COPERA CLAY 2024

A CONDITIONAL SAFETY CASE AND FEASIBILITY STUDY

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Contents


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Foreword

The principal objective of this report is to present an overview of the results and conclusions of the on-going work in the Netherlands on developing Safety Cases for a Geological Disposal Facility (GDF) in a Paleogene Clay formation. A major milestone in the clay studies was reached in 2017 with the publication of an initial Clay Safety Case based on the R&D work completed in the OPERA research programme, which focussed on a GDF in Boom Clay (one of the Paleogene clays). The present report updates this work with a second conditional Clay Safety Case, taking into account progress in the Netherlands and elsewhere in the intervening years.
As with the previous, OPERA Safety Case, the present report is termed a 'conditional' safety case, as it is recognised that, for eventual implementation of a GDF, various parameters will need to be updated, especially to match site-specific conditions, evolution of the GDF design and the exact waste inventory at the time of implementation.

This Clay Safety Case is accompanied by a parallel milestone report on a GDF in a Permo-Triassic salt formation (i.e., a Salt Safety Case) since both geological options are being considered. Because our intent is to ensure that the reports are consistent and can be read as stand-alone documents, significant sections of text are common. In addition, as much of the information in both reports has changed little since the 2017 OPERA Safety Case, some text has been brought forward from that report, amended with updated information if necessary.

Because both these reports mark major milestones in the Dutch overall radioactive waste management programme, they cover a wider and somewhat different scope from Safety Cases in other national disposal programmes that are closer to implementation. The principal objectives of the work described in both new COVRA reports are:

- To propose practical conceptual designs for a GDF and to examine their engineering feasibility;
- To assess the post-closure safety of a GDF based on these designs;
- To use the design information to provide a basis for estimating future costs and therefore to allow determination of the level of financial provisions to be made today by COVRA;
- To use the experience gained in producing the report to strengthen the national competences in scientific and technical areas related to geological disposal;
- To use the findings of the report to select and prioritise the R&D activities to be carried out in the Dutch disposal programme over the coming years;
- To inform decision-makers, the public and the scientific/technical community at large about the progress of geological disposal planning in the Netherlands.

The predecessor programme OPERA was financed by the Dutch Ministry of Economic Affairs and the public limited liability company Electriciteits-Produktiemaatschappij Zuid-Nederland (EPZ) and was coordinated by COVRA. The present on-going work is part of COVRA's OnderzoeksProgramma Voor Eindberging van Radioactief Afval (COPERA) work and is financed from the COVRA budget. COVRA acknowledges all the researchers in Dutch and foreign research organisations that are contributing to COPERA.

In line with current international practice, it was decided to structure COVRA's programme on geological disposal around the development of a series of Safety Cases for a GDF. However, the wider than usual range of objectives and the correspondingly wider target readership means that there are significant differences between COVRA's Safety Cases and GDF safety cases from other countries, which have often been prepared in order to meet some specific permitting or licensing requirements. The COPERA Safety Case is less comprehensive, being an early-stage report in a series of analyses that will be regularly updated and extended by further iterations as implementation comes closer.

This report focuses on clay as a host rock and the Netherlands has benefited greatly through the close cooperation with ONDRAF/NIRAS, which manages the Belgian waste disposal programme, in which comprehensive investigations on Boom Clay as a host rock have been in progress for many years. However, no decisions on possible locations for a GDF in the Netherlands will be taken for many years into the future and the next iterations of safety cases, whether in clay or in salt, are expected to continue to be generic and conditional in nature.

The present report extends beyond the scope normally used for a safety case for geological disposal of waste, in that:

- It contains additional material on some key engineering aspects of GDF implementation. This gives a firm basis for the safety assessments and allows early estimation of future costs;
- Emphasis is placed on embedding the safety case studies into the wider Requirements Management System (RMS) being developed to cover all of COVRA's radioactive waste management work;
- Additional information is included on the overall scope and structure of the R&D projects that underpin COPERA. Proposals for future scientific and technical studies leading eventually to implementation of a GDF are included at the end of the current report;
- As in the predecessor 2017 OPERA Safety Case, the wish to make the report accessible to a wide readership has required additional explanatory material to be included, to describe the basic concepts involved in geological disposal and to summarise current international consensus on the recognised approaches.

As with all its publications, COVRA welcomes any comment readers might have.

