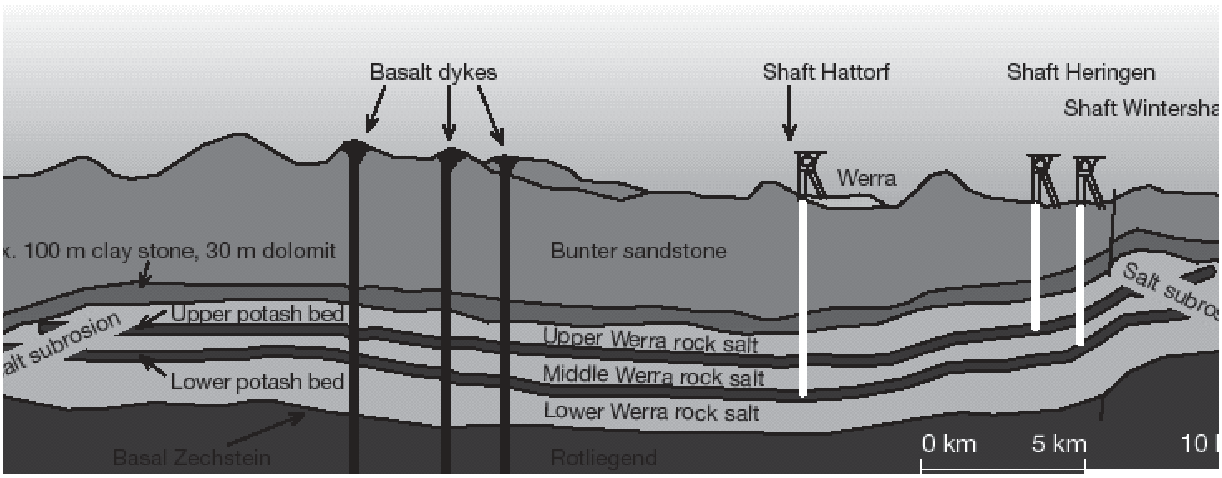
- Preferably, an exhausted mine needs to be available.
- If mining is still carried out, wastes must be disposed only in areas securely separated from mining activities.
- The cavities need to be solid so that there is no possibility of caving in during operation.
- The facility in which wastes are to be disposed must be sufficiently dry.
- The geological conditions at the site must allow the wastes to be adequately sealed against the biosphere.
- For the post closure phase, the site should not require any post-operational maintenance.

A2.8.1 Geological characteristics of salt rock formations suitable for hazardous waste disposal

The geological situation at the site is of crucial importance. The dimension of the host rock needs to be large enough for the intended disposal, and needs to be thick enough to provide a long-term barrier. As an example, the geological situation at the site of the underground waste disposal plant Herfa-Neurode in Germany is described below (Figure A2.7).

The mining field surrounding the underground waste disposal facility is part of the Werra basin salt deposit, encompassing 1 100 km² in Thuringia and Hesse; it is currently mined for potash salts. The salt deposit is flat, with a thickness of about 300 m. It was deposited during the Zechstein-age approximately 240 million years ago, and consists mainly of rock salt.

Figure A2.7: Geological cross-section example of disposal in salt mine at the Werra (Germany)



Imbedded in these large salt masses are two potash salt deposits, with a thickness of approximately 2.5-3 m. Both of these deposits are separated by about 60 m of medium Werra rock salt. The rock salt mass is covered by layers of clay and dolomite, which is again buried under 300-600 m of new red sandstone. Four of these clay layers, together approximately 100 m thick, seal the rock salt mass against the water-bearing new red sandstone.

These clay layers are pliable and waterproof. During previous movements within the earth's crust (for example during the folding of the Thüringen Forest) they maintained their sealing qualities. They provide reliable and enduring protection for the rock salt deposit below. About 20 million years ago, during the Miocene Period, the deposit was permeated by basalt dykes and pipes. Also created during this time were the typical basalt hilltops of the Rhön Mountains. Even though thermal and tectonic stress was extreme, the salt layers remained nearly unchanged.

During this time, carbon dioxide penetrated the salt. This gas, liquefied by the exceedingly high pressure, still exists in the salt layers today. The compact salt layer is so dense that even during the millions of years that followed; the pressurised gas has not been able to escape. This demonstrates that the underground salt deposit is well sealed.

These beneficial geological conditions were the primary reason for the decision to operate an underground hazardous waste disposal facility at this site.

The geology of the Minosus site in the United Kingdom comprises Cheshire salt beds and associated sedimentary rock sequence laid down approximately 200 million years ago during the Triassic period. As a result, the Mercia Mudstone (Keuper Marl) Group underlies the whole of the mine. The mine is situated on a faulted block of the Northwich Halite Formation, which is bounded to the west and east by two major faults trending in a NNW direction. At the surface, a varied thickness of Quaternay sands, gravels and boulder clay are present to a thickness up to about 60 m. Beneath the superficial cover of glacial silts and sands, a solution of salt within the middle mudstone creates "wet" rock-head conditions.

Within the Northwich Halite Formation, halite is the dominant mineral with silt inclusions also occurring as beds up to 10 m thick. Both the current mine workings and the waste disposal facility are in a zone near the base of the formation. The Mercia Mudstone group is considered to be an aquitard (a mineral which restricts the flow of groundwater), and where groundwater is present, it is generally highly mineralised.

A2.8.2 Facility design and construction

The first underground landfill facility for hazardous waste in Germany was established 1972 in Herfa-Neurode. Extraction of potash salt uses the "room and pillar" system. This system entails the construction of right-angled tunnels, leaving rectangular or square pillars supporting the overlying rock mass. These are sized to safeguard permanent stability of the cavities. Before the cavities can be used to deposit waste, they are again secured by mechanically clearing out any loose rocks from shaft walls and the deployment of rock anchors. This assures their stability after the operational phase. After the ceilings have been secured, the area for waste disposal is made accessible by roadways; during the disposal phase, lorries and forklifts can be used.

The ventilation of the disposal area will be independent of the ventilation of the active mining area.

A2.8.3 Waste acceptance criteria

In Germany, the composition, leachability, long-term behaviour and general properties of a waste to be disposed must be known sufficiently accurately to demonstrate compliance with the acceptance criteria. Waste acceptance at a facility can be based either on lists of acceptable waste, defined by nature and origin, or on waste analysis methods and limit values for the properties of the waste to be accepted.

Packaging for each particular type of waste is individually determined depending on the waste's characteristics. The packaging must withstand mechanical stresses, and must be resistant to corrosion caused by the material it contains. General criteria for the selection of packaging material are:

- toxicity;
- pH-value of the waste;
- moisture content of waste;
- particulate matter content (particularly relevant to worker safety during delivery and acceptance control).

In Germany stainless or carbon steel containers with plastic liners are typically used as packaging materials in underground waste disposal facilities.

Wastes are separated into single material groups and distributed through the disposal facility to ensure that different waste types do not react with each other. Even though all wastes are delivered and stored in sealed containers, and immediate contact is thereby excluded, they are distributed into separate storage areas, which are sealed from each other to avoid the spread of fire. Grouping wastes together also allows provision of appropriate fire extinguishing systems.

Wastes may derive from the following industries in Germany:

- incineration of municipal and hazardous wastes;
- smelters;

- metal processing industry;
- chemical industry;
- pharmacological industry;
- electrical industry;
- glass production;
- cleanup operations; and
- waste treatment facilities.

Examples of German types of waste are:

- fly ash from the incineration of municipal and hazardous wastes;
- waste from electro-plating;
- wastes from hardening salts;
- wastes from chemical distillation;
- wastes containing mercury;
- wastes containing PCB;
- wastes from fluorescent lamps;
- filtration and sewage filter wastes; and
- contaminated soils and building rubble.

For final disposal in rock salt, there is no requirement to define concentration limits for hazardous substances because of the long-term isolation of the waste from the biosphere.

A2.8.4 Operation

Monitoring is essential to ensure that hazardous waste is disposed through the most suitable waste management processes. To this end, all OECD countries rely on monitoring systems. Most monitoring systems monitor hazardous waste from cradle to grave.[9]

Transport of the wastes to a waste disposal plant is normally done by trucks or by rail. The vehicles initially stop at the entrance area to the waste disposal plant. A typical entrance area encompasses storage space for the delivery vessels, a scale and an office including an in-house laboratory. The entire compound is generally leakage-proof and at places may be fitted with separate collection systems. The entrance area also may include facilities for taking samples from waste deliveries, as well as for carrying out acceptance and identification controls.

Although the wastes taken at the site may be destined for subsurface disposal, wastes may also be unloaded, tested and possibly stored on the surface, before reaching their final destination. The reception facilities are typically designed and operated in a manner that will prevent harm to human

9. A typical example is the Ordinance on Waste Recovery and Disposal Records in Germany (OWRDR, 2006).

health and the local environment. They must fulfil the same requirements as any other waste reception facility.

The tasks covered by the entrance inspection and the acceptance control at an Underground Waste Disposal Facility typically include the following:

- control of the waste documents/chain of custody records and the accompanying documents;
- comparison of the information given in the waste documents/chain of custody records with those included in the record of disposal;
- quantity, or mass-determination; and
- identity control.

The identity control includes visual inspection and taking samples for retention and identification analysis. Before the vehicles reach the entrance area, they have already passed through a radioactivity measurement control system.

Before the waste containers are opened for visual inspection and sample extraction, an exhaust system is used to test for explosive gas/air mixtures. The ullage space within each container is normally inspected by insertion of a testing probe. The opening caused by these measures is sealed after the procedure.

After conduct of the acceptance controls and determination that the control results agree with the information provided by the disposal record, the waste is cleared for disposal.

A2.8.5 Closure of the facility

Based on German national regulations the landfill might be closed:

- if the relevant conditions stated in the permit are met;
- at the request of the operator, under the authorisation of the competent authority;
- by the reasonable decision of the competent authority.

As an example, in Germany the operator of a landfill has to prepare an inventory plan within six months of the end of the disposal phase of the landfill or landfill section, and must submit this to the competent authority. In particular, the inventory plan must include the declarations on landfill behaviour, as well as the technical measures implemented in the case of above ground landfills or landfill sections.

During the closure phase, the operator must promptly carry out all the measures required to prevent future adverse impacts from the landfill. Measures include the construction of a surface sealing system in the case of above-ground landfills or landfill sections.

If major subsidence of an above-ground landfill is anticipated, a cover may be provided prior to applying the final surface sealing system, until the main subsidence has abated. This temporary surface cover is intended to minimise the formation of leachate and prevent landfill gas migration.

A2.8.6 Monitoring programmes and need for post-closure and institutional controls

At underground disposal facilities, all information pertaining to the disposal time and waste location is typically recorded in detail. Documentation may include a mine map, containing all the information on the types of wastes disposed, as well as on the walls and barriers created. Typically, this makes it possible to locate any particular waste at any time. Normally this also makes it possible to retrieve the waste. Removal of wastes has repeatedly been done in the past and is being done on an even greater scale today, in order to recycle components contained in the waste and to feed them back into the economic cycle.

The most important criterion for worker safety in an underground waste disposal plant is the monitoring of the ventilation system, particularly for hazardous particles. This monitoring is carried out by gas detection instruments, by internal measurements at the separate workstations, but also by external auditing agencies. Additionally, there are stationary gas and fire detectors, which are permanently integrated into online reporting chains.

In addition to self-regulation, there may be inspections by external experts as well as by the relevant authorities.

Some underground waste disposal plants have implemented a quality management system. The scope of audits associated with these management systems are typically executed by external experts and includes all work processes at the underground waste disposal plant, together with the level of training and expertise of the staff.

Information and documentation

The following documentation is typically available in a hazardous waste facility:

- operating instructions and operating manual;
- operating log;
- annual overviews of the data in operating log;
- waste register to record disposed wastes;
- annual declaration on the behaviour of a surface landfill;
- measurements of emissions from the facility.

A2.8.7 Environmentally sound management

The underlying principle in all waste management is that the waste should be managed in an environmentally sound manner. This principle is embedded in all international waste-related agreements.

In late 1990s, it was recognised that the level of environmental safety varies widely between waste management facilities, even within OECD member countries. Therefore, the OECD started working towards international ESM guidelines to improve and harmonise the environmental protection of waste management facilities in OECD countries. The main output of this project was the Council Recommendation on ESM [C(2004)100] of waste, including the Guidance Manual for the implementation of the Recommendation C(2004)100.

The broad objectives of that work were:

- to provide facilities with common basic provisions for ESM in order to improve their environmental performance, if necessary;

- to achieve a more level playing field among facilities within the OECD area, to help ensure that facilities which have invested in environmentally sound technologies maintain their competitiveness;
- to use the implementation of these "guidelines" as a way of helping countries to have greater confidence that their waste shipments within the OECD were being sent to environmentally sound management facilities.

The Council Recommendation includes not only general policy recommendations for governments, but also practical "core performance elements" (CPEs) to be implemented by the waste management facilities. OECD recommendations are not legally binding, but there is an expectation that member countries will do their utmost fully to implement Recommendations.

This Recommendation applies to waste (hazardous and non-hazardous), whether imported or domestically generated, and to activities that collect, dispose, eventually store and recover wastes. Taking into account the size of the enterprise, especially the situation of SMEs, the type and amount of waste, the nature of the operation and domestic legislation, it recommends that facilities have an environmental management system, be inspected and/or audited in terms of environment, health and safety measures, and monitor and record their emissions and waste generation. Other measures are recommended to protect not only the environment but also the health of workers. To this end, facilities should ensure a safe and healthy occupational environment, adequately train the personnel to avoid unnecessary risks, and have an adequate emergency, closure and after-care plan for emergencies or definite cessation of activity.

Two international organisations, in addition to the OECD, have developed specific approaches to enhance ESM: the United Nations Environment Programme (UNEP), through the Basel Convention, and the North American Commission for Environmental Cooperation.

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Appendix 3

CASE STUDIES: THE MANAGEMENT OF COAL ASH, CO_2 AND MERCURY AS WASTES

This appendix presents case studies on the management of mercury (an example for theme 1 of this study) and coal ash and CO_2 following the development of carbon capture and storage (CCS) (theme 2 of this study).

Two of the primary sources of base load electricity in the future are expected to be coal equipped with carbon capture and storage capability and nuclear energy; both are likely to be need in significant quantities if the world is to meet demanding reductions on emissions of climate change gases. An objective of this study is to examine the differences in the way the waste products from these generation methods are managed. Coal ash and carbon dioxide are the main waste products from combustion of coal to generate electricity and this appendix presents an overview of some of the issues associated with their management. Management of radioactive wastes are considered in detail in Appendix 1. The aim of this appendix is to provide the basis for the broad comparison between the wastes from coal and nuclear electricity production that is presented in Chapter 3.

Mercury is an example of a highly toxic, hazardous metal. This case study explains some of its hazardous characteristics and aims to present a perspective on the management and eventual geological disposal of this highly toxic waste stream. Because the hazard from mercury does not diminish with time, when it is disposed of it must be isolated from man and the environment, effectively forever. In order to cope with safety requirements over long periods, without the need for monitoring and intervention, the trend for managing mercury waste is towards deep disposal. (Brasser, 2009) The long term isolation requirements for mercury wastes are therefore of a similar nature to those for high-level radioactive waste.

A3.1 Coal ash from power production

A3.1.1 Electricity production share and total production of ash

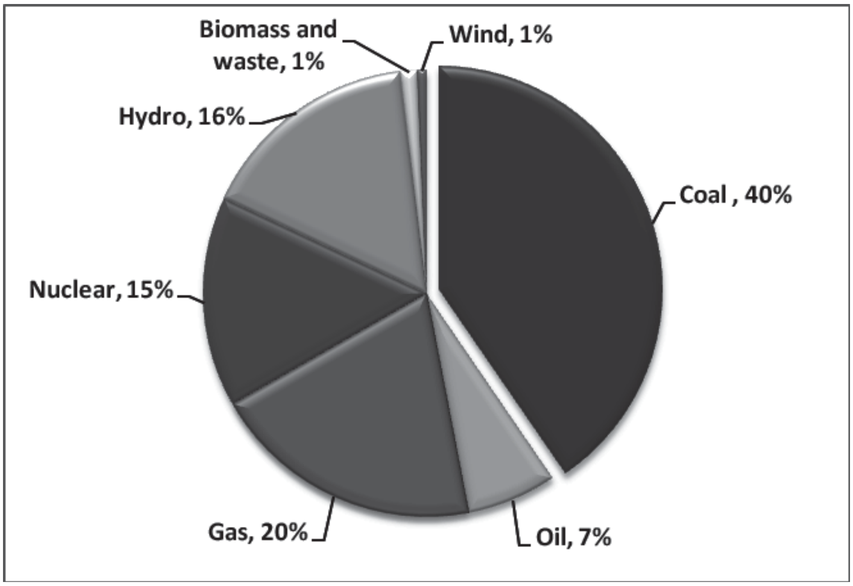
In 2005, about 40% of the world's electricity was generated by coal combustion (Couch, 2006), see Figure A3.1-1. Around 3.2 Gt of coal is used worldwide for thermo-chemical energy production each year giving rise to total of up to around 0.6 Gt of ash per year. A typical 500 MWe coal fired power station burns about 2 Mt/a of coal.

There is significant global concern about the climate change effects of CO_2 emissions from fossil fired electricity generation, which dominate anthropogenic releases to the atmosphere. However, other releases also have significant detrimental effects. Air pollution from coal-fired electricity production includes a mixture of pollutants, including fine particulate matter, carbon monoxide, nitrogen dioxide, sulphur dioxide, ozone and volatile organic compounds and inorganic substances. Air pollution control systems in modern coal fired power plants may include a scrubber system where most residues of sulphur and nitrogen oxides are removed, together with hydrochloric acid. Volatile substances like mercury and cadmium are released, to some extent, into the atmosphere along with fluorine, chlorine and bromine.

A European Environmental Agency study shows that 30% of the total PM_{10} (particles less than 10 microns in diameter) emissions in Europe result from energy production. It states that coal is a significant emitter of PM_{10} during electricity production, and should therefore be considered a significant source of health damage worldwide, even in advance economies. The *OECD Environmental Outlook* estimates that PM_{10} emissions caused 960 000 premature deaths in 2000, with 9.6 million years of life lost worldwide.

Coal combustion also releases radioactivity to the environment. The main sources of radioactivity include uranium, thorium and daughter products such as radium, radon, polonium, bismuth and lead. Although not a decay product, naturally occurring radioactive potassium-40 is also a significant contributor.

Figure A3.1-1: Electricity production share for different fuels in 2005



Sources: Couch, 2006; Joshi and Lothis, 1997; Sear, 2001; Sloss, 2007; Barnes and Sear, 2004.

Volumes of coal used and of ash generated for the purpose of power production are given in Table A3.1-1 for coal producing countries and in Table A3.1-2 for non-producing countries. It should be noted that different countries produce these data in different ways that may lead to apparent inconsistencies; these data are taken from a compiled source.