Summary

The principal objective of this report is to present an overview of the results and conclusions of the on-going work in the Netherlands on developing safety cases for a Geological Disposal Facility (GDF) in poorly indurated clay. The work is part of COVRA's broader COPERA programme, which is envisaged to run for decades and which also includes research on a GDF in rock salt and on multinational solutions. The structure of our long-term research programme can be used for several programming periods, and each decade will result in an iteration of two safety cases, one for a GDF in clay and one for a GDF in salt. The implementation of the European Directive on radioactive waste management requires an evaluation of the national programme every decade. The last Dutch national programme was published in 2016 and is currently being evaluated. A revision of the national programme needs to be completed in 2025. The safety cases for GDFs in clay and rock salt have been developed as input for this evaluation.

This safety case for a GDF in Paleogene clay updates and expands the OPERA (2017) clay safety case, taking into account progress in the Netherlands and elsewhere in the intervening years. The present COPERA(2024) safety case is less comprehensive than many other safety cases but wider in scope. The progress made in clay studies is mostly related to improved understanding of the physical and chemical processes involved in determining the safety of the multibarrier system with natural and engineered barriers. However, significant effort has also been put into examining more closely the practicability and efficiency of construction and operation of a GDF; this explains why the present report title refers to both safety and feasibility. Our intent is to ensure that the report can be read as a stand-alone document, and this means that information that remains the same as in the 2017 OPERA Safety Case has been brought forward from that report, amended with updated information only as necessary.

The present report is a scientific/technical document, describing engineering and geological requirements needed to assure that a safe GDF can be implemented in the Netherlands. We are, however, fully aware that a successful GDF programme must address both societal and technical issues. Globally, the greatest obstacles to geological disposal of waste have been those related to achieving sufficient public and political support for the concept itself and, most specifically, for siting work, including exploratory drilling. The Rathenau Institute is currently looking at a societally based approach to identifying possible siting areas and locations for a GDF. Information from their reports has been included in this safety case.

What is new or different from OPERA

Apart from Boom Clay, other Paleogene clay formations are also considered. This increases the range of potential siting regions that might be available in the Netherlands and implies that alternative disposal facility designs become feasible, including a multi-level option in which the types of waste can be disposed of according to their hazard potential.

Changes in design have been made in order to improve the practicality of the system for emplacement of waste packages in the GDF. In addition, radiation protection calculations have been initiated to demonstrate that operational safety can be provided.

Good quality Paleogene (Ypresian and Landen) clay borehole cores have now been obtained at a depth of about 400 m in Delft.

The safety case is being progressively interfaced with the Requirements Management System (RMS) that COVRA is developing; this will structure all of its activities from waste conditioning, through temporary waste storage to disposal operations, including ensuring that safety is provided after closure of the GDF. Further levels are defined, taking into account the need to be compatible with the parallel safety case in salt, and also with COVRA's waste storage programme.

The cost estimate has been updated with the waste inventory for Waste Scenario 1, made in 2022 in the framework of the national programme. In addition, new packaging assumptions have resulted in a significant decrease in the space required for disposal of some wastes.

Experimental measurements, especially with Paleogene clays and disposal representative concrete materials, have been analysed to provide some validation of the models and data used in the safety assessment.

Previous Dutch national disposal programmes OPLA, CORA and OPERA were all prepared at the conclusion of specific programmes with defined durations. COPERA is COVRA's on-going programme that will allow incorporation of recent foreign achievements in the prioritization for research into disposal of waste in the Netherlands, enhance national initiatives and support Dutch researchers working in international collaborations such as EURAD, in which 23 EU Member States develop the knowledge base for disposal of waste.

Introduction

Nuclear technologies are used in electricity generation, medicine, industry, agriculture, research and education. These technologies generate radioactive wastes that must be managed in a way that ensures safety and security at all times. For materials that remain hazardous for thousands to hundreds of thousands of years, the acknowledged approach to long-term isolation and containment is emplacement in a GDF in a stable geological environment beneath Earth's surface, and closing and sealing this GDF.

The Netherlands, along with other countries with significant quantities of long-lived radioactive wastes, has chosen geological disposal as the official national policy. The reference date for implementing a national GDF is around 2130, slightly more than 100 years from now. The extended timescale allow flexibility, in case options other than disposal in a national GDF become available, such as disposal of Dutch waste in a shared, multinational GDF.

COPERA is COVRA's on-going programme that started in 2020. It includes novel elements relative to the previous national programmes, OPLA (1993), CORA (2001) and OPERA(2017).

The main thrust of this COPERA Safety Case is to provide an overview of the arguments and evidence that can lead to enhancing technical and public confidence in the levels of safety achievable in an appropriately designed and located GDF. It addresses three important objectives:

- **Increase technical, public and political confidence** in the feasibility of establishing a safe GDF in the Netherlands.
- **Enhance the knowledge base in the Netherlands** related to geological disposal.
- **Guide future work** in the research for geological disposal of waste in the Netherlands.

The development of scientific and technical understanding, data and arguments that support the Safety Case has been structured by addressing specific research questions using a multidisciplinary approach, covering many areas of expertise.

How much waste is destined for disposal?

Three waste generation scenarios were made in 2022 in the framework of the national programme. Waste Scenario 1 is the same as that used in OPERA: Operation of Borssele Nuclear Power Plant until 2033 and replacement of the High Flux Reactor in Petten by Pallas. The expected eventual inventory of wastes from all sources that is destined for geological disposal is summarised below.
The design of the GDF in clay host rock can be easily extended with the other two Waste Scenarios, provided that the waste characteristics are the same as those used for Waste Scenario 1.

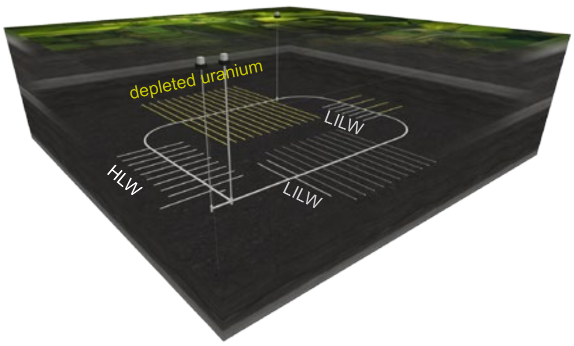
Waste Category	In storage		Packaged for disposal		
	Volume [m³]	Number of canisters / containers	Number of packages	Volume [m³]	Weight per package [tonne]
Spent research reactor fuel	49	244	244	1840	20
Vitrified HLW (vHLW)	86	478	478	3754	22
Compacted hulls & ends (Non heat generating HLW)	90	502	72	452	20
Dismantling waste (LILW)	3814	-	826	3814	Max 20
TE-NORM (LILW)	49360	-	12600	58070	Max 20
Processed LILW	31461	108400	108400	31461	Max 3

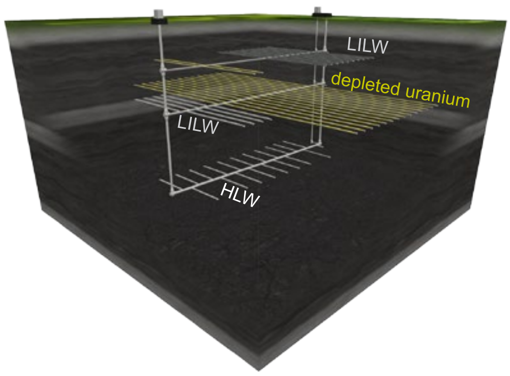
HLW = High Level Waste; LILW = Low and Intermediate Level Waste

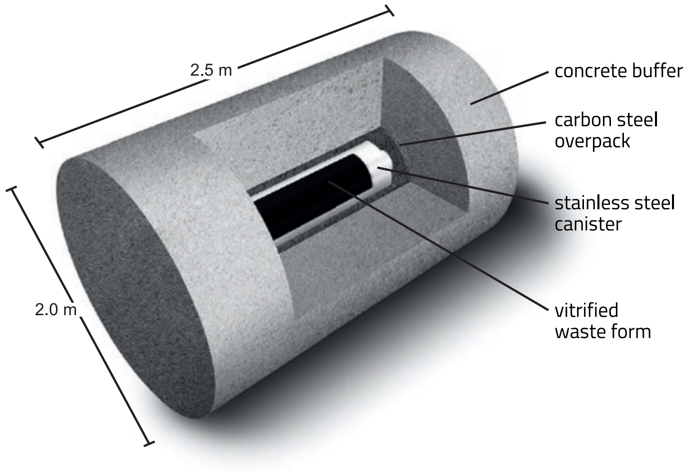
What could a Dutch a geological disposal facility look like?