Table A3.1-1: Amounts of coal used and of ash generated in power production for coal producing countries in 2002

Country	Production bituminous coal, Mt/y	Production brown coal, Mt/y	Coal consumption (net import+export) [for coke making]	Used for power production, Mt/y	Ash [Average % in the coal]	Total ash production, Mt/y
China	1 343	50	1 315 (-78) [240]	700-750	[24%]	160-185
United States	520	473	992 (-1) [19]	820	[10-15%]	90
India	335	24	379 (+20) [23]	285-290	[30-40%]	90
Australia	256	84	133 (-207) [4]	55 & 68	[30% & 4%]	18-23
Russia	168	85	214 (-39) [40]	165-175	[10-20%]	25-35
South Africa	215	0	156 (-59) [3]	80-90	[30-35%]	20-30
Germany	29	182	242 (+31) [10]	34 & 169	[10% & 8%]	15-20
Poland	102	85	146 (-41) [14]	40 & 60	[20% & 10%]	10-15
Indonesia	103	0	29 (-74)	20-22	[10%]	2-3
Ukraine	82	1	86 (+3) [27]	40	[20-25]	10-20
Kazakhstan	71	3	50 (-24) [3]	25	[40]	8-15
North Korea	53	15	31	na		na
Greece	0	70	68 (+2)	65	[10-15]	8-12
Canada	30	37	60 (-7) [4]	11 & 44	[10% & 15%]	5-10
Czech Republic	15	48	58 (-5) [5]	4 & 41	[15%]	4-8
Turkey	2	51	65 (+12) [4]	1 & 42	[15-20%]	6-10
Colombia	40	0	3 (-37)	–		–
Serbia & Montenegro	0	34	34	na		na
Romania	4	27	34 (+3) [2]	25	[10-25%]	2-5
United Kingdom	30	0	59 (+29) [5]	46	[10%]	4-6
Bulgaria	0	26	31 (+5) [1]	24	[25-30%]	5-8
Spain	10	12	44 (+22) [4]	37	[10% & 20%]	5-10
Thailand	0	20	20	15	na	na
Vietnam	15	0	10 (-5)	8	na	na
Hungary	0	13	14(+1)[1]	12	na	na
Total	**3 423**	**1 340**	**4 273 [409]**	**≈2 975**		**487-595**

Source: Couch, 2006.

Table A3.1-2: Volumes of coal used and of ashes generated for the purpose of power production for non coal-producing countries in 2002

Country	Coal consumption, Mt/y [production Mt/y]	Used for power production, Mt/y (estimated amounts)	Ash, [Average%] quantity, Mt/y
Japan	160 [1]	85	[12%] 7
South Korea	75 [3]	43	[12%] 5
Taiwan	51	42	[12%] 5
Italy	20 [2]	14	[12%] 1.7
France	19 [2]	9	[12%] 1.1
Brazil	18 [5]	4	[15%] 0.6
Philippines	13 [2]	na	na
Netherlands	13	9	[12%] 1.1
Israel	12	10	[12%] 1.2
Belgium	11	4	[12%] 0.5
Total	**392 [15]**	**≈220**	**≈23 Mt/y**

Source: Couch, 2006.

A3.1.2 Properties of coal and the combustion process

Coal is thought to originate from organic matter in the form of peat that has undergone various ageing processes (diagenesis and metamorphosis) during geological times of tens to hundreds of million years. Coal is a sedimentary rock that occurs in layers coalesced and modified from former peat deposits. The orientation is frequently horizontal but many seams are inclined due to folding, faulting and orogenic displacement of the rock.

Coals vary considerably in character. Recent coals having ages less than around 65 million years are often lignites with considerably higher contents of inorganic constituents than the typical value for older coals, which is around 15%. The geological and chemical processes involving high pressures and temperatures, working over time, have compressed and altered plant remains, increasing the percentage of carbon present, and thus producing the different ranks, or varieties, of coal. Coals are classified based on fixed carbon, volatile matter, and heating value. The incombustible matter in coal, which acts to lower the relative amounts of carbon and thus the rank of coal, becomes ash after burning. Minerals represent the inorganic parts of coal and include clay (the most abundant inorganic constituent), carbonates, sulphides and quartz, which were either washed into the original swamp plant materials that ultimately were compressed to form peat, or portions of confining rock beds inadvertently mined with the coal. Radionuclides are incorporated into coal as they may be found in the original peat beds or in layers of interspersed inorganic material, or because of intrusion during or after coalification by leaching from surrounding rocks and soils (EPA, 1973; EPA, 1977; DOI, 1963).

The quality of coals also varies considerably with regard to coking properties. Dry distillation (pyrolysis, heating without access to air) of coal gives rise to gas as well as liquids. The proportions of coke, tar and gas depend highly on the individual type of coal used. The same can be said of the mechanical integrity of the coke that is dependent on formation of tar, which on further heating decomposes to form an efficient binding agent between the grains of the coal.

Coal processing before utilisation and burning in furnaces includes blending, pulverisation, washing and flotation to remove as much incombustible mineral material as possible. This increases the heating content of the coal, and serves to minimise, though not eliminate, the amount of ash and clinker generated in the combustion process. Modern coal-fired thermo-chemical plants utilise pulverised fuel to achieve a good contact between the coal grains and the surrounding gasses. Air jets are used to ensure rapid and efficient contact.

There are two main types of furnaces, those with and those without a fluidised bed of fine sand material. The sand assists in transferring heat from the burning particles and to the heat transfer pipes.

A particle in a coal powder burner oxidises in a few milliseconds (Wooley, *et al.*, 2000). Typical temperatures in the hottest parts approach 1 650°C. The maximum temperature is intentionally kept at least 100°C lower than that for stoichiometric composition in the feed in order to reduce the formation of oxides of nitrogen for which strict limits apply for emission. Additional air is added in the form of jets a little higher in the furnace to ensure excess of oxygen everywhere in the flue gasses. Typical residence time for the fuel particles in the furnace is 3-4 seconds (Wooley, *et al.*, 2000). In this way, the combustion process becomes completed with a high efficiency.

A3.1.3 Air pollution control systems and means of ash removal

Some of the ash simply falls down by gravity to the lower parts of the furnace (including the reheater and economiser parts). Other fractions of the ash are collected by means of a cyclone. Frequently, both of these are referred to as "bottom ash" (and they may be mixed in the process of removal) as opposed to the ash leaving the furnace area together with the flue gasses which is referred to as "fly ash". In a modern coal combustion facility, most of the ash (around 80%) (Wooley, *et al.*, 2000) is collected in the form of fly ash.

In the majority of cases, most of the fly ash is removed by electrostatic precipitation.[1] In addition – or alternatively – bag filters[2] may be applied, sometimes in conjunction with dry or semi-dry[3] chemical air pollution control.

Air pollution control in modern coal combustion facilities may also include a scrubber system where residues of sulphur and nitrogen oxides are removed, together with hydrochloric acid. The main reaction product from such systems is gypsum (calcium sulphate).

Mercury, and to a certain extent cadmium, are much more volatile than other heavy metals present in fumes from coal combustion. They do therefore not condense efficiently in the ash and may be emitted and become an environmental and health hazard even if the fumes are cleaned by

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1. The fumes pass areas of high electrostatic fields that make the charged particles move and attach to surfaces from which they are intermittently removed.
2. The ash is removed by recurrent back flushing.
3. Lime sludge is sprayed into the fumes. The feed is adjusted such that the spray will dry before reaching the filters.

mechanical filters (electrostatic filters and bag filters). Efficient removal of these species may be achieved by adding active carbon to the fumes and/or use wet air pollution control systems (scrubber) in which case the combustion residue will be contaminated with material that has not been combusted.

The need for chemical pollution control is strongly dependent on the quality of the coal, mainly its content of sulphur and mercury. It is also strongly dependent on the type of furnace. Fluidised bed types of furnaces have lower temperatures resulting in formation of less nitrogen oxides. The bottom ash from such furnaces, or rather "bed ash" as it is usually called in this case, invariably contains some of the bed material as well.

Thus, no visible combustion fumes leave a modern coal-fired plant. The only "smoke" that can be observed is some condensation in the air leaving the cooling towers (of water previously evaporated inside the tower). Under normal weather conditions, this condensation soon re-evaporates as the air from the cooling towers is mixed with the surrounding air. Large amounts of invisible carbon dioxide leave the stacks from coal-fired thermo-chemical plants, and this is of great concern since it is a major contributor to climate change.

The means of removing ash vary, and may not be in exact one-to-one correspondence to the processes installed. It was noted above that bag filters for fly ash might be combined with semi-dry chemical air pollution control. The ash removal systems may be designed in such a way that ash from individual removal points may not be taken out separately. This may apply to ash from different units as well.

The bottom ashes (and/or bed ashes and/or cyclone ashes) in particular may be very hot at the point where they are removed. This may make them difficult to handle due to the continuing combustion of residues of burnable material. Therefore, such ashes are often removed by passing them through a water bath. Wet ashes are handled and managed separately from dry ones.

A3.1.4 Ash classification schemes

Not unexpectedly, ashes are classified differently in different countries and also between different utilities, plants and combustion units. Interpretations and usage of the various terms may vary since precise definitions of the categories may exist only at a particular plant level. It is important to note that the categories include large-volume categories as well as small-volume ones, and that the pertinent strategies for the management of the ashes may vary considerably depending on the volume of the material in question.

As an example, the United States Environmental Protection Agency in their *Report to Congress on Wastes from the Combustion of Fossil Fuels* [EPA, 1999] uses the following categories for the large volume residues:

- fly ash;
- bottom ash;
- boiler slag;
- flue gas desulphurisation (FGD) sludge.

The following categorisation is mentioned for small-volume residues:

- coal pile runoff;
- coal mill rejects/pyrites;
- boiler blow-down;
- cooling tower blow-down and sludge;

- water treatment sludge;
- regeneration waste streams;
- air heater and precipitator wash water;
- boiler chemical cleaning waste;
- floor and yard drains and sumps;
- laboratory wastes;
- waste water treatment sludge.

It should be remembered that even with a perfect categorisation, there are variations in properties from one time to another. The main reasons for this are variations in the fuel and in the thermal load of the unit in question. In addition, since the residues are usually reactive with regard to moisture and carbon dioxide, the properties may vary with time after the waste products have been removed.

Some reasons were given in the previous section why ashes from different removal points in the same unit usually have very different properties. However, the most important differences are those related to chemical composition and to the partitioning processes that take place because of evaporation and fractional condensation in the furnace. A contributing factor here is also differences in thermal history, e.g. differences in the rate of cooling.[4]

It is the larger ash particles that form the bottom ash while fly ash has a small particle size where most of the material is in the 0.005-0.02 mm range. (Wooley, *et al.*, 2000) The reason for the small particle size and the partitioning with regard to particle size is the transient and rapid events in the furnace. There is little time for diffusion of condensing matter to the larger particles, and therefore volatile material preferentially condenses on the small particles.

A classification of a number of trace elements in coal ash with respect to their behaviour in a furnace environment is presented in Table A3.1-3.

Table A3.1-3: Classification of trace elements with regard to their volatility in a furnace environment

Group	Elements
3	Hg, Br, Cl, F
2+3	B, Se, I
2	As, Cd, Ga, Ge, Pb, Sb, Sn, Te, Tl, Zn
1+2	Ba, Be, Bi, Co, Cr, Cs, Cu, Mo, Ni, Sr, Ta, U, V, W
1	Eu, Hf, La, Mn, Rb, Sc, Sm, Th, Zr

Note: The elements in Group 3 are the most volatile, and those in Group 1 are the least volatile.

Source: Sloss, 2007.

The major elements are not included in Table A3.1-3. They are nonetheless important since there is a competition between various elements with regard to e.g. chlorine. Thus, sodium and potassium are over-represented in the fly ash. They tend to condensate as chlorides. Silicon and aluminium are over-represented in the bottom ash while calcium and magnesium may not exhibit a preference.

4. Very rapid cooling (quenching) gives rise to a more reactive material as compared to slow cooling (other factors being the equal).

A3.1.5 General chemical composition of coal and coal combustion residues

The chemical composition of coal and the major elements[5] in the corresponding ash is presented in the form of a few examples in Table A3.1-4b, c. The examples are taken from a wide range of coals, mainly from exporting countries. The work was carried out at a test facility; therefore, the ash in this case represents all of the ash except that which is typically absorbed in the chemical cleaning of the flue gasses. Trace elements in coal and their intervals of occurrence are presented in Table A3.1-4a.

Table A3.1-4a: Trace elements in international thermal coals compared with Australian coals, (mg/kg)

	International coals			Australian coals		
Element	**Average**	**Low**	**High**	**Average**	**Low**	**High**
As	3.3	0.32	26	0.93	0.1	2.7
B	59	6	143	21	4	36
Be	0.95	0.1	3.2	0.82	0.2	2.1
Br	7	2	38	5	2	17
Cd	0.07	0.01	0.19	0.09	0.01	0.28
Cl	310	10	1 470	320	10	1500
Co	4.7	1	13	3.7	1.2	12
Cr	12	2	34	9	2.9	24
Cu	9	1	28	14	6.2	32
F	100	15	305	98	35	340
Hg	0.066	0.01	0.19	0.021	0.006	0.08
I	3	2	7	6	2	14
Mn	44	8	123	99	4	700
Mo	1.1	0.07	4.2	0.85	0.1	2.7
Ni	9	1	22	8.6	1.4	31
Pb	7.2	0.5	22	5.8	2.2	14
S.%	0.65	0.115	3.0	0.6	0.21	0.95
Sb	0.37	0.02	1.4	0.46	0.05	1.2
Se	1.4	0.1	5.3	0.47	0.12	1.1
Th	3.1	0.1	12.2	2.6	0.5	6.9
U	1.2	0.02	5.5	0.93	0.27	2.5
V	20	1.5	54	23	7	62
Zn	12	4	55	14	4	51

Sources: Couch, 2006; Dale, 2005.

5. The major elements are figured as hypothetical formula units.

Table A3.1-4b: Examples of the chemical composition of coal and the corresponding ash (continuation)

Coal	Harworth	Bailey	PRB1	PRB2	Bowen basin	Hunter Valley	Prodeco	Goedehoop	Talcher	JR/PRB1 blend	PRB1/Bailey blend
Origin	**UK**	**Eastern US**	**Western US**	**Western US**	**Australia**	**Australia**	**Colombia**	**South Africa**	**India**	**US**	**US**
H_2O, %	2.5	2.3	18.0	19.7	4.8	3.2	3.7	2.8	9.7	18.9	11.2
ash, %(ar)	14.4	8.9	3.7	5.8	7.6	9.8	8.6	13.1	39.7	4.8	5.6
VM, %(ar)	31.4	34.5	34.5	35.0	27.8	30.7	35.5	25.4	24.0	43.7	34.8
GCV, MJ/kg(ar)	28.9	31.0	24.1	22.5	28.9	29.9	29.6	28.2	15.1	23.4	27.5
NCV, MJ/kg(ar)	27.8	29.9	22.9	21.3	28.0	28.9	28.5	27.4	14.3	22.2	26.3
S, %(ar)	2.3	1.3	0.33	0.33	0.4	0.46	0.66	0.7	0.37	0.46	0.8
Cl, %(ar)	0.20	0.21	0.01	0.01	0.01	0.02	0.5	0.01	0.01	0.02	0.17
C, %(daf)	82.4	83.7	76.3	77.4	81.8	83.0	81.4	83.5	74.5	75.9	80.6
H, %(daf)	5.5	5.3	4.4	4.4	4.5	4.9	5.5	4.5	4.9	4.4	4.9
N, %(daf)	1.78	1.66	0.98	1.12	1.77	1.86	1.68	2.03	2.07	1.1	1.41
O, %(daf)	7.4	7.7	17.9	18.6	11.5	9.7	10.1	9.2	17.7	18.0	11.9
VM, %(daf)	37.8	38.9	44.1	47.0	31.7	35.3	40.5	30.2	47.4	45.2	41.8
Fuel ratio	1.65	1.57	1.27	1.13	2.15	1.83	1.47	2.31	1.11	1.20	1.39

Notes: Coal analysis determines the amount of fixed carbon, volatile matters (VM), moisture and ash within the coal sample. The variables are measured in weight percent (wt. %) and are calculated in several different bases. AR (as-received) basis is the most widely used basis in industrial applications. AR basis puts all variables into consideration and uses the total weight as the basis of measurement. DAF (dry, ash free) basis neglect all moisture and ash constituent in the coal. GCV is gross calorific value; NCV is net calorific value.

Sources: Couch, 2006; Wigley and Williamson, 2005.

Table A3.1-4c: Normalised ash compositions, (wt%) (continuation)

Coal	Harworth	Bailey	PRB1	PRB2	Bowen basin	Hunter Valley	Prodeco	Goedehoop	Talcher	JR/PRB1 blend	PRB1/Bailey blend
Origin	**UK**	**Eastern US**	**Western US**	**Western US**	**Australia**	**Australia**	**Colombia**	**South Africa**	**India**	**US**	**US**
SiO_2	50.8	56.4	36.3	39.2	61.5	81.6	63.4	43.1	67.2	38.6	48.5
Al_2O_3	26.1	25.4	19.7	20.9	31.0	13.2	20.0	33.3	24.3	19.1	24.9
Fe_2O_3	14.5	10.7	6.2	6.8	4.1	2.8	7.2	4.8	2.9	6.4	9.7
CaO	1.2	2.1	20.9	23.0	0.5	0.3	2.5	10.9	1.1	22.6	7.6
MgO	1.2	0.9	5.6	4.6	0.2	0.3	2.4	2.6	0.8	5.4	2.4
K_2O	3.9	2.3	0.7	0.5	0.3	0.9	2.4	0.5	1.8	0.5	2.0
Na_2O	0.8	0.5	7.6	1.5	0.1	0.1	0.8	0.3	0.1	4.0	3.0
TiO_2	1.0	1.6	1.8	1.8	1.9	0.7	1.0	1.7	1.3	1.8	1.5
BaO	0.1	0.1	0.9	0.7	0.1	0.03	0.2	0.4	0.1	0.8	0.3
Mn_3O_4	0.05	0.03	0.06	0.03	0.01	0.03	0.07	0.08	0.03	0.05	0.02
P_2O_5	0.3	0.15	0.21	1.1	0.39	0.07	0.16	2.4	0.46	0.68	0.16
SiO_2/Al_2O_3	1.94	2.22	1.84	1.87	1.99	0.20	3.18	1.29	2.76	2.02	1.95
Base/acid ratio	0.26	0.20	0.71	0.59	0.05	0.05	0.18	0.24	0.07	0.66	0.33

Source: EPA, 2006.

A3.1.6 Environmental and health properties of coal ash

The environmental and health properties of coal ash are determined by examining the exposure pathways. Generally, oral intake of liquids (drinking water) and solids (including food) together with inhalation are the pathways considered for exposure for most hazardous substances. In most cases, oral intake from drinking water is the dominant exposure pathway for inorganic components and organic compounds to humans.

For radioactive elements arising from coal combustion, external radiation and inhalation need to be considered as well. For radioactive elements, the principal exposure pathways are through external radiation and inhalation (radon gas and particulates), but this varies by radionuclide and radiation source of exposure.

Use of efficient particle filters at thermo-chemical coal-fired plants has reduced inhalation impacts from smoke stack emissions, but not necessarily in other exposure situations.

Exposure scenarios for living organisms other than humans may be dominated by uptake from surface and groundwater as well as direct radiation exposure. However, these protection criteria are currently designed for protection of populations, not individuals primarily due to lack of data and understanding of health and environmental impacts to animal systems.

Inorganic compounds

Typical leach data for shake tests[6] can be found in Table A3.1-5. The test used resembles the European Union standard test prEN 12457-2 for acceptance for landfills.

Table A3.1-5: Typical ranges for leach data (in mg/litre) for ashes from the United Kingdom using the shake test DIN 38414-S4

Element	Typical range of leachable elements	Element	Typical range of leachable elements
Aluminium	<0.l*-9.8	Magnesium	<0.l*-3.9
Arsenic	<0.l*	Manganese	<0.l*
Boron	<0.l*-6	Molybdenum	<0.l*-0.6
Barium	0.2-0.4	Sodium	12-33
Calcium	15-216	Nickel	<0.l*
Cadmium	<0.l*	Phosphorus	<0.l*-0.4
Chloride	1.6-17.5	Lead	<0.2*
Cobalt	<0.l*	Sulphur	24-510
Chromium	<0.l*	Antimony	<0.01*
Chromium VI	<0.l*-l	Selenium	<0.01*-0.15
Copper	<0.l*	Silicon	0.5-1.5
Cyanide	<0.01*	Tin	<0.l*
Fluoride	0.2-2.3	Titanium	<0.l*
Iron	<0.l*	Vanadium	<0.l*-0.5
Mercury	<0.01*	Zinc	<0.l*
Potassium	1-19	pH	7-11.7

* Value below detection limit. Water to solids ratio is 10/1 litres per kilogram. The data include a seawater-conditioned sample; hence, the high chloride values.

Source: Sear, 2001.

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6. Where a sample is gently shaken or tumbled for 24 hours with e.g. ten times its dry weight of de-ionised water.

Organic compounds

The presence of organic compounds such as polycyclic aromatic hydrocarbons (PAH) and dioxin are of constant concern. Historically, their impact on health has been huge due to bad combustion and lack of air pollution control (APC). Extensive research has been carried out to reduce these emissions. Today emissions and their impact are low due to relatively extensive efforts at power plants (the APC building is usually much larger than the furnace building). However, it is difficult to extract all PAH and dioxin from the ash and so there is a debate as to whether it is all measured.

Two classes of organic compounds are of primary interest from a health and environment point of view: polyaromatic hydrocarbons and dioxins. Each of these classes comprises a number of different individual compounds of variable toxicity. Some of the species are very toxic, and may also be carcinogenic, and consequently they have to be restricted to very low levels. Even though the content of polyaromatic carbons in ash is low, the volumes of coal combusted are large.

Extensive research has been performed to evaluate the levels of these compounds in ashes from power production. According to a review in 1995 (Sear, 2001; Wild and Jones, 1995) the major source in the environment, apart from gasworks sites, were found to be coal-fired electricity generation (3 140 tonnes per year in the United Kingdom). These results have been challenged to some extent. It has been said that polyaromatic hydrocarbons in ash are not available to the environment (the half-life of dioxin in ordinary soil is about 2 years), and leach tests in accordance with the method of the United Kingdom Environment Agency have indicated levels for the major species to be less than 0.2 micrograms per litre.

According to Sear (2001), dioxins are unlikely to form under conditions found in coal combustion furnaces, and only traces can be expected in the resulting ash. Various researchers (Sear, 2001) have confirmed that no dioxins over 0.000025 mg/kg are generally found in ashes from coal-fired power plants. This is similar to levels found in typical soils. However, more recent research (Sear, Weatherley and Dawson, 2003) with reference to (JEP, 2003) reports that more efficient techniques have been utilised to extract the polyaromatic carbons from the ash resulting in total values up to 25 mg/kg, though more than half of the values determined were reported to be less than 10 mg/kg. Even if the new data represents significantly higher values than those reported previously, the overall values are relatively low.

Radioactive elements

All of the radon present in the coal is emitted to the air during combustion. (Smith, *et al.* 2001) However, the source for future generation of radon remains in the coal ash. Radon has three radioactive isotopes (see Table A3.1-6).

Table A3.1-6: The isotopes of radon

Natural decay series	Isotope	Named as	Half-life
Uranium	^{222}Rn	Radon	3.82 days
Thorium	^{220}Rn	Thoron	55 seconds
Actinium	^{219}Rn	Actinon	4 seconds

Source: Brune, *et al.*, 2001.

It is clear from the half-lives shown in Table A3.1-6 that radon gases formed in the ash will reach near equilibrium with their parents in periods of between one minute and two months. The radon present in the coal at the time of combustion leaves via the stack during combustion and so does not appear in the ash. However, this is the case only for a short time as the radon gases then "grow back" into the ash. It is important to be aware that radon behaves differently from all other potentially hazardous components.

The source for Radon-222 is Radium-226, which has a half-life of 1 620 years. The chemistry of radium is very similar to that of barium, which probably acts as a carrier for the radium. According to Chandler, *et al.* (1997), barium is not emitted with the flue gasses but stays in the ash, who states that the radioactivity stays in the ash on combustion (with the exception of the radon already formed).

Actual data on radionuclide content of various coal ashes can be found in Table A3.1-7 and data on natural radionuclide in building materials and extract of relevant parts are presented in Table A3.1-8.

Table A3.1-7: Radioactivity in some coal fly ashes (Bq/kg)

Reports from	Ash from	U-Series			Th-series		
		Min	Max	Average	Min	Max	Average
Germany	Germany	93	137	119	96	155	121
	United Kingdom	72	105	89	3	94	68
	Australia	7	160	90	7	290	150
	Poland			350			150
				189			118
Italy	Italy	130	210	170	100	190	140
Denmark	Denmark	120	210	160	66	190	120
Sweden	Sweden	150	200		150	200	
Belgium	Belgium	112	316	181	88	277	150
Spain	Spain	80	106	91	77	104	89
Czech Republic	Czech Republic	35	190	129	62	142	90

Sources: UNIPEDE/EURELECTRIC, 1997; EPA, 1995; EPA, 1984; Push, *et al*, 1997; IAEA, 2003.

Table A3.1-8: Extract of data for concrete and coal ash from European Commission report

Material	Typical activity concentration (Bq/kg)			Maximum activity concentration (Bq/kg)		
Building material	Ra-226	Th-232	K-40	Ra-226	Th-232	K-40
Concrete	40	30	400	240	190	1 600
Coal fly ash	180	100	650	1 100	300	1 500

Source: EC, 1999a.[7]

7. According to the foreword, a working party of the Group of Experts established under the terms of Article 31 of the Euratom Treaty has examined the issue of regulatory control of building materials with regard to their content of naturally occurring radionuclides.

   The working party developed guidance based on a study providing information about natural radioactivity in building materials and relevant regulations in Member States. This guidance was adopted by the Article 31 Group of Experts at its meeting on 7-8 June 1999 and was published with a view to harmonisation of controls by Member States, in particular in order to allow movement of building products within the European Union.

   This guidance was expected to be a useful reference document for the European Commission when considering possible regulatory initiatives at Community level. The Member States have now implemented the Euratom Directive in their national legislation, but despite the Commission's guidance documents, there may very well be significant differences in the national regulations. (Van der Steen, 2006)

Typical concentrations are population-weighed national means of different Member States. Maximum concentrations are maximum values reported in EC (1999b). Higher values might have been reported elsewhere.

A3.1.7 Recycling of coal ash versus disposal

Fraction of coal ash that is recycled

The fraction of coal ash that is recycled varies significantly between countries. Some country specific data can be seen in Table A3.1-9 (United States), Table A3.1-10 (15 EU countries), Table A3.1-11 (Canada) and Table A3.1-12 (Japan). The structure of these tables differs to reflect the different structuring of combustion categories in these countries.

Table A3.1-9: Generation of various residues in 2002 from coal-fired power plants in the United States together with their utilisation (units: tonnes)

Category of residue	Total generation	Total utilisation	Utilisation %
Fly ash	76 500 000	26 628 881	34.8
Bottom ash	19 800 000	7 689 589	38.8
Gypsum*	11 400 000	7 770 000	68.2
Wet scrubbers*	16 900 000	560	3.3
Boiler slag	1 919 579	1 549 972	80.8
Dry scrubbers*	935 394	371 404	39.7
Other*	0	0	
Fluidised bed combustion ash	1 248 599	95 341	76.4
Total	128 703 572	45 523 256	35.37

* From desulphurisation.

Source: Barnes and Sear, 2004. Data from plants responding to survey extrapolated to include all except for categories in italics for which no extrapolation was carried out.

Table A3.1.10: Generation of various residues in 2002 from coal-fired power plants in Europe (EU 15*) together with their utilisation
(units Mtonnes)

	Ash production	Total utilisation excluding reclamation	Total utilisation excluding reclamation %	Total utilisation including reclamation	Total utilisation including reclamation %	Reuse of stockpiled coal combustion residues	Total production
Fly ash	39.947	18.745	46	35.755	88	0.638	40.585
Bottom ash	5.84	2.42	41	5.211	89	0	5.84
Boiler slag	2.24	2.24	100	2.24	100	0	2.24
Fluidised bed-ash	1.06	0.568	54	0.711	67	0	1.06
Other	0.218	0.218	100	0.218	100	0	0.218
Spray dry absorption-product	0.515	0.297	58	0.482	94	0	0.515
Flue gas desulphurisation-gypsum	9.767	7.088	73	8.326	85	0	9.767
Total	59.587	31.576	52.4	52.943	87.9	0.638	60.225

* EU 15 = Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxemburg, Netherlands, Portugal, Spain, Sweden and United Kingdom.

Source: Barnes and Sear, 2004.

Table A3.1-11: Generation of various residues in 2002 from coal-fired power plants in Canada together with their utilisation (units: Mt)

Category of residue	Total generation	Disposed/stored	Removed from storage	Total use	Utilisation %
Fly ash	5.030	3.985	0	1.094	22
Bottom ash	1.558	1.472	0.138	0.196	13
Gypsum*	0.421	0	0	0.570	135
Other	0.128	0.124	0	0	0
Total	**7.137**	**5.582**	**0.138**	**1.860**	**26.1**

* From desulphurisation.

Source: Barnes and Sear, 2004.

Table A 3.1-12: Coal consumption for energy production and generation of coal ash together with the degree of utilisation in Japan during 2001-2005 (units: Mt)

Fiscal year	Coal consumption	Total ash generation	ash content %	Utilisation	Utilisation %
2001	59.159	6.785	11.5	5.271	77.7
2002	64.251	6.920	10.8	5.495	79.4
2003	68.981	7.475	10.8	6.105	81.7
2004	74.270	8.052	10.8	7.128	88.5
2005	78.092	8.334	10.7	7.899	94.8

Source: Watanabe, personal communication.

Specific uses of coal ash in society

The overall prerequisites for use and disposal of residues from coal combustion are that the practice should be:

1. sound and acceptable from a health and environment perspective;
2. technically feasible;
3. logistically feasible.

Although the levels and availabilities of various potentially harmful species are low or moderate in coal combustion residues, it is important that each case be evaluated based on its specific conditions. The presence of certain species at elevated levels may prohibit or impede utilisation for certain purposes, e.g. as soil amendment.

The technical feasibilities include a number of possible properties:

1. Fineness, such that voids can be filled and reactivity is high.
2. Rounded shape of the (fly ash) particles such that such that the shear resistance is low (good flow properties) in slurries with high particle loadings. This facilitates mixing, filling up of pore space, compacting, etc.

3. Pozzolanic[8] reactions (fly ash) improve properties of concrete and mortar above that of good pore filler. It makes the material tighter to penetration of water and more resistant to chemicals and weathering.

4. Low heat of curing (fly ash) facilitates the use in large constructions.

5. Good draining properties make a material (e.g. bottom ash or bed sand) useful in geotechnical constructions.

6. Content of fertilisers and alkaline buffer capacity are valuable in additives to soil.

Data on the specific uses of coal combustion residues in the United States is presented in Table A3.1-13. The data in Table A3.1-13 correspond to the data in Table A3.1-9. Data on the various specific uses of coal combustion residues in 15 countries in the European Union are presented in Tables A3.1-14. The data in Table A3.1-14 correspond to the data in Table A3.1-10.

8. Some activated silicate-aluminate systems react with lime. They are called pozzolana after the Pozzol volcano where the Romans found material for their cement. It was made of a mixture of lime and volcano ash or a mixture of lime and crushed burnt clay.

Table A3.1-13: Generation of various residues in 2002 from coal-fired power plants in the United States together with their utilisation (units: tonnes)

	Coal combustion residue category =>	Fly ash	Bottom ash	FGD gypsum*	Wet scrubbers*	Boiler slag	Dry scrubbers*	FGD other*	FBC ash
1.	Concrete/concrete products/grout	12 579 136	406 255	60 606	0	9 000	35 436	0	0
2.	Cement/raw feed for clinker	1 917 690	585 480	303 807	0	0	3	0	0
3.	Flowable fill	455 018	0	0	0	0	1 014	0	0
4.	Structural fills/embankments	4 200 982	2 046 545	0	427 000	12 103	0	0	0
5.	Road base/sub-base/ pavement	767 182	1 472 291	0	616	4 484	2 558	0	0
6.	Soil modification/stabilisation	904 745	98 509	0	0	0	0	0	0
7.	Mineral filler in asphalt	103 173	96 218	0	0	38 496	2 852	0	0
8.	Snow and ice control	2 645	767 455	0	0	8 612	0	0	0
9.	Blasting grit/roofing granules	61 964	137 455	0	0	1 440 706	0	0	0
10.	Mining applications	1 888 855	802 582	0	131 600	0	258 043	0	760 000
11.	Wallboard	0	0	7 247 856	0	0	0	0	
12.	Waste stabilisation/solidification	3 187 773	19 091	0	0	0	67 053	0	193 410
13.	Agriculture	0	6 873	77 700	0	0	0	0	
14.	Aggregate	0	678 109	6 216	0	3 200	1 448	0	
15.	Miscellaneous/other	559 718	572 727	73 815	784	33 371	0	0	
CCP Category use totals		26 628 881	7 689 589	7 770 000	560 000	1 549 972	371 404	0	953 410
CCP Category production totals		76 500 000	19 800 000	11 400 000	16 900 000	1 919 579	935 394	0	1 248 599

* From desulphurisation. FDG = flue gas desulphurisation. FCB = fluidised bed combustion.

Source: Barnes and Sear, 2004. Data from plants responding to survey extrapolated to include all except for categories in italics for which no extrapolation was carried out.

Table A3.1-14: Generation of various residues in 2002 from coal-fired power plants in Europe (EU 15*) together with their utilisation (units Mt)

Ash utilisation (Mt)		Fly ash	Bottom ash	Boiler slag	FBC ash	Other	SDA-product	FGD-gypsum	Total	%
Cement raw material	1	4 465	170						4 635	7,7
Blended cement	2	2 042	122		16				2 180	3,6
Concrete addition	3	5 510	0	150	4				5 664	9,4
Aerated concrete blocks	4	746	16						762	1,3
Non-aerated concrete blocks	5	342	1 169						1 511	2,5
Lightweight aggregate	6	107	0				2		109	0,2
Bricks + ceramics	7	90	27			18			135	0,2
Grouting	8	523		170	3				696	1,2
Asphalt filler	9	187							187	0,3
Subgrade stabilisation	10	188	81		41				310	0,5
Pavement base course	11	356	195	1 220	55				1 826	3
General engineering fill	12	1 589	474		4	67	78		2 212	3,7
Structural fill	13	1 445	119		52				1 616	2,7
Soil amendment	14	94	13				0		107	0,2
Infill	15	616	11		368		147		1 142	1,9
Blasting grit	16	22		580					602	1
Plant nutrition	17	4					22		26	0
Set retarder for cement	18							760	760	1,3
Projection plaster	19							726	726	1,2
Plaster boards	20							4 131	4 131	6,9
Gypsum blocks	21							226	226	0,4
Self levelling floor screeds	22							1 239	1 239	2,1
Other uses	23	419	23	120	25	133	48	6	774	1,3
Total utilisation 1-23		18 745	2,42	2,24	568	218	297	7 088	31 576	52,4

* Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxemburg, Netherlands, Portugal, Spain, Sweden and United Kingdom. SDA = spray dry absorption. FGD = flue gas desulphurisation.

Source: Barnes and Sear, 2004.

A3.1.8 Waste acceptance and disposal

In the European Union, there are three types of landfills: for inert waste, for non-hazardous waste and for hazardous waste. The acceptance of waste is dictated by the Council Decision of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills. (EC, 2003) This decision is implemented in the legislation of the various member countries.

According to the acceptance criteria, a number of specific waste categories are mentioned together with the respective destinations allowed. Residues from coal combustion are not included in these listings.

In general, wastes not specifically listed are to undergo so-called "basic characterisation" which implies short-term shake and column tests. The values obtained in these tests are compared with limits listed in tables for landfills for inert, non-hazardous and hazardous waste. Waste that does not meet the criteria even for acceptance at a landfill for hazardous waste cannot be deposited, but has to be treated until it meets any of the criteria.

There is one exception to this, and the following is stated in section 2.2.1:

> "Municipal waste as defined in Article 2(b) of the Landfill Directive that is classified as non-hazardous in Chapter 20 of the European waste list, separately collected non-hazardous fractions of household wastes and the same non-hazardous materials from other origins can be admitted without testing at landfills for non-hazardous waste."

Consequently, in Europe, residues from combustion of coal may be deposited on landfills for inert, non-hazardous or hazardous waste depending on their chemical compositions as well as on their leaching properties.

The broad waste management strategy is similar in the United States. Generally, non-hazardous waste can be deposited on ordinary landfills, and hazardous waste can be deposited at landfills for hazardous waste if the leach criteria are met. In the United States, residues from combustion of coal have been classified as non-hazardous by the Environment Protection Agency (EPA)[9] and this classification has been adopted in many states. Sates have the right to impose their own, more demanding classification and some have established testing conditions (including leach tests) or landfill design requirements for disposal. There have been instances where naturally occurring radionuclides have posed an environmental problem. The state of New Jersey does not allow fly ash to be used as daily cover because of its radioactivity. (NJUS, 2009) In two cases, landfilled coal ash has contributed to the radon and radionuclide levels of Superfund sites. (EPA, 1996, 2005b)

Of concern is that conditions of extreme pH in groundwater are common in ash disposal areas associated with coal-fired power plants. (NRC, 1984) This relationship of pH to uranium leaching is important because uranium is soluble in both alkaline and acidic conditions. Radium, to a smaller extent, is also soluble in water and both uranium and radium may be found in coal ash. A discussion of this matter is found in EPA (2007) which references associated publications on leachability of radionuclides.

In the most cases however, in Europe as well as in the United States, residues from coal combustion may be expected to pass the criteria for disposal on sites for non-hazardous waste.

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9. There is apparently now some reconsideration of this classification.

A3.1.9 Coal ash from power production – summary

- Around 40% of the world's electricity is generated using around 3.2 Gt/a of coal and creating 0.5 to 0.6 Gt/a of ash. The mass of these ash residues are 13 to 16% of the initial coal mass.
- In most countries coal ash is not regarded as a hazardous waste.
- Table A3.1-15 provides a perspective on the global quantities of selected elements that are released to the environment primarily in gaseous form or primarily as ash. These data assume elemental concentrations in international coal (see Table A3.1-4) and a combustion rate of 3.2Gt/a.

Table A3.1-15: Global discharge rates of some elements from coal generation plants

Examples of elements released primarily in gaseous form	**Global discharge rate (t/a)**
Mercury	210
Bromine	22 000
Fluorine	320 000
Chlorine	990 000
Examples of elements released primarily with ash	
Beryllium	3 000
Uranium	3 800
Thorium	9 900
Arsenic	11 000
Lead	23 000

- In the United States, about 35% of coal ash is recycled (46 Mt/a) whilst in the former EU15 about 88% is recycled (53 Mt/a).
- Coal ash generally has low specific radioactivity, with average concentrations ranging from 157 Bq/kg in the United Kingdom to 500 Bq/kg in Poland. Maximum radioactivity concentrations of 2 900 Bq/kg have been reported.
- The main recycling uses of coal ash are:
  - concrete products and cement;
  - structural fills and embankments;
  - road base construction;
  - mining applications.
- In addition, calcium sulphate produced from flue gas desulphurisation plants is recycled into wallboards and boiler slag is reused for grit blasting.

Clearly, the world of coal ash is different to that of radioactive waste in many respects, for example:

- In comparison with radioactive waste, the solid residues from coal generation have very large mass.
- A large fraction of the residue is reused in the economic cycle to replace large volumes of virgin raw materials; very little radioactive waste is recycled.

- Because such a large fraction of coal residue is reused, the distinction between a waste and a product is not as clear-cut as it is for radioactive waste.
- The nature and oversight of the regulations as well as the waste acceptance criteria for waste disposal are less demanding for coal residues.
- However, the ethical principles that form policies for the management of the two waste types, including the overall aim to protect the environment, are broadly similar.

A3.2 Mercury containing waste

A3.2.1 Background

Because of its unique chemical and physical properties, mercury has proved to be useful in numerous products and chemical processes. As a result, mercury is present throughout the environment and levels have increased over time. Because of its toxicity, considerable efforts have been made to find substitutes. Consequently, by 2020 there is expected to be a surplus of mercury in the world. Mercury exposure can cause serious health effects and a key strategy in reducing exposure is reduction in the use of mercury containing products and processes, efficient filtering when mercury or mercury compounds occur as by-products in industrial processes and disposal in a safe way to ensure isolation from man over long time periods.