The GDF design is based on the universally adopted 'multibarrier system' concept of natural and engineered barriers that contain and isolate the wastes and prevent, reduce or delay migration of radionuclides to the biosphere.

The conceptual design consists of surface and underground facilities, connected by vertical shafts. The underground facilities are networks of tunnels. Two options are considered: a single level GDF at a depth of about 500 m in a Paleogene Clay formation with a thickness of about 100 m, and a multi-level GDF with HLW being disposed of at 500 m depth and LILW disposed of at a smaller depth. Several Paleogene clay formations exist at different depths across the Netherlands, potentially allowing a multi-level design. A thickness of clay of as little as 20 m has been demonstrated to be sufficient for the construction of disposal tunnels.







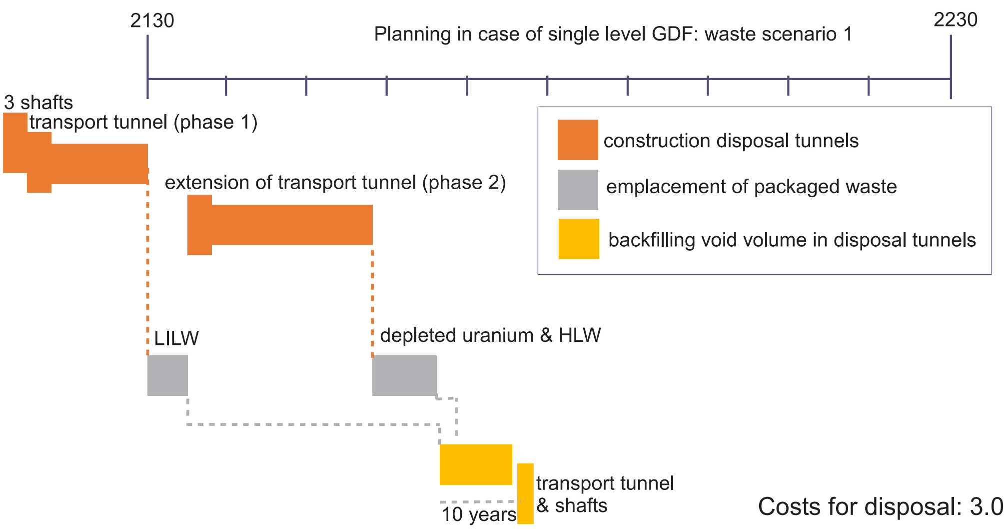
The GDF contains three groups of disposal tunnels: for HLW (vitrified high-level waste (vHLW), spent fuel from research reactors (SRRF), and non-heat-generating HLW) and for

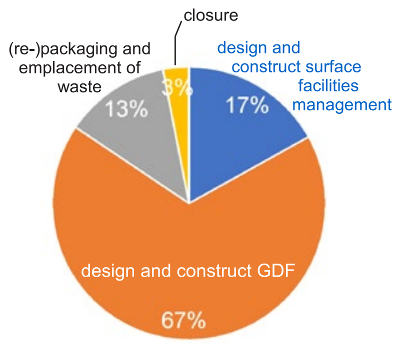
LILW and depleted uranium. Non-heat generating HLW is encapsulated in concrete containers. All heat generating HLW (vHLW and SRRF) is encapsulated in a supercontainer, adapted from the Belgian concept, consisting of a carbon steel overpack and a concrete buffer. A supercontainer for a single canister of vHLW is illustrated above.

A distinguishing feature of the disposal concept is the large amount of cementitious materials in the disposal tunnels and the waste containers. The disposal package for HLW includes a thick concrete buffer, the tunnels have a thick concrete liner, and a porous cementitious backfill is used to fill the gaps between the disposal package and the tunnel walls.

What are the costs?