Mercury and mercury containing waste will always remain toxic and hence are examples of wastes which require long-term safe storage. Because they maintain their toxicity over time, the isolation requirements needed for disposal of pure mercury and its compounds are of similar nature to those needed for disposal of spent nuclear fuel or long-lived radioactive waste from reprocessing.

A3.2.2 Health effects

Mercury has an impact on health on local, regional and global scales. Mercury and its compounds can be highly toxic to humans, ecosystems and wildlife. High doses can be fatal but also relative low doses can have serious adverse impacts to developing nervous system and there are indications of possible harmful effects on the cardiovascular, immune and reproductive systems.

The toxic risks from mercury depend on its chemical form, the manner of exposure, level and duration of exposure and vulnerability of persons exposed. The effects are increased by environmental bioaccumulation and biomagnifications through the food chain, especially through fish. In particular, mercury in the form of methyl mercury is hazardous to both humans and wildlife by ingestion as this compound passes the placental barrier and the blood-brain barrier. Elemental mercury is more toxic by the inhalation pathway.

Human exposure can result from several different pathways. Most important is the intake in food, primarily fish. Fish is an extremely valuable component of the human diet all over the world and mercury can be a major threat to this.

For elemental mercury, inhalation of mercury vapour that is then absorbed by lung tissue is the most important source in unhealthy working environments. To some extent, dental amalgam is another source of vapour. For other inorganic compounds, diet is the main source for exposure.

Many people are exposed to these ingestion and inhalation pathways. Their risks from mercury depend on a range of factors including employment, geographic location and diet, all of which contribute to determining levels of exposure.

Mercury has caused a variety of significant adverse impacts on human health and the environment throughout the world. The Minamata disease in Japan was caused by spilled mercury that converted to methyl mercury and bio-accumulated in fish and seafood that was the main source of food for local people. Around 3 000 people were affected. The case of Iraq mercury poisoning affected more than 6 000 people and was due to consumption of seed that had been treated with fungicides containing mercury.

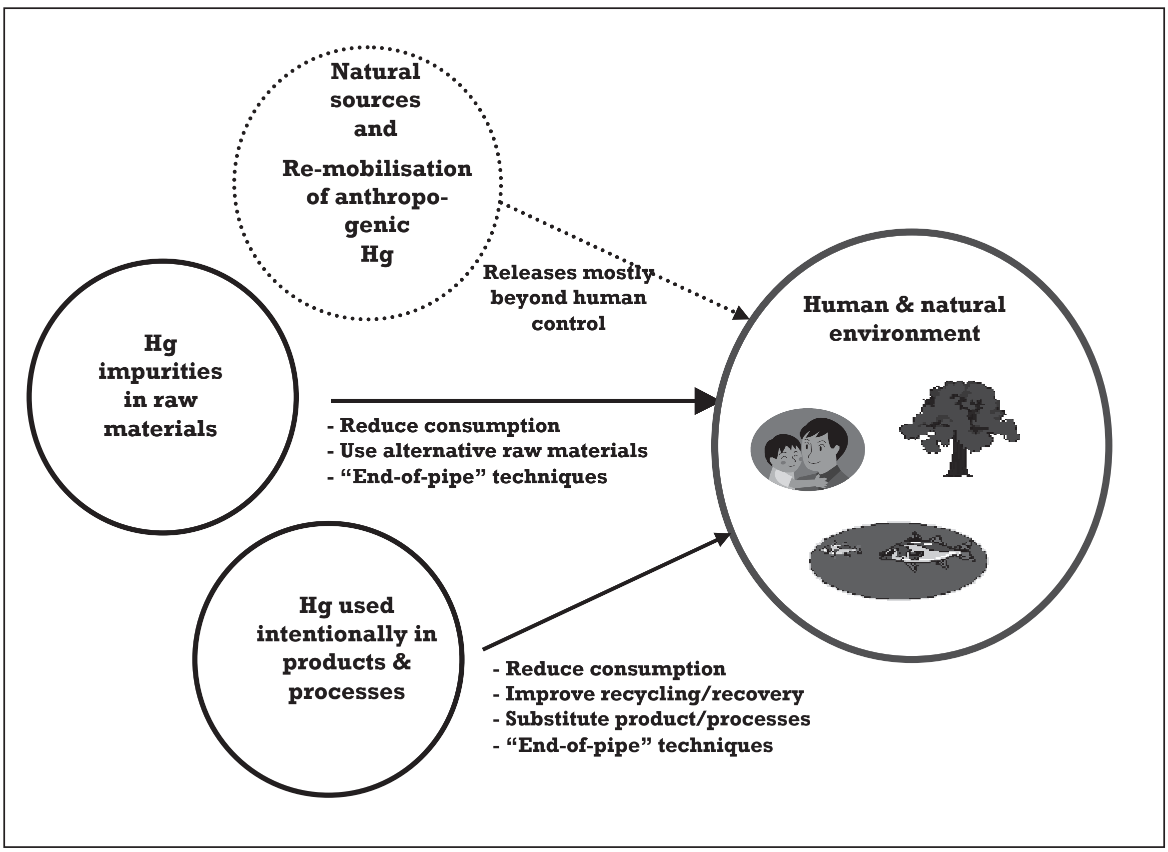
A3.2.3 Sources for releases and exposure

The releases of mercury to the biosphere can originate from several different sources, as shown in Figure A3.2-1:

- natural sources – naturally mobilised from the earth's crust and also emissions from forest fires;
- impurities in raw material – anthropogenic releases related to mobilisation of impurities in fossil fuels, in particular coal, but also in oil and gas and also in the extraction of minerals;
- use of mercury in products and processes;
- re-mobilisation of historic mercury deposited in soil, sediments, water and tailings.

In order to cope with safety requirements over long periods, without the need for monitoring and intervention, the trend for managing long-lived hazardous waste is towards deep disposal. Several countries are developing such facilities.

Figure A3.2-1: Sources of mercury releases to the environment and the main control options



Source: UNEP, 2003.

A3.2.4 Amounts and cycling of mercury in the global environment

Mercury is available in soil and sediment in the ground, in water and air. In nature, mercury will change its properties and consequently participate in a number of biochemical cycles.

Possible routes for intake and damage are connected to its chemical form, methyl mercury being the most hazardous form. The most significant releases of mercury pollution are emissions to the air but mercury is also released from sources related to land and water.

Once released, mercury persists in the environment where it circulates between air, water, soil, sediments and biota in various forms. Thus, emissions add to a mobilised global pool of mercury that is deposited on land and water from where also will be re-mobilised. The time scale for the circulation between the different compartments contributing to the mobilised pool of mercury can be from some years up to thousands of years.

Estimates of the amounts of mercury include 5 000 t of mercury in the atmosphere, another 10 000 t in seas, 400 000 t in inland lakes and sediments and around 1 500 000 t in soil. The annual contribution to the mobilised pool has been estimated as 13 500 t.

A3.2.5 Efforts to reduce mercury releases and exposures

As local releases of mercury cause global problems, mercury is an issue much studied on global, regional, national and local levels. Despite reducing use and releases from industry, the emissions to air are increasing due to increased power production by fossil fuel combustion, especially coal. Artisan small-scale gold mining using mercury is causing huge health problems among native people in Asia, Africa and South America. To avoid damage to man and the environment, many improvements are needed.

Reduction of risks demands:

- reduced use of mercury in mining;
- efficient use of filters and other clean-up plant to avoid releases of impurities;
- collection, treatment and permanent disposal of mercury products and waste.

A3.2.6 Mercury waste – international activities

The United Nations Environment Programme (UNEP, 2003) carries out a comprehensive programme to understand mercury issues and to coordinate actions to reduce risks for humans and nature.

The *Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal* (Basel Convention) (UN, 1989) is the world's most comprehensive agreement on hazardous and other waste and aims to protect human health and the environment from inappropriate management of waste. A programme on mercury waste and its environmentally sound management is being carried out under the Basel Convention. Draft technical guidelines inform the practical steps needed to ensure sound and safe management.

The EU has a strategy and an active programme on mercury striving to reduce emissions and exposure, cutting supply and demand and looking for long-term disposal solutions including the support and promotion of international action such as within UNEP. Proposed legislation includes an export ban outside the EU and matters relevant to storage of surplus mercury.

The legal framework of EU concerns the following issues: (EC, 2003; EEC, 1991; 1999; EU, 2006)

- regulating releases into the environment (Directive 2006/11/EC) on releases to water environment;
- regulating wastes containing mercury (Directive 91/689/EEC) on hazardous waste;
- environmental standards for drinking water and foodstuffs such as fish;
- regulating storage and disposal (Directive 1999/31/EC) and (Decision 2003/33/EC) on disposal;
- a proposal (Regulation COD/2006/0206) on an export ban and for disposal of liquid mercury.

EU members are obliged to transpose and implement EU Directives into their own national legislations.

There is ongoing discussion in the EU aimed at revising (Directive 1999/31/EC) and (Decision 2003/33/EC) to allow future disposal of liquid elemental mercury in underground disposal facilities.

A3.2.7 Management of waste containing mercury

Mercury occurs in society in many forms from a large number of sources. Therefore, environmentally sound management of mercury is, in all respects, a complex task. In some industries, mercury is managed in a well-controlled manner whereas others are much less controlled. A variety of wastes such as gas filtering products, sludge from industrial processes, ashes and mineral residues, including used batteries and dental waste, is nowadays well looked after, at least from a short term perspective. Releases from historic waste, some coal power production and artisan gold mining are examples of areas that need to be improved.

Treatment of waste containing mercury

Hazardous waste, including mercury waste, is treated by a number of methods based on thermal, physical, chemical or biological processes. After collection and identification, the waste is sorted and packed in barrels, industry bags and containers for disposal.

Waste in powder form, materials from filters, sludge and similar products are often stabilised by being mixed with cement or fly ash. Recycling and reprocessing are used for batteries, contaminated soil etc, resulting in mercury in liquid form for storage and eventually disposal.

To dispose of surplus elemental mercury, methods have been developed to stabilise the liquid mercury by mixing with sulphur into a much more stable sulphide. Such products can be disposed of in hazardous waste landfills, on or in the ground, but not in an acid environment.

Disposal technology

Waste containing mercury is disposed of in general to specially engineered landfill, underground in caverns and pits close to the surface and deep underground in stable geological formations.

The bulk of waste containing mercury is disposed of in hazardous waste landfills, although historic waste may appear in many unqualified landfills. The disposal strategy and technology can differ significantly between countries.

For hazardous waste landfills in the EU, see Figure A2.4, requirements for design, safety and operation are stipulated in detailed directives implemented in the environment legislation of the member countries. These landfills require monitoring and control of releases and are therefore not suited to long-term storage where such maintenance cannot be guaranteed.

Landfill in caverns and pits near the surface

Different types of chemical waste have been disposed of in caverns, excavated mine openings and pits and quarries near the surface. These allow better conditions for avoiding long-term leakage than surface landfills. In favourable geological situations, such facilities can be used for long-term safe disposal.

Underground landfills: disposal in deep geological formations

Disposal in deep stable geological formations is currently carried out in chambers situated in 700 m deep salt formations in Germany as shown in Figure A2.7. Several countries see such disposal techniques as the best and safest way available to manage long-lived hazardous waste (such as mercury containing waste). In Germany, large quantities of hazardous wastes – from Germany and some other European countries – are currently being disposed of in four mines.

The trend for the disposal of long-lived hazardous waste is toward such technology. Facilities are being developed in several countries to allow long-term safety without the need for monitoring and intervention.

Sweden was the first EU country, in 2005, to pass legislation requiring deep geological disposal for all waste with mercury content above 0.1%. To meet legislative requirements, Sweden is currently building a disposal facility in granite rock connected to a deep mine.

A3.2.8 Safety assessments

Although the basic principles are the same, the details of safety assessments for chemical and radioactive waste management are in general treated in different ways.

Although a few attempts have been made, there exists no common system to evaluate risks. From the viewpoint of society, it is desirable to judge risks in a way that can be applied to both categories. Some attempts to discuss an "overall risk" have found it useful to separate effects leading to cancer from those that have other serious effects on health. Hazardous waste exhibits a range of characteristics that have serious effects on health, including explosive, flammable, oxidising, poisonous, infectious and toxic. These tend to be "non-cancer" risks. The primary hazard from radioactive waste is exposure to radiation, which can lead to cancer.

However, the boundary is not always clear, as some toxic chemicals can cause cancer and some compounds that are radioactive are also toxic. A primary risk from uranium in drinking water, as an example, is from its toxicity to the kidney.

Management and disposal of waste containing mercury and its compounds is regulated through national regulations for hazardous materials that derive, in general, from EU Directives and Basel Convention statements.

Safety regulation is however focussed on temporary storage and monitored disposal over short time periods – 30 to 200 years. The long-term safety assessments required for final safe disposal of mercury and mercury waste are in general only briefly mentioned in the regulations of most countries.

The EU Directives give requirements and guidance on issues related to geological repositories and requirements on safety assessments for licensing and use. These requirements ask for consideration of waste characteristics, the technology used and in particular the geological properties of the repository. In most aspects, requirements to demonstrate safety are of a similar nature to those stipulated for long-lived radioactive waste. However, the Directives are less detailed on the time periods to be considered, mentioning thousands of years or geological time periods. Safety assessments for the licensed disposal facilities in deep salt mines in Germany deal with the long term by stating that the geological conditions of the salt formation itself provides stability and containment over millions of years.

A3.2.9 Attitudes of the public, politicians and regulators

The public's attitudes and perception of risks are different for hazardous waste and its disposal if the waste has a toxic chemical content or if it is radioactive, see Appendix 4.

However, regulators are active in both areas and requirements on polluting industry and disposal are stringent for both categories of wastes.

A3.2.10 Comparison with radioactive waste

Occurrence, exposure and health effects

Mercury and its compounds are highly toxic and present risks to human health and the environment over long periods that require precautions that are similar in some ways to those needed for long-lived radioactive waste, particularly safe permanent disposal. In both cases, releases are often local but the impacts can be on a global scale if releases are to the atmosphere.

The annual global contribution to the mobilised pool of mercury has been estimated as 13 500 tonnes. To provide a perspective, this amount is in the same order of magnitude as the annual global spent fuel arising from nuclear power plants, which is estimated to be about 15 000 tonnes. However, the hazards from the two waste types are, of course, very different. Mercury mobilised by man is distributed around the globe in relatively small concentrations, but with the potential to affect the health of very large numbers of people. Spent fuel is securely contained in a limited number of locations with the potential to affect only a small number of people, and then only in the event of a very low probability accident.

Safety

Safe management and disposal must be demonstrated in both the short and the long term for waste containing mercury and for radioactive waste. Because mercury is stable it will always be a risk to human health and the environment, and the very long-term scenarios are even more important than for radioactive waste, where decay will eventually reduce the risk (albeit the timescale for the activity in spent fuel to decay to around the level of the original uranium ore is around 100 000 years).

In both cases, regulations regarding tolerable releases (radiation dose, content of mercury in fish/water, etc.) and short-term issues are well established. Compared with the large R&D programmes for the long-term management of radioactive waste, corresponding management of mercury waste is currently less well studied.

Final disposal

Currently, final disposal of mercury waste is carried out in landfills, particularly engineered facilities for hazardous waste, and in stable geological formations, primarily deep salt mines. As the landfills must be monitored and managed the trend is toward disposal in stable geological formation where there is less need for institutional control in the long term. The best examples are salt mines.

State of knowledge

Comprehensive R&D is carried out for management of both radioactive and hazardous waste. However, the level of data collected and resources spent are higher for radioactive waste. Considering the number of chemical substances to be addressed, R&D resources must be directed to a much broader range of problems in the case of hazardous waste and are not primarily directed towards final disposal.

Legislative and regulatory framework

Comprehensive and detailed regulation and legislation exists for the management and disposal of both mercury and radioactive waste.

Regulation concerning mercury waste, by being a part of overall environment legislation, is more general and harmonised on both regional and international levels. On the international level, UNEP and the Basel Convention explore the needs and give recommendations for efficient and environmentally sound management. EU regulation and legislation stipulates requirements for management and disposal within EU. The EU regulation is in turn mandatory for member states and must be implemented in national legislation.

Regulation and legislation on management and disposal of radioactive waste is also based on very active international cooperation but matters are finally decided and regulated in specific national legislation.

A3.3 Potential future management of CO_2: carbon capture and storage (CCS)

A3.3.1 Background

Worldwide concern over human-induced climate change has led to the signing of the Kyoto protocol whereby Governments have made binding commitments to reducing greenhouse gas emissions. In addition, the introduction of carbon trading provides an economic stimulus to reduce fossil fuel usage. Governments are pursuing a number of parallel policies in their attempts to fulfil their Kyoto obligations. These include energy conservation and subsidies to producers and users of renewable energy devices. Governments are also investing in research into ways of reducing the carbon footprint of the more traditional means of electricity generation, especially the burning of coal and other fossil fuels. Foremost amongst the proposed solutions is carbon capture and storage (CCS, Figure 1). This technology will necessarily impose penalties in terms of additional cost and additional energy usage. As with new-build nuclear power, critics argue that it is a distraction from the need to invest in the development of renewable energy sources.

In line with current practice in the carbon capture and storage business, the word "storage" is used throughout this section of Appendix 3. It is interesting to note the contrast with the terminology used in radioactive waste management where "storage" always implies an intention to retrieve and where, if there is no intention to retrieve, the word "disposal" is used. Similarly, in carbon capture and storage, CO_2 is never referred to as "waste" – another difference from radioactive waste management perhaps recognising that, when used for enhanced oil recovery, it is a useful product. Enhanced oil recovery, a process whereby CO_2 is injected into diminishing oil reservoirs to boost production, has been in routine use for more than 30 years.

A3.3.2 Sources and amounts of current release

The International Panel on Climate Change (IPCC, 2007) states that emissions of the greenhouse gases covered by the Kyoto Protocol were 49.0 Gt of CO_2-equivalent (eq.) in 2004, an increase of 24% since 1990. The largest fraction (29 Gt) was from carbon dioxide (CO_2) itself. Electricity generation is by far the largest and fastest growing source of CO_2. Around 40% of global primary energy was used as fuel to generate 17 408 TWh of electricity in 2004 with about 67% of this being fossil fuelled.

IPCC (2007) estimates that, when applied to both coal- and gas-fired electricity generation, CCS could result in a 0.81 Gt CO_2 eq. total reduction in greenhouse gas emissions by 2030. This is broadly similar to the figures for hydro and wind (0.87, 0.93 Gt CO_2 eq. respectively). Emission reductions from applying CCS to coal-fired generation are estimated to be 0.49 Gt CO_2 eq. IPCC estimates that nuclear energy could reduce emissions by a further 1.9 Gt CO_2 eq. beyond the 1.7 Gt CO_2 eq. already anticipated by reference to IAE's World Energy Outlook 2004. (IEA, 2004a)

A3.3.3 Carbon capture

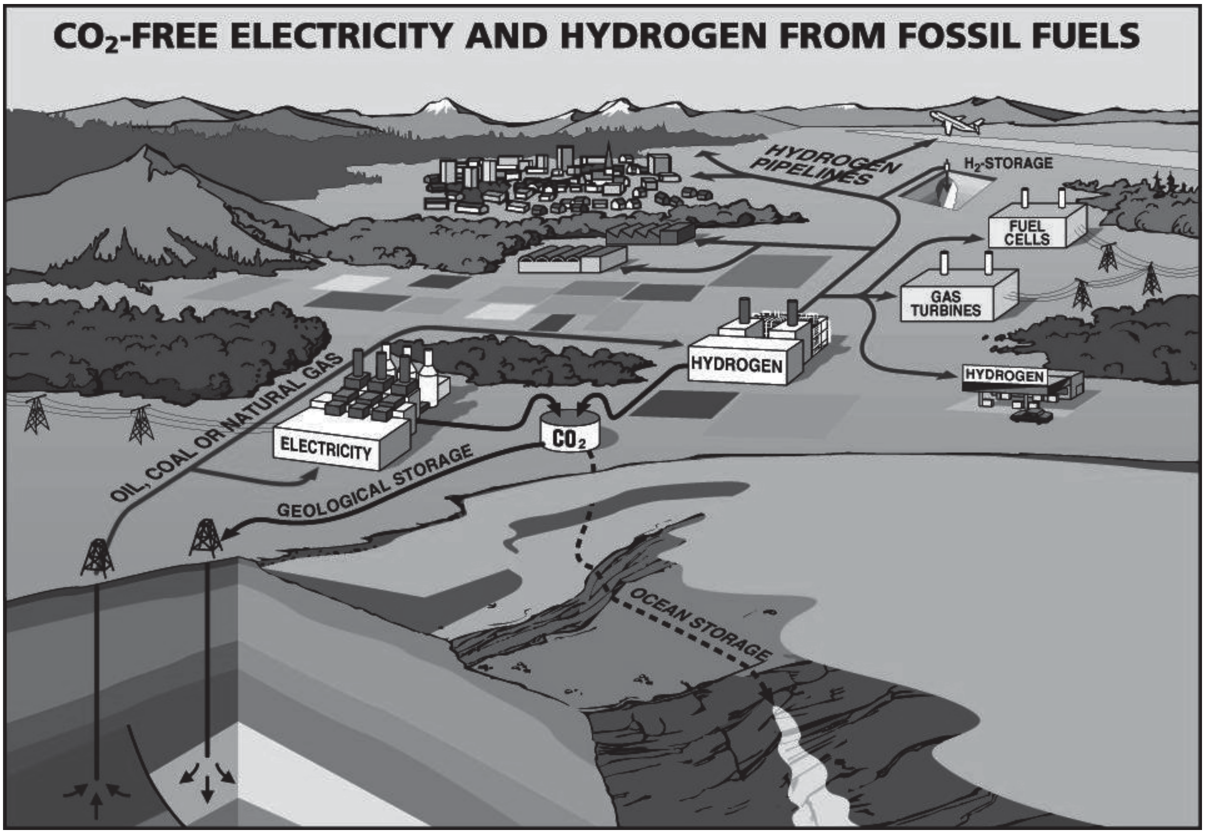
Carbon capture (IEAGHG, 2007) requires a very significant investment so that the technology is only suitable for large producers of CO_2. Primarily, these are fossil-fuelled electricity producers (emitting 10.5 Gt CO_2 per annum) and, to a lesser extent, cement manufacture, refineries, steel production, etc. (IPCC, 2005) A single 1 600 MW lignite-fuelled power station emits around 10 million tonnes of CO_2 per year. (Vattenfall, 2008)

CO_2 capture technology can be deployed to good effect with combined cycle gas turbine plant. CCGT have high thermal efficiency and may burn either natural gas or hydrogen and carbon monoxide produced from coal. The fact that the fuel is, or is made to become, gaseous allows the possibility that CO_2 may be captured either before or after combustion.

The pre-combustion method is used with coal-fired CCGT where, in the absence of CO_2 capture, proprietary compounds such as Selexol are is used to remove sulphur oxides from the H_2 and CO gas mix prior to combustion. These compounds will also remove CO_2 although, in a normal coal-fired CCGT, this is an unwanted reaction. If CO_2 capture is wanted, however, oxidation of the coal during gasification is allowed to go a little further to produce hydrogen and CO_2 so that the latter may be removed.

Most conventional coal power plants burn pulverised coal and would, therefore, need post-combustion capture technologies. The UK government, for example, is specifically supporting this option because of its application to China and other emerging economies with large numbers of conventional coal power plant. There are two post-combustion methods. In the first, the CO_2 is removed from the flue gas by means of a chemical or physical reaction. Most often, proprietary organic compounds (based on amines) react chemically with the CO_2 and are then regenerated by reaction with steam. CO_2 can then be cooled, dried and pumped away. A complication with this method is that steps must be taken to remove the oxides of sulphur and nitrogen so that they cannot react with the organic chemicals. If they do, they will form stable products that prevent the organic compounds from being regenerated.

Figure A3.3-1: Outline scheme illustrating carbon-free electricity generation from fossil fuels using terrestrial or marine-based geological storage



Source: IEAGHG, 2007.

The second form of post-combustion CO_2 capture is known as oxy-combustion. This, again, may be used with conventional pulverised coal plant if the coal is burned in pure oxygen. The oxygen is produced on-site using an air separation plant. The flue gas consists almost entirely of water and CO_2 so that post-production processes can be conducted with higher efficiency. A possible offset against the cost of air separation is the fact that the flue gas may need little cleanup. This is because sulphur oxides are removed with the CO_2 and burning in oxygen results in the flue gas having low levels nitrogen oxides. Note, however, that the pilot CCS plant at Spremberg in Germany does have flue gas desulphurisation.

A3.3.4 Principles of CO_2 storage

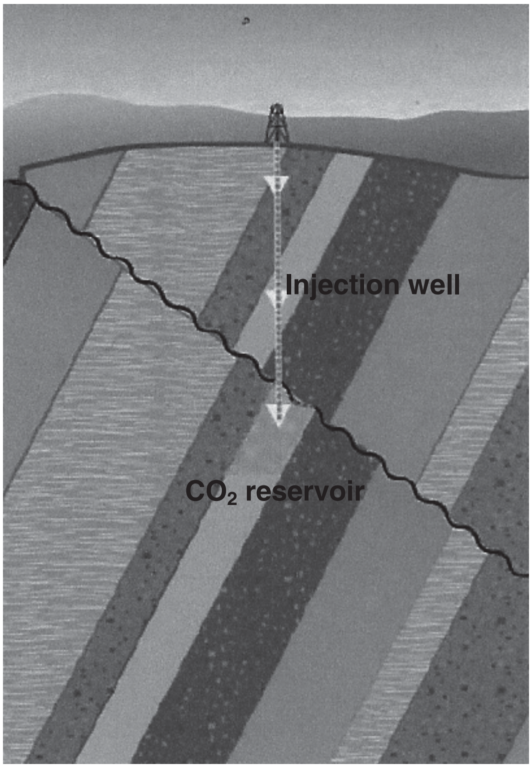
All current underground storage designs aim to store the CO_2 at a depth of greater than 800 m because these depths produce a pressure at which CO_2 exists in a supercritical state. (IEAGHG, 2008a) A supercritical state is one in which the material is neither liquid nor gas but, rather, behaves like both. The advantages are twofold: there is a volume reduction (compared to the gas at room temperature and pressure) of at least 200 times and the supercritical CO_2 can flow easily (like a gas) into the pore spaces between mineral grains in the host rock.

Using natural gas fields as an analogue, the general argument is that rock formations are capable of containing gases for millions of years. Mechanistic explanations are available that explain how the gas comes to be trapped and why there is reason to believe that trapping will be permanent (see Box 1).

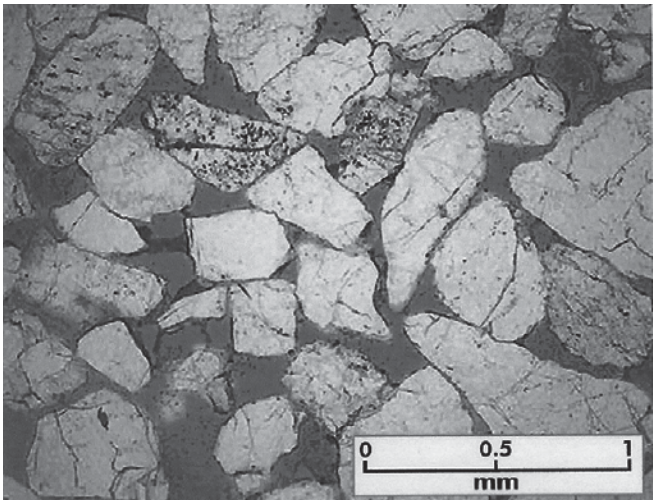
BOX 1: Trapping mechanisms

Trapping of CO_2 occurs by four different mechanisms (IEAGHG, 2007):

- stratigraphic/structural;
- residual;
- solubility;
- mineral.



Stratigraphic and structural trapping refer to large-scale geological features that allow gas or liquids to be trapped underground. Almost invariably, this arises because an impermeable formation lies above a reservoir formation as a result of the stratigraphy or as a result of some disturbance to the stratigraphy due to faulting (see figure right).



During CO_2 injection, the applied pressure must be high enough to allow the CO_2 (which appears blue in the figure left) to displace formation fluids (e.g. water or oil) from the rock pores. At the same time, the pressure must not be not so high as to break the stratigraphic or structural seal. When injection stops, the pressure drops and the surrounding fluid moves back into the pores (propelled by capillary action), trapping the CO_2 – this is known as **residual** trapping.

CO_2 may then dissolve in the water (**solubility** trapping) forming a more dense fluid that may slowly sink through the formation. Over thousands of years, the dissolved CO_2 may react with the surrounding minerals to form solid products (**mineral** trapping). The timing of these processes means that CO_2 trapping becomes more secure with time, and hence the risk of leakage decreases with time. (IPCC, 2005)

Source of images: CO2CRC.

A3.3.5 Cost of CCS

CCS places additional energy demands, principally from separation and compression. Depending on the type of plant and the nature of the fuel, a power plant equipped with CCS would need roughly 10-40% more energy than an equivalent plant operating without CCS. The additional energy requirement will itself produce CO_2 and the net result is that a power plant with CCS should reduce CO_2 emissions to the atmosphere by approximately 80-90% compared to a plant without CCS. (IPCC, 2005)

Figures presented by IPCC (IPCC, 2005) indicate that carbon capture alone increases the cost of electricity by:

- 1.8 to 3.4 US$ct per kWh for a pulverised coal power plant;
- 0.9 to 2.2 US$ct per kWh for an integrated gasification combined cycle coal power plant;
- 1.2 to 2.4 US$ct per kWh for a natural gas combined-cycle power plant.

Transportation and storage would add between -1 and +1 US$ct kWh^{-1} and about half this for gas plants. The negative figure recognises the revenue that would arise if CO_2 were used for enhanced oil recovery. If we (i) ignore the highest capture costs (for a pulverised coal plant); (ii) assume that transport and storage are cost-neutral; and (iii) take a mean wholesale cost of electricity of 4 US$ct per kWh, these figures represent a percentage increase in the cost of electricity of between 22 and 60%.

A3.3.6 Suitable geological formations

According to the IPCC, the potential storage capacity in geological formations worldwide far outstrips the likely demand. The main requirements of a CO_2 storage site (using the standard industry terminology) are: (IEAGHG, 2008a)

- accessibility – a geological formation that is accessible by borehole;
- capacity – the ability to hold useful quantities of gas;
- injectivity – the speed with which the formation can receive gas;
- storage security – leak tightness of the formation.

Many geological formations are thought to meet these needs but the current front runners are:

- depleted oil and gas reservoirs;
- deep saline formations;
- un-mineable coal seams.

Depleted oil and gas fields will probably be the first sites to be used for CO_2 storage because of their known location, their known properties, their availability and the greater certainty with respect to the underlying science. CO_2 injection is already used as a means of enhancing oil and gas recovery and it is possible that such enhanced recovery could be a means of offsetting the cost of storage. On the other hand oil and gas fields will not usually be located close to the CO_2 production sites and there may be concerns that abandoned wells may not have been sufficiently well sealed to ensure leak-tightness.

In the longer term, the extremely wide distribution of deep saline formations will probably allow them to constitute the majority of CO_2 disposal sites. A possible limiting factor is that these formations may not always occur at a convenient depth: either too deep, which will increase cost, or too shallow, which will not allow CO_2 to reach the supercritical state. This type of geology should have good long-term retention properties for CO_2 although stratigraphic/structural trapping (Box 1) may not always be as obviously present as it is for former oil or gas reservoirs. Abandoned wells are less of an issue than for former oil and gas reservoirs but, still, cannot be wholly dismissed.

Un-mineable coal seams are a more distant prospect: it is known that coal can hold significant quantities of gas in micropores but the mechanisms are imperfectly understood at present. An advantage of these formations is that the cost of injection could be offset if the CO_2 displaced methane, which could then be extracted for use as fuel.

A3.3.7 Pilot projects

As already noted, CO_2 is routinely injected into oil reservoirs for the purpose of enhanced oil recovery. Typically, natural gas (methane) is pumped to an installation where it is partially oxidised or "reformed" to create hydrogen and CO_2. The CO_2 is then separated and pumped to an oil well whose production is diminishing. The CO_2 boosts oil production by displacing oil from the reservoir formation. These arrangements appear to form the basis of many of the 50 or so completed, ongoing or planned pilot projects for CO_2 storage worldwide. (SCCS, 2008) Three projects are particularly noteworthy for their size. The Weyburn-Midale CO_2 storage and monitoring project in Canada injected more than 5 Mt of CO_2 into a depleted oilfield. The CO_2 is supplied from a coal gasification plant in North Dakota, United States. An extensive monitoring network failed to detect any leakage. In the Sleipner project, 10 Mt of CO_2 have been injected into a deep saline formation off the Norwegian coast. (IEAGHG, 2008a) The Krechbah processing plant in Algeria has, since 2004, re-injected 1.2 Mt CO_2 per year into the gas field it came from.

There appears to be only one operational project that is attempting to demonstrate both carbon capture and storage. This is a 30 MW(e) coal-fired oxy-combustion plant near Spremberg in Germany. CO_2 is collected, compressed and trucked 350 km to an empty gas field for injection. It is expected that 100 000 t of CO_2 will be injected over 3 years. The plant has been funded by Vattenfall (the Swedish power generator) at a cost of 70 M . Interestingly, the flue gas is cleaned to remove sulphur dioxide and fly ash. Other projects are being proposed and their feasibilities investigated around the world. In particular, the EU ZEP programme (Zero Emission Fossil Fuel Power Plants) (EU, 2008) aims to have up to 12 large scale CCS projects operational by 2015 so as to demonstrate commercial viability by 2020.

A3.3.8 Risk assessments

Risk assessments are used in the oil industry to demonstrate the safety of CO_2 injection for enhanced oil recovery. Increasingly, methodologies developed for radioactive waste disposal are being used to assess long-term effects. For instance, assessments commonly use base (normal) and alternative scenarios to address possible future states of the storage and the surrounding environment. Similarly, standardised lists of features, events and processes (FEPs) may be used for auditing assessments and there is frequent reference to natural analogues and site-specific analogues such as groundwater residence times. Box 2 describes the approach to, and the lessons drawn from, risk assessment in the Weybourn project. (IEA, 2004b)

The unresolved issues identified in the Weybourn project are characteristic of safety assessment in radioactive waste disposal: typically, they hinge on the need for the assessment model properly to represent the disposal environment and, in particular, for the model to explain the characteristic features of the host rocks.

Other similarities include the need to assess seismicity and vulcanism, geochemical effects (including the action of CO_2 on repository seals) and the effects of minor constituents on repository behaviour.

IEAGHG (2008a) points to the many monitoring techniques available to verify the amount of CO_2 injected and the integrity of the storage. As with radioactive waste disposal, monitoring is of limited use when attempting to verify long-term containment but this is less of an issue in Carbon dioxide Capture and Storage (CCS) because leakage is most likely during or soon after injection so that CO_2 storage becomes more secure with time. Consequently, IPCC guidance, London Dumping Convention, OSPAR Treaty and EU CCS Directive all allow monitoring to decrease with time and cease if all evidence indicates secure storage.

A3.3.9 Regulation

CCS is a new technology and regulation is evolving. The IPCC special report on CCS states that: (IPCC, 2005)

> "Existing laws and regulations regarding inter alia mining, oil and gas operations, pollution control, waste disposal, drinking water, treatment of high-pressure gases and subsurface property rights may be relevant to geological CO_2 storage. Long-term liability issues associated with the leakage of CO_2 to the atmosphere and local environmental impacts are generally unresolved."

According to Vattenfall (2008), responsibility for post-injection monitoring (and, presumably, remediation, if monitoring found something untoward) could rest with the operator, the government, a third party brought in for the purpose or any combination of these. As with radioactive waste disposal or abandoned mines, governments will invariably be the long-term guarantors of safety. The key issue for operators (for which read investors in CCS) will always be the duration of the operator's responsibility.

An essential precondition for development of CCS is the ability to profit from reduced CO_2 emissions. The IPCC Greenhouse Gas Inventory Guidelines (2006) provide a methodology for assessing the effect of CCS on greenhouse gas emissions, thus enabling countries to report emissions reductions in their inventories from CCS, and providing the basis for its inclusion in emissions trading schemes. The EU Greenhouse Gas Emission Trading Scheme (ETS) started allowing trading in CCS emission reductions in 2008.

With respect to sub-sea storage of CO_2, the London Dumping Convention and its 1996 Protocol applies; the parties to the Protocol agreed in 2006 to permit sub-seabed storage of CO_2. OSPAR did the same in 2007.

BOX 2: Risk assessment

The risk assessment performed for the Weyburn project addressed five possible release scenarios (IEAGHG, 2007).

1. *Rapid "short-circuit" release* (via fracture, borehole, or unconformity). Typically, short circuit releases would cause acute environmental or health effects such as might be produced by high concentrations of CO_2 in low-lying areas on the surface. The presence of unknown or poorly sealed wells penetrating into the storage formation is generally considered to be the most important release pathway.
2. *Potential long-term release*. Long-term releases may be impossible to measure but are important because they determine the overall effectiveness of CCS.
3. *Induced seismic event*. Induced seismicity was first seen in the 1960s at some underground storage sites for natural gas. Raised gas pressure allows small movements (micro-seismicity) along active faults. Since then storage sites have aimed to avoid active faults but even so, it is necessary to have an understanding of the process and to know, for example, how high the gas pressure needs to be to trigger such an event.
4. *Disruption of host rock*. As with the induced seismic event, it is important to understand how gas pressures might cause failure of the sealing formation and to know how large the gas pressure needs to be to cause such an effect.
5. *Release to aquifer*. This is an important issue not least because regulations are often framed in terms of maintaining groundwater quality. Risks to shallow water aquifers may arise from acidification, unwanted mineralogical effects and upwards displacement of briny waters.

As a result of the assessments, issues requiring further development were identified. These include:

- the use of more direct monitoring to demonstrate effective storage;
- more effective use of existing seismic data;
- determine the fate of gaseous impurities: H_2S and mercaptans;
- characterise conductive natural fractures in strata overlying the reservoir (if they exist) and their flow properties;
- obtain core samples to determine mechanical properties of any weakened overlying/underlying strata and properly preserve;
- assess the impact of fractures on seismic images (anomalies may be due to more than the presence of CO_2);
- in long-term fate assessment, account for additional mechanisms that may dissolve reservoir rock or minerally fixate CO_2 (e.g. dissolution due to convective mixing) and perform sensitivity analyses for various long-term assessment models.

A3.3.10 Attitudes of public, governments and regulators

The IEA acknowledges (IEAGHG, 2008b) that public acceptance will be needed if CCS is to progress and IPCC frequently mentions its importance. (IPCC, 2005) Few public opinion surveys have been conducted (Tokushige, *et al*, 2007) and these few have not been given wide publicity. Neither of the two largest CO_2 storage projects (Weyburn and Sleipner) have public acceptability as part of their remit. Given that CO_2 injection is already used as a standard method of enhanced oil recovery, it is possible that the CCS industry considers that public acceptability is unlikely to be a "show stopper". Anecdotal evidence from Spremberg (a coal mining town), where the pilot CCS plant is located suggests that the public broadly approves of the project with comments like "It's bound to bring jobs, that's what matters, but if it makes us famous for saving the world, that would be cool". (Smith, 2008)

Green groups vary in their view of CCS. Friends of the Earth International (FoE) classes CCS and nuclear energy alike: as "unsustainable technologies" (FOEI, 2005), though some national FoE groups may be more accommodating in their approach. Greenpeace International opposes the application of CCS to coal-fired power stations as a means to combat climate change. (Greenpeace International, 2007) WWF is in favour of CCS, but does not support the Clean Development Mechanism (CDM), an arrangement under the Kyoto Protocol that allows certain countries to invest in projects that reduce emissions in developing countries as an alternative to more expensive emission reductions in their own countries.