The GDF design and the proposed implementation process allow an estimate to be made of the future costs that will be incurred. These estimates determine the financial contributions that are being paid by current waste generators in order to ensure that COVRA's provision will be sufficient for GDF implementation. The total costs for disposal in 2130, based on the timetable shown below, are estimated to be 3 billion EUR(2022), 70% of this being for



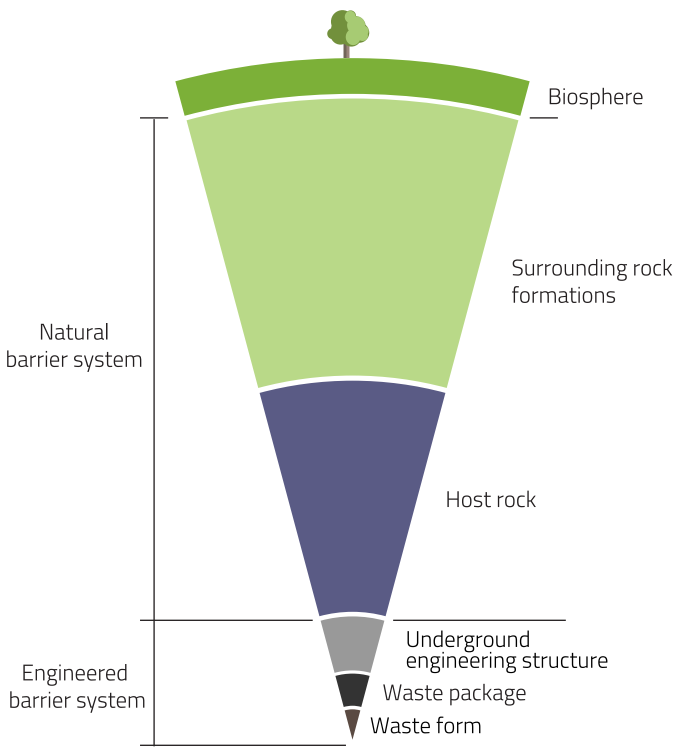


Costs for disposal: 3.0 billion euro (price level 2022)

design and construction. The cost estimate assumes that a definitive decision on the disposal method is made around 2100. There are several activities before waste packages can be emplaced at disposal depth such as the construction of the GDF that is composed of shafts and a structure of tunnels.
The construction of tunnels in the clay host rock can be done in several periods The first constructional phase is preceded with a site selection process. An underground observation phase of ten years is included. If this phase is extended to 50, or even 100 years, costs will not change significantly. The development of the disposal concept and costs for licensing are not included in the cost estimate. The planning scheme shows only activities related to the construction, operation and closure of the GDF.

The multibarrier system

The basis of geological disposal, which has been firmly established internationally for the last 45 years, is the concept of the multibarrier system, in which a series of engineered and natural barriers act in concert to isolate the waste and contain the radionuclides in the waste.



The relative contributions to safety of the various barriers at different times after closure of a disposal facility and the ways that they interact with each other depend upon the design of the disposal system. The design itself is dependent on the geological environment in which the facility is constructed. Consequently, the multibarrier system can function in different ways at different times in different disposal concepts.

Analysing safety

Quantitative analysis of the safety of the GDF is the central theme of a Safety Case. Estimates of potential radiological impacts to people are made for various future scenarios describing how the multibarrier system might evolve. The Normal Evolution Scenario (NES) is the central case considered and assumes undisturbed construction, operation and closure of the GDF, with no significant external disturbances of the multibarrier system in the future.
The OPERA safety assessment already recognised that, within the next 100,000 years to 1 million years, major climate change is to be expected, leading to periods of global cooling, lowering of sea level and the formation of permafrost and mid-latitude ice sheets, which might cover the GDF area. In COPERA, it is emphasized that this potential cover by ice sheets would be predominantly in the Northern part of the Netherlands, so that the safety assessments will be region specific. OPERA also identified a range of 'Alternative Evolution' scenarios for future assessment, as well as a range of speculative 'what-if' scenarios that might also be considered. Human intrusion scenarios have been added in this COPERA Safety Case. To date, results have been calculated only for the NES.

What is the Natural Barrier System?

The host rock for the GDF, a Paleogene Clay formation, along with the overlying and underlying geological formations, comprise the natural barriers within the multibarrier system.