Governments face a dilemma: increasing domestic demand for electricity coupled with a need (or even binding commitments) to reducing CO_2 emissions. It is clear that no single measure, whether energy saving, renewable electricity sources or nuclear power will solve the problem. In this situation, governments will aim to adopt a wide range of measures in parallel; these measures will include CCS. President Bush, for instance stated in 2001: "We all believe technology offers great promise to significantly reduce [greenhouse gas] emissions – especially carbon capture, storage and sequestration technologies."

As one might expect, regulators appear to be content to regulate CSS provided that they have the necessary powers and funding. It is clear that many regulators are informing themselves about CCS and (presumably) assessing the need for new regulations. US EPA say that it aims to ensure that geological sequestration does not endanger underground sources of drinking water. The US regulations cover well siting, well construction, well operation, and well closure and there have been over 800 000 regulated wells injecting a variety of fluids over the past 30 years. The EC DG Environment has proposed a Directive to create an enabling legal framework in the EU and to remove existing regulatory barriers.

In responding to the UK Government announcement of new coal-fired power stations, the Environment Agency (responsible for waste disposals in England and Wales) goes further stating that: "new and replacement coal-fired power stations should only be permitted where they are capable of capture and storage of carbon dioxide"; and "the Environment Agency can help to assess all new plant, subject to an appropriate role and funding".

A3.3.11 Discussion and conclusions

General differences and similarities with radioactive waste disposal

The main differentiating feature between radioactive waste disposal and CCS lies in the nature of the disposed material. In the case of CCS, the stored CO_2 is simple chemically but complicated physically since it may exist as a liquid, a gas or neither (i.e. it may be a supercritical fluid) and these different phases may be simultaneously present in different parts of the storage system. It also has very high volume. The phase changes make the system difficult to model, and the large volumes have the potential to affect the evolution of the system. For radioactive waste disposal on the other hand, the waste inventory may be complicated chemically but it is predominantly composed of solid material. Furthermore, the overall waste volumes are relatively small and radionuclides are present only in trace amounts so that, with the possible exception of alkaline plumes emanating from cement-based repositories, radioactive waste disposal does not greatly affect the natural evolution of the system.

Another point of difference is that, in general, emplacement of solid radioactive wastes is intended to be performed in underground facilities whereas CCS is intended to be performed from the surface using boreholes. Both technologies have advantages and disadvantages: disposal from the surface will clearly be cheaper but it will also hinder detailed characterisation of the repository host rocks both in their natural state and in post-injection.

In searching for a suitable site, there are, once again, similarities and differences. Both technologies would try to avoid seismically and volcanically active areas. Both would also aim to understand the evolution of the site so that the past might be used as a guide to the future. However, whereas radioactive waste disposal usually aims to combine engineered and natural barriers to contain the radionuclides in the waste, CCS uses only natural barriers. So, for instance, a repository for spent nuclear fuel may place the spent fuel inside steel or copper canisters while a repository for intermediate-level wastes may use large quantities of concrete. With the exception of the seal to the injection well, a geologic storage for CO_2, would not use such methods.

Another possible difference is that radioactive waste disposal would generally try to avoid so-called "complex sites". This may not be an option for CCS given the large number of sites needed and, indeed, the geology of some pilot project sites may be regarded as complex (e.g. Weybourn).

Safety assessments

In developing appropriate risk assessments, CCS appears to have borrowed widely from safety assessment methodologies for radioactive waste disposal. Consequently, we find familiar approaches such as the use of scenarios to encompass possible future states of the repository and its surroundings; standardised lists of features, events and processes (FEPs); and natural analogues.

In assessing long-term impacts, radioactive waste disposal generally has very well defined calculational end points that are directly derived from numerical limits and constraints imposed by regulators. An example is the annual radiation dose to an exposed individual that can be traced back to documents such as the Basic Safety Standards. (IAEA, 1996) It seems that there is no such universally adopted measure of health detriment for CCS risk assessments but, rather, a wide range of human and environmental safety issues that are not always precisely defined.

Indicative costs

Accurate cost estimation is difficult and the simplest method, perhaps, is to compare the additional costs of disposal in terms of the premium that needs to be placed on the cost of electricity generation.

In the case of radioactive waste disposal, the cost probably ranges between 5 and 10% of the cost of electricity. As described above, the add-on costs of CCS range between 22 and 60% mostly depending on the type of plant.

State of knowledge

The US DOE (2008) calls for further work to show that CCS:

- is effective and cost-competitive;
- provides stable, long term storage; and
- is environmentally benign.

Examining these in turn, US DOE states that using present technology, sequestration costs are in the range of 100 to 300 USD/ton of carbon emissions avoided. The goal of DOE's programme is to reduce this to 10 USD or less by 2015.

Storage of natural gas in underground formations has been practised for around 100 years while CO_2 injection for the purpose of enhanced oil recovery has been performed for almost 40 years. From these it is clear that CO_2 can be stored in deep underground formations without detectable losses over these timescales. It seems, however, that the accuracy of the measurements is not sufficiently high to provide confidence for CO_2 retention in the long term – evidence for this is more general, coming from natural analogues. In developing a methodology to allow specific CCS schemes to claim credit under the Kyoto Protocol, the IPCC has made allowance for this uncertainty. (IPCC, 2006)

The final issue, environmental safety, is discussed above.

Legislative framework

As noted above, some countries already have regulations controlling CO_2 injection for enhanced oil recovery. No doubt, these will form the basis of regulations that address long-term retention of CO_2 also. In the long term, only governments can bear the liabilities that might accrue from failure of CO_2 storage. The crucial issue for operators and investors in CCS is the timing of the changeover from a private to a public liability.

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Appendix 4

RISK AND PERCEIVED RISK

A4.1 Introduction

For almost all activities in society risk, and how risk is perceived, are important considerations for decision making by governments as well as by industries and consumers. Societal acceptance of risk depends not only on scientific evaluations, but also on perceptions of risk and benefit. Perceptions of both risks and benefits must be considered when seeking to understand what drives social risk acceptance behaviours and why some interventions are more acceptable and successful than others. (WHO, 2002)

In the Executive Summary of the IAEA report on the "Global public opinion on nuclear issues and the IAEA: final report from 18 countries" (Globescan, 2005), it is stated that:

> "While majorities of citizens generally support the continued use of existing nuclear reactors, most people do not favour the building of new nuclear plants".

In the light of the retirement of almost the entire current nuclear power plant fleet by the year 2050, this worldwide public perception will have a large impact on the construction of future electricity generating capacity (both replacing old and building additional capacity). One important aspect in the public's reluctance to accept new nuclear power is the production of radioactive waste during the plants' lifecycle and the industry's perceived lack of capability to manage it.

Radioactive wastes are a danger to human health and the environment if not properly managed. Today, the siting of radioactive waste disposal facilities does not depend only on resolving technical matters, but also requires public values and concerns to be addressed, because the public (at the local or national level, or sometimes both) may have a low acceptance of such facilities. However, there are many examples of hazardous wastes (including wastes with toxic and biohazard characteristics) being safely disposed over many decades. This demonstrates, at least in principle, that safe disposal of inherently dangerous substances can be achieved, provided that there is public acceptance to support the construction of properly designed disposal facilities.

Nonetheless, there is ongoing debate all over the world regarding the disposal of hazardous and radioactive wastes. Disposal site selection is based on many factors including waste and site characteristics, national and regional laws and regulations, and public acceptance. The public acceptance factor plays an increasing role in the decision-making procedure. This factor depends heavily on whether the public believes that they or their environment will be harmed by the proposed new disposal facility – they have an intuitive view of whether the facility will be risky. The public perceives and judges the acceptability of risk differently from experts in the field who see riskiness as synonymous with expected annual mortality.

This appendix seeks to provide a broad perspective on perceived risk, a vital issue to understand, if new waste disposal facilities are to be built for either radioactive or hazardous wastes.

A4.2 Risk

Risk can be defined in a number of ways. The WHO's *World Health Report 2002* defines risk as: (Short, 1984)

"A probability of an adverse outcome, or a factor that raises this probability."

The US Agency for Toxic Substances and Disease Registry defines risk as:

"The probability that something will cause injury or harm."

Science and engineering typically define the risk associated with a particular adverse event as the product of the probability of the event and the magnitude of its consequence. (Rayner and Cantor, 1987) This definition can be applied to a waste disposal facility.

Thus, for a defined event: R = P x C, where:

- R is the risk from the event (typically risk of death per year);
- P is the probability of the event occurring (typically expressed per year);
- C is consequence (typically expressed as the likelihood of death per event).

An aggregated risk can be determined by adding the risks from the internal and external events and processes (which should be independent) that may adversely impact the facility. Internal events are those whose probability can be controlled by design and operation (such as failure of engineered barriers); external events are those over which the designer and operator has no control (such as seismic events).

In this report when talking about "risk" or "actual risk", we mean the scientific definition described above. This defines risk in an objective manner appropriate for engineering calculations, and particularly for assessments comparing potential environmental detriment. However, this definition does not represent the degree of risk that affected individuals might feel. This is known as "perceived risk". Perceived risk is subjective and depends on both the actual risk and a number of individual and societal risk perception factors that are discussed below.

A4.3 Risk perception

The decision-making process for any proposed infrastructural project, whether it is a new road, airport, nuclear power plant or waste disposal facility, will (consciously or not) involve a judgement about risk by all the stakeholders involved. In general, for a range of reasons, stakeholder judgements are made based on perceived rather than actual risk. This in turn directly influences their acceptance level for the proposal (as well as for example, in the case of a road or airport, noise levels). How stakeholders' perceptions of risk are acknowledged affects the level of trust they place in the project developers and in their elected representatives. An additional problem with nuclear facilities is that stakeholders do not necessarily have sufficient personal experience to form a judgement on whether safety criteria are acceptable, especially when they are presented as numerical risk.

Risk perception in this report is defined as the public's subjective assessment of the probability and consequences of a specified type of accident.

Risk perception for a specific activity can be considered in terms of a set of risk perception factors. (Sandman, 1991 and 1993) These are shown in Table A4.1. These factors indicate that an activity like driving a car is likely to have a lower perceived risk because it is voluntary, under the driver's control, familiar, has clear benefits and the process is well understood. The reverse is, in general, true for a proposal to site a radioactive waste disposal facility close to someone's home: the perceived risk is higher because the facility is not under the person's control, is not familiar and, importantly, the person sees that he is being involuntarily exposed to what he regards as a hazard. Of course, on a scientific basis driving has a higher risk than does living close to a radioactive waste disposal facility. However, this is not what is perceived and does not correspond with the level of acceptance.

Table A4.1: Some risk perception factors

Risk perception factor	Perceived risk of an activity will be greater when the activity is seen as:
Volition	Involuntary or imposed
Controllability	Under the control of others
Familiarity	Unfamiliar
Equity	Unevenly and inequitably distributed
Benefits	Having unclear or questionable benefits
Understanding	Poorly understood
Uncertainty	Relatively unknown or having highly uncertainty
Dread	Evoking fear, terror or anxiety
Reversibility	Having potentially irreversible adverse effects
Trust in institutions	Requiring credible institutional response
Personal stake	Placing people personally and directly at risk
Ethical/moral nature	Ethically objectionable or morally wrong

An early study (Slovic, 1987) compared the perceived risk from different societal activities by analysing responses from a range of different groups in the United States. His results are presented in Figure A4.1. In this figure, "Dread risk" is defined at its high end as perceived lack of control, dread, catastrophic potential, fatal consequences or the inequitable distribution of risks and benefits. Nuclear weapons and nuclear power score highest on the characteristics that make up this factor. "Unknown risk" is defined at its high end by hazards judged unobservable, unknown, new or delayed in their manifestation of harm. Chemical technologies score particularly high on this factor. The further an issue moves towards the upper right-hand corner of the figure, the more sensitive the issue is for the public. Events related to such activities will trigger intense mass media attention.

The public perception of risk is closely related to the position of the hazard along the dread risk axis. The higher the dread risk, the more the public wants to see risks reduced and strict regulation imposed to achieve this reduction. In contrast, experts' perception of risk are not related to dread or unknown risk. Instead, experts see riskiness as synonymous with expected annual mortality. As a result, conflicts over risk result from experts and the public having different definitions of the concept.

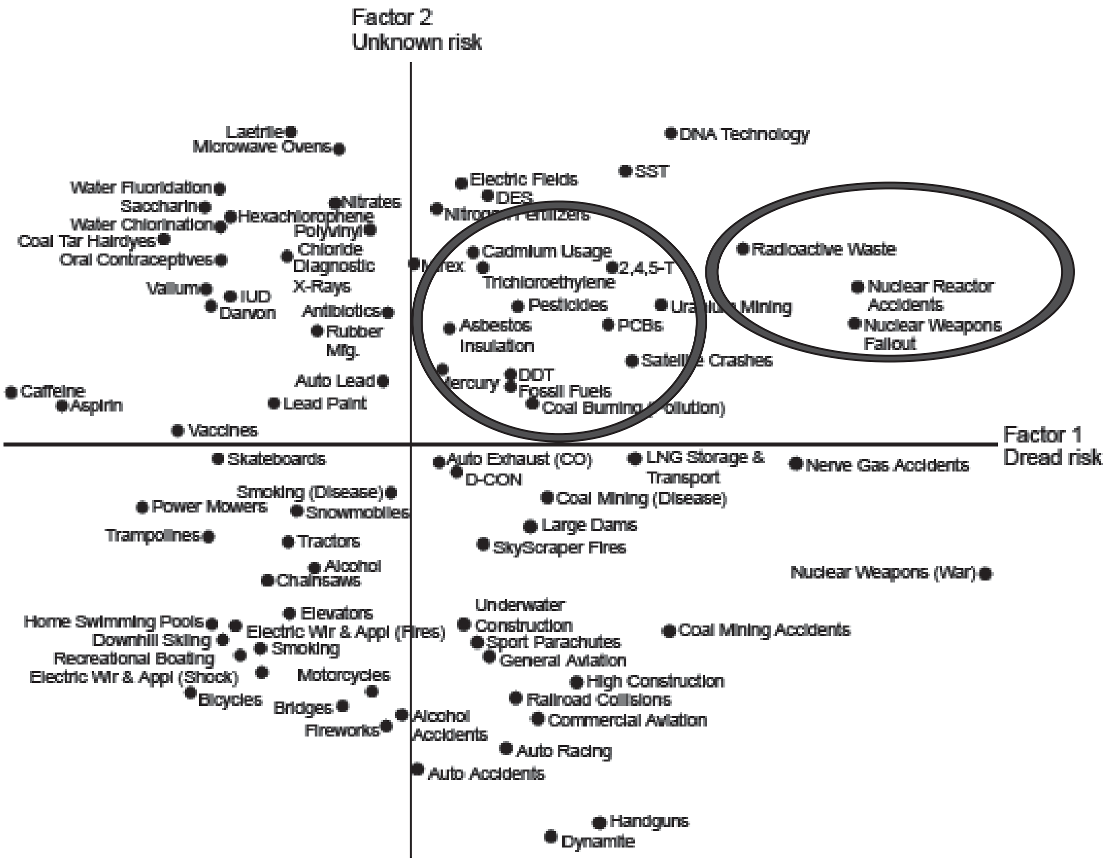
According to Slovic, an explanation for radioactive and hazardous waste having a high perceived risk is:

> "The rapid development of chemical and nuclear technologies which has been accompanied by the potential to cause catastrophic and long-lasting events."

He also stresses that the mechanisms underlying these complex technologies are unfamiliar and incomprehensible to most citizens.

Slovic also noted that making a set of hazards more or less specific (for example partitioning nuclear power into radioactive waste, uranium mining and nuclear power plant accidents) has little effect on risk perception of either the part or the whole.

Figure A4.1: The relationship between perceived knowledge and fear



Source: Slovic, 1987.

A4.4 A perspective on the difference between perceived risk and actual risk

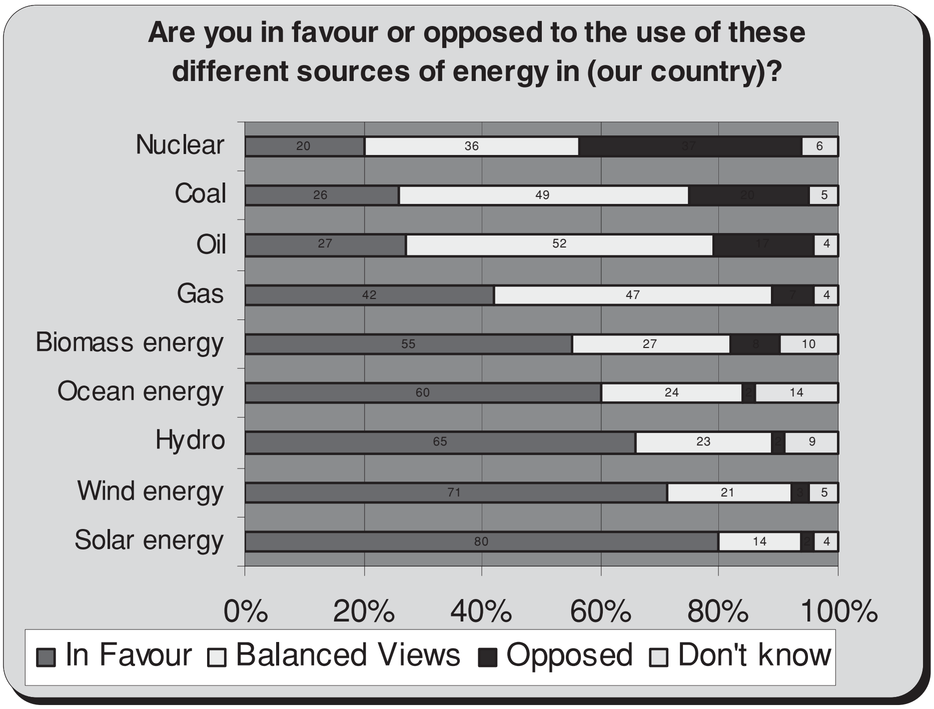
Background

This section aims to provide a broad perspective on the difference between actual risk and the public's perception of risk. This is been done by comparing the consequences of severe accidents in the energy sector with public attitudes and risk perceptions.

Figure A4.2, reproduced from a Eurobarometer survey, shows that nuclear power is the public's least favoured way to produce electricity, with only 20% in favour. People are less opposed to fossil generation, with 42% in favour of gas generation. A significant majority – 65% – is in favour of hydroelectricity generation.

It is judged here that these attitudes to nuclear power are partly shaped by perceptions of risk; the data gathered in this poll also show that perceived levels of knowledge and personal experience of nuclear energy have an impact on views about nuclear energy. This judgement is broadly confirmed by responses to another question where respondents were asked whether the advantages of nuclear power outweighed the risks. The risks of nuclear power as an energy source were judged to outweigh its advantages by 53% of respondents, whilst only 33% judged that the advantages outweigh the risks it poses. It should be noted that this was a closed question: no option was given to provide a balanced view.[1]

Figure A4.2: Attitudes towards use of different energy sources in the respondent's home country



The Eurobarometer data cited above show that a majority thinks that the risks of nuclear power outweigh its advantages and that nuclear is the least favoured way to produce electricity.

If we make the assumption that the public's attitude to different energy sources is linked to the risk that the public perceives from the same energy sources, we can broadly compare public attitudes (such as those shown in Figure A4.2) with the consequences of a range of severe energy-related accidents to allow a broad perspective on the difference between actual risk and the public's perception of risk.

It should be noted that this comparison aims to provide a general perspective on public perception of risk; it is not intended to be specific to radioactive waste management. However, the Eurobarometer data show (see for example Figure A4.5) that many people do not differentiate between the risks associated with nuclear power stations and the risks from radioactive waste disposal facilities. It is therefore judged that the relationship between actual and perceived risk for radioactive waste shows similarities with that for nuclear power production.

1. Respondents were asked to choose between two answers: "The advantages of nuclear power as an energy source outweigh the risks it poses" and "The risks of nuclear power as an energy source outweigh its advantages". Six percent of people spontaneously said "neither" whilst 8% responded "don't know".

Severe accident data analysis

Severe accidents are the most controversial in terms of public perception and energy politics. There are many ways of defining a "severe" accident. PSI has adopted a definition that contains seven criteria describing different consequence categories, and an accident is considered severe if one or several of these criteria are met: (Burgherr and Hirschberg, 2008a; Hirschberg, *et al.*, 1998)

- at least 5 fatalities; or
- at least 10 injured; or
- at least 200 evacuees; or
- extensive ban on consumption of food; or
- releases of hydrocarbons exceeding 10 000 t.; or
- enforced clean-up of land and water over an area of at least 25 km^2; or
- economic loss of at least 5 million USD (2000).[2]

Generally, the number of fatalities constitutes the most reliable indicator of an accident's severity because it is collected with most administrative thoroughness. (Burgherr and Hirschberg, 2008b) Therefore, results presented in this overview focus on the number of fatalities, with exception of two tables at the end that provide additional information on injured and evacuees. The analysis presented here covers severe accidents that occurred worldwide in the period from 1970 to 2005. (Burgherr, *et al.*, 2008)

PSI's database ENSAD (Energy-related Severe Accident Database) comprises real historic accident data from a wide variety of sources encompassing fossil, hydro and nuclear energy chains, all of which entail significant health, environmental or socio-political risks. ENSAD contains data on 8 688 energy-related accidents, of which 2 368 resulted in five or more fatalities (Burgherr, *et al.*, 2008). These amount in total to 90 374 immediate fatalities summed over all energy chains. (When assessing energy-related accidents and risks, it is essential to consider full energy chains because accidents at power plants are minor compared to the other chain stages). Of the 2 368 severe accidents with at least five fatalities, the coal chain accounted for 67.1% (1 588 accidents), whereas only one occurred in the nuclear chain (Chernobyl).

Table A4.2 summarises the severe (≥ 5 fatalities) accidents that occurred in the fossil, hydro and nuclear energy chains in the period 1970-2005. The largest numbers of immediate fatalities in the fossil energy chains was for coal and oil. The energy chain responsible for the largest number of immediate deaths was hydroelectricity, because the Banqiao/Shimantan dam failure in China in 1975 alone resulted in 26 000 victims.

Results are provided separately for OECD and non-OECD countries because of large differences in levels of technological development and safety performance, including regulatory frameworks and safety culture.

2. To take account of inflation, USD values were extrapolated using the US Consumer Price Index (CPI) to obtain year 2000 values.

Table A4.2: Summary of severe accidents with at least 5 immediate fatalities that occurred in fossil, hydro and nuclear energy chains in the period 1970-2005

	OECD			Non-OECD		
Energy chain	**Accidents**	**Fatalities**	**Fatalities/GWey**	**Accidents**	**Fatalities**	**Fatalities/GWey**
Coal	81	2 123	0.128	144 1 363 (818) (a)	5360 24 456 (11 302) (a)	0.587 3.079 (6.279) (a)
Oil	174	3 338	0.103	308	17 990	0.814
Natural gas	103	1 204	0.082	61	1 366	0.121
LPG	59	1 875	1.607	61	2 610	13.994
Hydro	1	14	0.003	12 11	30 007 4 007 (b)	8.175 1.092
Nuclear	0	0	–	1	31 (c)	0.036
Total	**418**	**8 554**		**1 950**	**81 820**	

* Accident statistics are given for OECD and non-OECD countries. For the coal chain, non-OECD w/o China and China alone are given separately.

(a) First line: Coal non-OECD w/o China; second and third line: Coal China 1970-2005, and in parentheses 1994-1999. Note that data for 1994-1999 are fully representative, whereas particularly earlier years are subject to substantial underreporting. (Burgherr and Hirschberg, 2007; Hirschberg, *et al.*, 2003a; Hirschberg, *et al.*, 2003b)

(b) Banqiao/Shimantan dam failure (China, 1975) alone caused 26 000 fatalities.

(c) Only immediate fatalities. In the case of Chernobyl estimates for latent fatalities range from about 9 000 for Ukraine, Russia and Belarus to about 33 000 for the whole northern hemisphere in the next 70 years (Hirschberg, *et al.*, 1998) According to a recent study (Chernobyl Forum, 2005) by numerous United Nations organisations (IAEA, WHO, UNDP, FAO, UNEP, UN-OCHA and UNSCEAR) up to 4 000 persons could die due to radiation exposure in the most contaminated areas. This estimate is substantially lower than the upper limit of the PSI interval, which, however, was not restricted to the most contaminated areas.

Source: Burgherr, *et al.*, 2008.

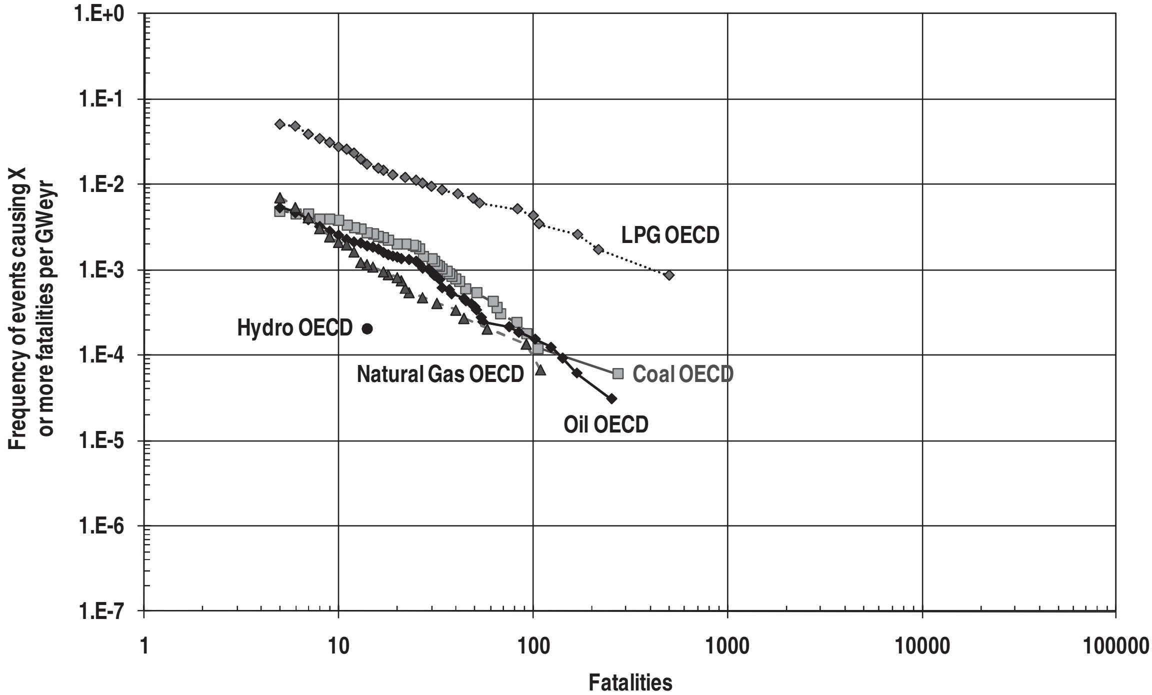
Frequency-consequence (F-N) curves are a common way to express collective or societal risks in quantitative risk assessment. They show the probability of accidents with varying degrees of consequence, such as fatalities. F-N curves provide an estimate of the risk of accidents that affect a large number of people by showing the cumulative frequency (F) of events having N or more fatalities, usually presented in a graph with two logarithmic axes.

Figure A4.3 shows F-N curves for severe energy-related accidents (≥ 5 fatalities) in OECD and non-OECD countries. In both sets of countries, fossil energy chains show higher historic frequencies of actual severe accidents than hydro, with liquefied petroleum gas (LPG) exhibiting the worst performance and natural gas the best. In OECD countries, there is only one data point for hydro because there was only one severe hydro accident in the period being analysed (Teton, United States in 1976 with 14 fatalities).

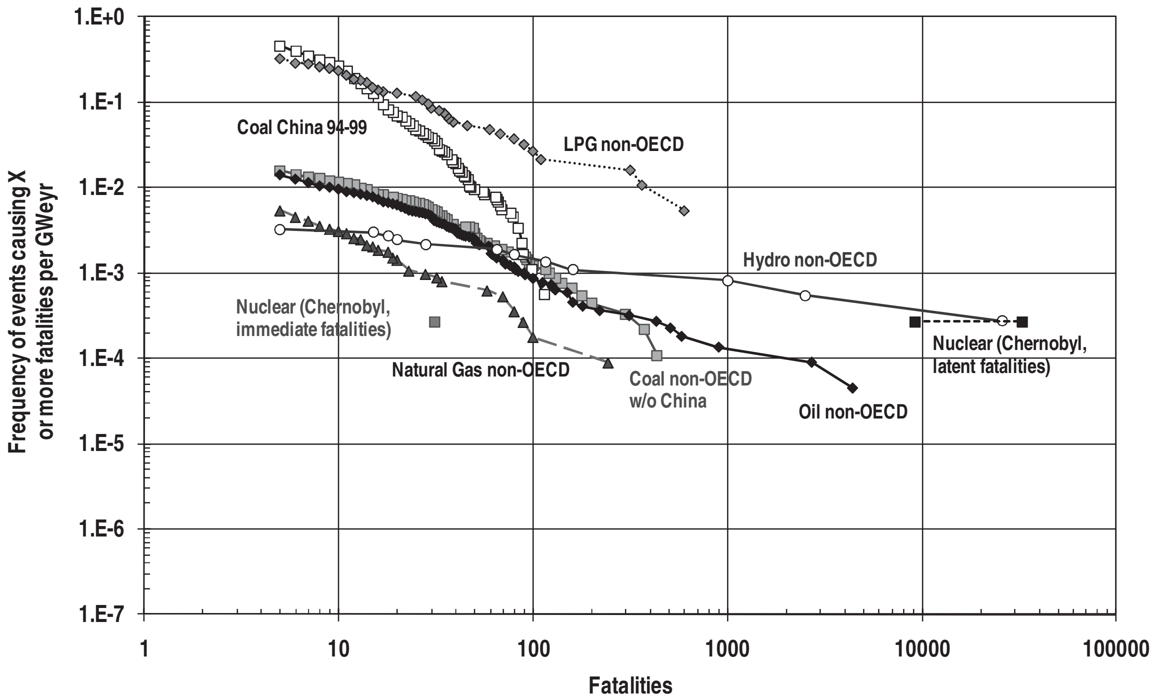
In non-OECD countries, there were 31 immediate fatalities following the Chernobyl accident, with latent deaths estimated to be between 9 000 and 33 000 over the next 70 years (Hirschberg, *et al.*, 1998). (Extrapolating these nuclear energy risks to current OECD countries, where demonstrably safer technologies are operated under a strict regulatory regime, is not appropriate – and this is predominantly true for the current situation in non-OECD countries). Latent deaths from fossil energy chains (through either health effects or climate change) have not been considered in this analysis. Despite the extremely serious nature of the Chernobyl event, it is clear that, for non-OECD countries, the probability and consequence of the world's most catastrophic nuclear event is comparable with fossil generation (even without consideration of the latent effects of fossil generation) and marginally better than hydropower.

Figure A4.3: Comparison between frequency-consequence curves for full energy chains, based on historical experience of severe (≥ 5 fatalities) accidents

A4.3a: OECD countries (1970-2005)



A4.3b: Non-OECD countries (1970-2005 except for “Coal China” 1994-99)



Source: Burgherr, *et al.*, 2008.

Outcome

The following three tables present the ten energy-related severe accidents that had the highest numbers of immediate fatalities, the highest numbers of injured and the highest numbers of evacuees. These tables are a way of demonstrating the consequence of accidents associated with different energy chains to allow comparison with the public's perception of risk, as judged by attitudes to different energy sources. All the data come from PSI and refer to the period 1970 to 2005. (Burgherr, *et al.*, 2008)

Table A4.3: The ten energy-related severe accidents with the highest number of immediate fatalities

Energy chain	Date	Country	Energy chain stage (Facility)	Fatalities	Injured	Evacuees	Costs (Mio USD 2009)
Hydro	05.08.1975	China	Power Plant (Banqiao / Shimantan dam)	26 000	–	–	–
Oil	20.1219.87	Philippines	Transport to Refinery (collision oil tanker with ferry)	4 386	26	–	–
Oil	01.11.1982	Afghanistan	Regional Distribution (tank truck collision with other vehicle)	2 700	400	–	–
Hydro	11.08.1979	India	Power Plant (Macchu 2 dam)	2 500	–	150 000	1 563
Hydro	18.09.1980	India	Power Plant (Hirakud dam)	1 000	–	–	–
Oil	18.10.1998	Nigeria	Regional Distribution (petrol pipeline explosion)	900	100	–	–
LPG	04.06.1989	Russia	Long Distance Transport (LPG pipeline explosion)	600	755	–	–
Oil	02.11.1994	Egypt	Regional Distribution (railway derailment, blaze of aviation fuel)	580	–	20 000	202
Oil	25.02.1984	Brazil	Regional Distribution (explosion and fire at gasoline pipeline)	508	150	2 500	–
LPG	19.11.1984	Mexico	Regional Distribution (explosion and fire at LPG terminal)	498	7 231	250 000	4

Source: Burgherr, *et al.*, 2008.

Table A4.4: The ten energy-related severe accidents with the highest number of injured

Energy chain	Date	Country	Energy chain stage (Facility)	Fatalities	Injured	Evacuees	Costs (Mio USD 2009)
Natural Gas	23.12.2003	China	Extraction (natural gas well explosion)	243	10 175	61 000	105
LPG	19.11.1984	Mexico	Regional Distribution (explosion and fire at LPG terminal)	498	7 231	250 000	4
Oil	17.01.1980	Nigeria	Extraction (blow-out of Funiwa No. 5 well)	180	3 000	–	–
Oil	22.04.1992	Mexico	Regional Distribution (petrol pipeline leak)	252	1 600	5 000	457
Oil	04.10.1988	Russia	Regional Distribution (fuel explosion after train collision)	5	1 020	–	–
Oil	19.12.1982	Venezuela	Power Plant (storage tank fire)	160	1 000	40 000	115
Hydro	05.06.1976	United States	Power Plant (Teton dam)	14	800	35 000	3 759
LPG	01.07.1972	Mexico	Regional Distribution (explosion and fire of rail-tanker cars)	8	800	300	7
LPG	04.06.1989	Russia	Long Distance Transport (LPG pipeline explosion)	600	755	–	–
Oil	25.03.1999	United States	Refinery (fire and explosion)	0	603	–	317

Source: Burgherr, *et al.*, 2008.

Table A4.5: The ten energy-related severe accidents with the highest number of evacuees

Energy chain	Date	Country	Energy chain stage (Facility)	Fatalities	Injured	Evacuees	Costs (Mio USD 2009)
LPG	19.11.1984	Mexico	Regional Distribution (explosion and fire at LPG terminal)	498	7 231	250 000	4
LPG	11.11.1979	Canada	Regional Distribution (series of explosions after LPG tankcars derailed)	0	8	250 000	29
Nuclear	28.03.1979	United States	Power Plant (Three Mile Island)	0	0	200 000	7 394
LPG	14.09.1997	India	Refinery (LPG release followed by explosion and fire)	60	39	150 000	20
Hydro	11.08.1979	India	Power Plant (Macchu 2 dam)	2 500	–	150 000	1 563
Nuclear	26.04.1986	Ukraine	Power Plant (Chernobyl)	31	370	135 000	462 125
Oil	25.05.1988	Mexico	Regional Distribution (explosion and fire at storage site)	0	70	100 000	–
Natural Gas	23.12.2003	China	Extraction (natural gas well explosion)	243	10 175	61 000	105
Oil	26.02.1988	United States	Regional Distribution (roadtanker fire)	1	–	60 000	2
Oil	19.12.1982	Venezuela	Power Plant (storage tank fire)	160	1 000	40 000	115

Source: Burgherr, *et al.*, 2008.

Nuclear power appears in these "top ten" lists only for highest numbers of evacuees after the accidents at Three Mile Island, United States and Chernobyl, Ukraine, where there were zero and 31 immediate fatalities respectively. Although high, the numbers of evacuees in these nuclear power plant accidents was less than that for LPG regional distribution accidents in Mexico and Canada.

Comparison of these consequence data with judgements of public risk perception shows that the consequences of severe accidents do not necessarily correlate with the public's perception or acceptance of risk.

A4.5 Public opinion on nuclear power and radioactive waste

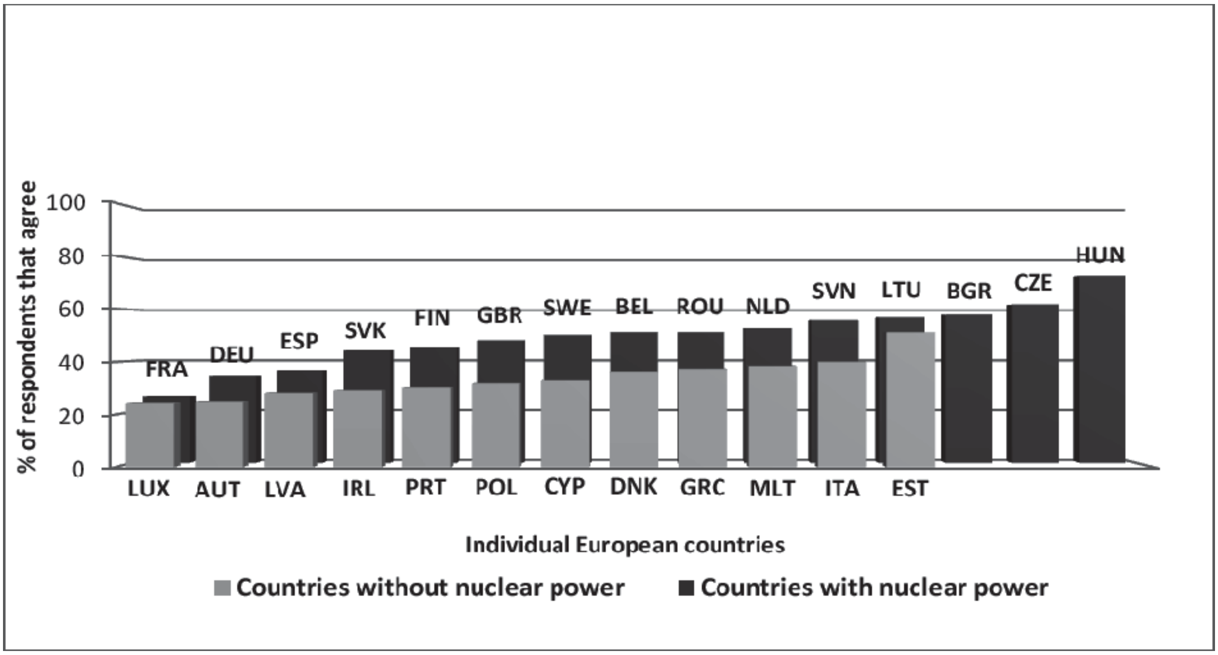
For many people nuclear power represents complex technology that is difficult to understand. Many have the misconception that nuclear power facilities can explode like nuclear weapons. As noted above, many people do not differentiate between the risks associated with nuclear power stations and the risks from radioactive waste disposal facilities.

A Eurobarometer[3] poll (EC, 2007) shows that disposal of radioactive waste is seen by many Europeans as a significant reason to oppose nuclear energy. The fieldwork for this poll was carried out in 2005.

Firstly, we should recognise from this poll that a majority of Europeans (59%) believes that nuclear plants can be operated safely, against 31% who do not. Respondents believe the biggest risks associated with nuclear power include disposal of radioactive waste, with only 39% agreeing that it can be done safely.

3. Since the text of this document was produced, a further Eurobarometer poll has been conductued, see http://ec.europa.eu/public_opinion/archives/ebs/ebs_297_en.pdf. This shows that, over the three years between which the data was collected, support for nuclear power has generally increased. However, the messages derived from the 2005 poll still remain valid.

Figure A4.4: Europeans' views on disposal of radioactive waste



Data presented in the Eurobarometer poll allows insight into the changes in attitudes to nuclear energy that might occur if the radioactive waste problem were solved.

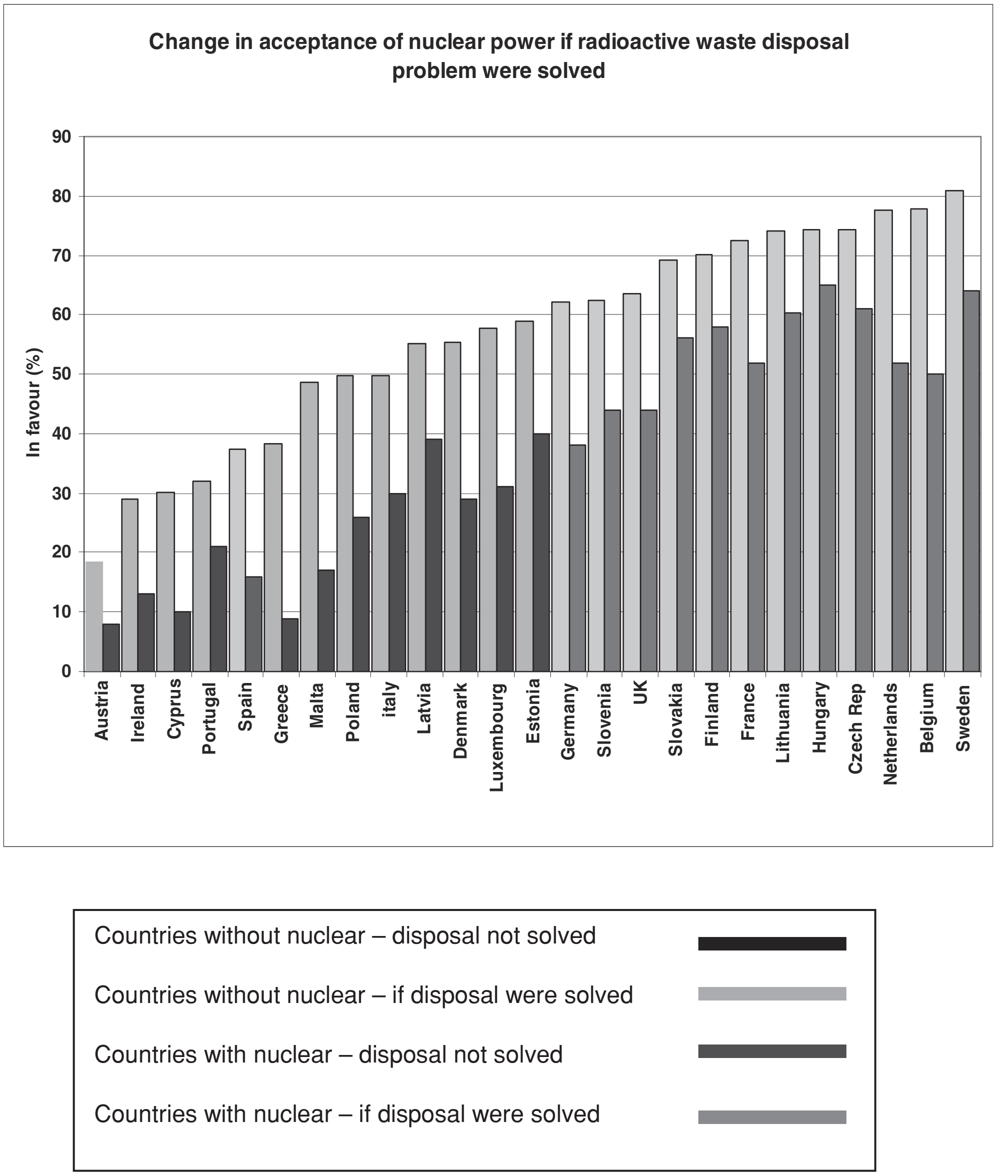
The poll first asked, "Are you totally in favour, fairly in favour, fairly opposed or totally opposed to energy produced by nuclear power stations?" This showed 55% of people to be opposed to nuclear and 37% to be in favour. Opponents of nuclear energy were then asked to what extent they would be in favour of nuclear energy **if** the problem of radioactive waste were resolved.

Responses to this question show that 38% of those opposed to nuclear energy would support it, if the issue of radioactive waste disposal were to be resolved. Just over a half (57%) of people opposed to nuclear would continue to be opposed if the issue of waste were resolved.[4]

This outcome is shown in Figure A4.5, split between countries with and without nuclear power. This shows that citizens of 16 of the (then) 25 EU countries would support nuclear if the waste problem were solved, whilst citizens of only 8 countries would support nuclear with the issue unresolved. The somewhat anomalous position of Spanish public opinion is evident in this figure.

4. The more recent poll (2008) referred to above showed support for nuclear power had grown from 37% to 44% and opposition reduced from 55% to 44%. Of those opposed, 39% would change their mind if the radwaste issue was resolved, 48% would not and 8% considered there was no safe solution to radwaste disposal. Hence the 38/57 split of the earlier poll remained virtually unchanged.

Figure A4.5: Europeans' change in acceptance of nuclear power if the radioactive waste disposal problem were solved



More evidence of the depth of concern on radioactive waste disposal comes from responses to further questions in the Eurobarometer poll.

- 92% agree that a solution for highly radioactive waste should be developed now and not left for future generations;
- 81% believe that it is politically unpopular to take decisions about the handling of any dangerous waste;
- 79% think that the delay in making decisions in most countries means there is no safe way of disposing of highly radioactive waste.

In June 2007, a poll by the Ministry of Industry in France asked, "Which are the two most important disadvantages with nuclear power?" 37% of respondents said the production and disposal of radioactive waste. An annual opinion survey among young Slovenians (NSS, 2007) found that around 35-37% of the respondents consistently saw the disposal of spent fuel as the most important disadvantage of nuclear power, more than those who cited the risk of a major accident.

The issue of radioactive waste is of significant concern to Canadians. (NRC, 2007) A large majority (82%) agree that new nuclear power plants should not be constructed until the problem of radioactive waste disposal is solved.

Support for nuclear energy would be expected to increase considerably if the matter of radioactive waste disposal were resolved.

The outcomes of these various opinion polls show that the future of nuclear power is dependent on managing radioactive waste, including its disposal, in a way that is acceptable to the public. Currently, the perceived risk from managing radioactive waste is high, but if the public sees that waste can be disposed safely, it is possible (but clearly by no means certain) that perceived risk might eventually reduce as has been seen in the study of hazardous waste management facilities described in the next section. Resolution of the waste issue in one country might have a positive impact on the public's perception of radioactive waste disposal elsewhere.

A4.6 Public opinion on hazardous waste

According to the surveys that lie behind the information presented in Figure A4.1, public reaction to hazardous waste disposal is similar to, but perhaps not as extreme as, the reaction to radioactive waste disposal.

Waste disposal facilities have become a focal point for environmental concerns and create intense public opposition. A possible reason for this is that the public has grown more mistrustful of government and industry, what Laird has referred to as the "decline of deference". (Laird, 1989) It is no longer obvious that the public regards those entities as having requisite legitimacy for taking decisions on their own. In addition, the public now recognises that it is possible to stop the introduction of new facilities, or shut down existing ones, by working with community groups and national environmental organisations. It is thus not surprising that the rate of commissioning new hazardous waste facilities (treatment, disposal and incineration) has decreased in the past 15 years.

Public empowerment in risk-management decisions poses strong challenges when siting waste management facilities, largely because the process of communication shifts from a didactic, one-way process to a shared process in which the form of a project may change in the light of public values. Those concerned with finding a home for a new facility need to be aware of how public values about technology are framed, their perceptions of institutional credibility and trust, the agendas of the different interested parties that motivate their participation in siting debates, and the uncertainties that surround the effectiveness of different participation processes. (Kasperson, 1986)

The risk perception effects on the psychological well being of people living near an incinerator have been studied by Maria Luisa Lima. (Lima, 2004) Four rounds of surveys took place before and four after an incinerator for hazardous waste started working in Portugal. The study included the assessment of psychological symptoms (anxiety, depression and stress), risk perception and overall attitudes towards the incinerator. Some of the results were:

- In the beginning, the perceived risk was higher for residents living closer to the site, who also had a less favourable attitude towards the new plant. This caused an increased amount of anxiety, depression and stress for these residents.

- There was an adaptation effect for those living close to the operating incinerator. After some time they became less opposed to the plant and held a lower level of perceived risk.

During the 1990s, considerable attention was focussed on studies of organised citizen opposition to hazardous and nuclear waste facilities. (Alley, *et al.* 1995; Aronoff and Gunter, 1994; Brown and Masterson-Allen, 1994; Fitchen, 1991; Murdock, *et al. Eds.,* 1983) In 1997, Solheim tried to identify the nature of public concerns associated not only with possibly hosting a landfill for hazardous waste but also with the process through which such decisions are reached. Their study provides clear evidence that excluding the public from the siting approval process is likely to result in a negative response to proposed waste management facilities. (Solheim, *et al*, 1997)

A4.7 Stakeholder involvement

The mid-1990s saw a growing expectation on the part of the public that it would be more directly involved in decision making about technology in general. This, of course, represented a clear challenge to the way in which such decisions had traditionally been taken. In liberal democracies, duly elected governments had been understood to have a mandate to take those decisions and to delegate authority to a whole range of expert bodies to oversee the implementation and operation of technologies. Consultation with interested parties was always a part of this overall process, but the complex nature of many of the issues at stake made it seem natural that much would remain the preserve of the experts in the various fields. Therefore, for many of the traditional decision makers in the 90s, the notion that a broad range of "stakeholders", many perhaps without any expertise in the field in question, should be involved in decision making raised apparently difficult questions.

In 2000, the NEA formed the Forum on Stakeholder Confidence (FSC) (NEA website), which facilitates sharing of experience in addressing the societal dimension of radioactive waste management. The Forum explores means of ensuring an effective dialogue with the public with a view to strengthening confidence in the decision-making processes. The FSC convenes a series of alternating meetings and national workshops focusing on stakeholder involvement in waste management issues in the host country. Such workshops have been held in Finland in 2001, Canada in 2002, Belgium in 2003, Germany in 2004, Spain in 2005 and Hungary in 2006.

A clear outcome from the NEA discussions is that the time when exchanges between waste management institutions and society were confined to rigid mechanisms is over. A more complex interaction is now taking place among players at national, regional and especially at local levels, as large industrial projects are highly dependent on siting and other local considerations, and a broader, more realistic view of decision making is taking shape. It is clear that several useful goals are achieved through stakeholder involvement, including:

- incorporating public values into decisions;
- increasing the substantive quality of decisions;
- resolving conflict among competing interests;
- building trust in institutions;
- educating and informing the public.

These findings are in agreement with other recent work in this area, notably at the OECD [the Public Management programme] (Vergez, 2003), and the European Commission. (RISKGOV, 2004; TRUSTNET, 2004; Atherton, 2003)

Involving the public in decision-making is essential since the public includes individuals who will have to live with the decisions made by the policy makers for decades to come. In addition, they are likely to identify factors and issues – especially socio-political matters – that policy makers had not necessarily considered. It has been said: (Slovic, 1987)

> "There is wisdom as well as error in public attitudes and perceptions. Lay people sometimes lack certain information about hazards. However, their basic conceptualisation of risk is much richer than that of experts and reflects legitimate concerns that are typically omitted from expert risk assessments. As a result, risk communication and risk management efforts are destined to fail unless they are structured as a two-way process. Each side, expert and public, has something valid to contribute. Each side must respect the insights and intelligence of the other."

The public participation element is also stressed in the 1998 Aarhus *Convention on Access to Information, Public Participation in Decision-making and Access to Justice in Environmental Matters*. The Aarhus Convention recognises that:

> "In the field of the environment, improved access to information and public participation in decision-making enhances the quality and the implementation of decisions, contributes to public awareness of environmental issues, gives the public the opportunity to express its concerns and enables public authorities to take due account of such concerns".

A4.7 Risk perception: conclusions

For almost all activities in society, risk – and how risk is perceived – are important considerations for decision making by governments as well as by industries and consumers. Societal acceptance of risk depends not on scientific evaluations, but on perceptions of risk and benefit. The public perceives that both radioactive and hazardous waste management are high-risk activities compared to many other activities in society. As shown in Figure A4.1, radioactive waste has the higher perceived risk of the two waste types.

This appendix has shown that the public perceives risk differently from "experts" in the field, who see risk as synonymous with expected annual mortality. However, this report does not discuss whether the public's judgement is correct or not. The same applies for "expert" perception. This report simply concludes that risk perceptions are different. Risk is assessed in an objective manner in engineering calculations, and particularly for assessments comparing potential environmental detriment. However, this definition does not represent the degree of risk that affected individuals might feel. This is known as "perceived risk". Perceived risk is subjective and depends on both the actual risk and a number of individual and societal risk perception factors such as whether the risk is seen as voluntary or imposed, whether an individual feels in control of the risk or if it is under the control of others. Risk perception worsens if it is seen as unfamiliar, poorly understood or relatively unknown; public consultation and participation appear to be the best ways to gain support when trying to site radioactive waste disposal facilities.

The public's perception of risk in the energy-related industries does not appear to be impacted by the actual or estimated consequences of severe accidents. In considering the consequences of severe energy-related accidents, in terms of the numbers of immediate fatalities, injuries and evacuations, nuclear power only appears in the top ten accidents with the highest evacuations – for Three Mile Island and for Chernobyl.

Disposal of radioactive waste is seen by many Europeans as a significant reason to oppose nuclear energy. Many people do not differentiate between the risks associated with nuclear power stations and the risks from radioactive waste disposal facilities. A large majority (79%) think that the delay in most countries on making decisions about disposal of highly radioactive waste means there is no safe way to do it. Support for nuclear energy would increase considerably if the matter of waste disposal were resolved.

Public acceptance plays an increasing role in the decision-making procedure for siting a new waste disposal facility and depends heavily on whether the public believes that they or their environment will be harmed by it – they have an intuitive view of whether the facility will be risky. Phased decision making and consultation has come to the fore as the preferred approach for development of deep disposal facilities for radioactive waste. Besides allowing for continued research and learning, phased decision making provides the opportunity to build broad societal confidence in the concept and to develop constructive relationships with the most affected regions.

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Appendix 5

LIST OF PARTICIPANTS

BELGIUM	
Guy COLLARD, Co-Chair	SCK•CEN
CZECH REPUBLIC	
Antonin VOKAL	Nuclear Research Institute
GERMANY	
Joachim WUTTKE	Federal Environment Agency
HUNGARY	
Zsuzsanna HAUSZMANN	ETV-Erőterv Zrt.
Szabolcs HORVÁTH	Ministry of Environment and Water
Péter ORMAI	RHK-Kht (PURAM)
ITALY	
Francesco TROIANI	ENEA
JAPAN	
Shigenobu HIRUSAWA	Institute of Applied Energy
Shinji KAWATSUMA	Japan Atomic Energy Agency
KOREA	
Jongwon CHOI	KAERI
Jei-Won YEON	KAERI
RUSSIAN FEDERATION	
Anna TALITSKAYA	Scientific and engineering centre for nuclear and radiation safety
SPAIN	
Mariano MOLINA MARTÍN Co-Chair	ENRESA
SWEDEN	
Sten BJURSTRÖM	Swedish Department of the Environment

SWITZERLAND

Matthias BRENNWALD	NAGRA

UNITED STATES

Alton HARRIS	US Department of Energy

IAEA

Jan-Marie POTIER	Division of Nuclear Fuel Cycle & Waste Technology

OECD Environment Directorate

Henrik HARJULA	National Policy Division/Waste Environment

OECD NEA

Torsten ENG	Nuclear Development Division
George BROWNLESS	Radiation Protection & Radioactive Waste Management Division
Elizabeth FORINASH	Radiation Protection & Radioactive Waste Management Division

Appendix 6

LIST OF ABBREVIATIONS

AEC	Atomic Energy Commission (United States)
ALARA	As low as reasonably achievable
ALARP	As low as reasonably practicable
APC	Air pollution control
AR	As-received
BAT	Best available techniques
BDAT	Best demonstrated available technologies
CCGT	Combined cycle gas turbine plant
CCS	Carbon dioxide capture and storage
CDM	Clean development mechanism
CPEs	Core performance elements
DAF	Dry, ash free
DK	Deponieklassen
DNA	Deoxyribonucleic acid
DPUI	Dose per unit intake or Sv/Bq
EC	European Commission
EPA	Environmental Protection Agency (United States)
ESM	Environmentally sound management
ETS	Greenhouse Gas Emission Trading Scheme (EU)
EU	European Union
EU-WSR	EC Regulation on shipments of waste
EW	Exempt waste
EWL	European Waste List (EC)
FEP	Features, events and processes
FGD	Flue gas desulphurisation
F-N	Frequency-consequence
FoE	Friends of the Earth International
FP	Framework Programme (EC)
FSC	Forum on Stakeholder Confidence
GCV	Gross calorific value
GWM	Groundwater monitoring

HLW	High-level waste
HM	Heavy metal
IAEA	International Atomic Energy Agency
IEA	International Energy Agency
ILW	Intermediate-level waste
INPRO	International Project on Innovative Nuclear Reactors and Fuel Cycles
IPCC	Intergovernmental Panel on Climate Change
LDR	Land disposal restrictions
LDU	Land disposal units
LGP	Liquid petroleum gas
LILW	Low- and intermediate-level waste
LL	Long-lived waste
LLW	Low-level waste
LNT	Linear no-threshold dose
LQGs	large quantity generators
LWR	Light water reactor
NAS/NRD	National Academy of Sciences/National Research Council (United States)
NCV	Net calorific value
NDC	The Committee for Technical and Economic Studies on Nuclear Energy and the Fuel Cycle
NEA	Nuclear Energy Agency
NEWMD	Net Enabled Waste Management Database
NGO	Non-governmental organisations
NPP	Nuclear power plant
OECD	Organisation for Economic Co-operation and Development
PAH	Polycyclic aromatic hydrocarbons
PCB	Polychlorinated biphenyls
PHWR	Pressurised heavy water reactor (CANDU)
POP	Persistent Organic Pollutants
PSA	Probabilistic Safety Assessment
PSI	Paul Scherrer Institut
PWR	Pressurised water reactors
QA	Quality assurance
RCRA	Resource Conservation and Recovery Act (United States)
R&D	Research and development
RWMC	Radioactive Waste Management Committee
SDA	Spray dry absorption

SL	Short-lived waste
SNF	Spent nuclear fuel
TCLP	Toxicity Characteristic Leaching Procedure
TSDF	Treatment, storage and disposal facilities
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
UNIPEDE	Now EURELECTRIC
URLs	Underground research laboratories
USDOE	United States Department of Energy
VLLW	Very low-level waste
VM	Volatile matters
WAC	Waste acceptance criteria
ZEP	Zero Emission Fossil Fuel Power Plants (EU)

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Large volumes of hazardous wastes are produced each year, however only a small proportion of them are radioactive. While disposal options for hazardous wastes are generally well established, some types of hazardous waste face issues similar to those for radioactive waste and also require long-term disposal arrangements. The objective of this NEA study is to put the management of radioactive waste into perspective, firstly by contrasting features of radioactive and hazardous wastes, together with their management policies and strategies, and secondly by examining the specific case of the wastes resulting from carbon capture and storage of fossil fuels. The study seeks to give policy makers and interested stakeholders a broad overview of the similarities and differences between radioactive and hazardous wastes and their management strategies.

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