Paleogene clays

The Paleogene Clay host rock is the principal natural barrier and the most important barrier in the complete multibarrier system.
The clay contributes to post-closure safety by providing a low permeability barrier that provides long-term containment of radio-nuclides by ensuring that their transport away from the EBS can only occur by the extremely slow process of diffusion through stagnant porewaters. Paleogene clays are old and stable.
The Paleogene marine clays were sedimented on the seafloor over the period from c.23 million to c.66 million years ago. All Paleogene clays have the capability to contain the waste for at least one million years. Across the Netherlands, the top of the Paleogene sediments is usually deeper than 250 m, with a thickness of more than 200 m in most areas, implying a wide potential choice of useable clay host formations. For OPERA, a generic case for Boom Clay was selected, with the GDF at 500 m depth in a clay layer 100 m thick.

Paleogene clays are poorly indurated and are considered aquitards in groundwater management terms, due to their low permeabilities. The porewaters within these clays are virtually stagnant (i.e. there is no water movement) and diffusion can be assumed to be the dominant process by which chemical species can move through them. The clays are sufficiently plastic that they do not contain open fractures that could act as pathways for water (and radio-nuclide) movement. All clays display a strong retention or retardation capacity for many radionuclides.

It is recognised that there are current uncertainties related to the properties of the Paleogene clays and that these need to be studied in the future. For example, permeability measurements of these clays at relevant disposal depth have not yet been made in the Netherlands; validation of the retardation of radionuclides in these clays has started in COPERA but more experimental research is necessary for a more reliable quantification; the potential impact on radionuclide transport of gases produced by corrosion of GDF materials needs further study.

Overlying and underlying rock formations

The thick sequence of older Paleogene (c.66 to 23 million years old) and more recent Neogene (c. 23 to c.2.6 million years old) sediments is called the North Sea Group; it broadly forms the upper hundreds of metres of the landmass across the Netherlands. The rock formations that overlie the clay host rock contribute to the post-closure safety by isolation of the waste and protection of the Engineered Barrier System (EBS) and clay host rock from dynamic natural processes. The sedimentary formations that immediately underlie and overlie the Paleogene clays are sandy and permeable. These sandy formations contribute to post-closure safety because any radionuclides that diffuse out of the Paleogene clays and move through the large bodies of groundwater they contain will be dispersed and diluted, thus reducing their concentrations and their consequent hazard potential. Most Paleogene clays are surrounded by saline sandy formations that are too saline for groundwater extraction; aquifers are usually present in Neogene or Quaternary (c.2.6 million years ago to the present) sediments.

How might climate change impact the natural barriers?

During the Quaternary glacial cycles, the Netherlands has periodically been covered by ice sheets extending down across the Baltic and North Sea areas from a Scandinavian ice cap. Not every glaciation has been sufficiently intense to cause ice cover as far south as the Netherlands and, even in the more intense glacial periods, not all of the present country has been covered by ice.

Especially in the northern part of the Netherlands, ice-sheet loading can affect hydraulic conditions in the Paleogene clays at depth and potentially result in water movement in the clay. These region-specific studies have not been modelled since the CORA programme. The modelled ice-sheet thickness in CORA was 1000 m, which is now considered unrealistically large, based on OPERA research. Outward advective flow from the clay formation during compaction by ice sheet loading is thus expected to be smaller than was calculated in the CORA programme.

A concern in siting the Dutch GDF will be to avoid the possibility of deep erosion by glacial meltwaters after a future intense glaciation, during the change in climate from a glacial to an interglacial state. This is considered to be the only potentially detrimental geological process that could substantially affect the normal evolution of the multibarrier system. In a future GDF siting programme, it will be essential to look in more detail at the likelihood and consequences of such a scenario. Current understanding is that the current interglacial conditions are likely to persist for at least the next 100,000 years. If a further glacial period, followed by deglaciation and potential deep erosion, does not affect a GDF until some time after 100,000 years, the radioactivity of the HLW will already have been markedly reduced.

The OPERA safety assessment made the simplifying assumption of a constant interglacial climate for the next million years, and radionuclide transport was calculated assuming present climate conditions. For at least the next 100,000 years, this is considered reasonably realistic and also generally conservative, in that relatively warm conditions are characterised by higher flow in the overlying formations than during colder periods. Inclusion of glacial climates will be dealt with in future scenario analysis work.

What is the Engineered Barrier System?

The EBS, which provides both physical and chemical containment of the radionuclides in the wastes, is protected by the stable Paleogene clay formation which limits movement of groundwater to the EBS. Some decades after closure, the EBS will essentially be comprised of a heterogeneous, concrete-dominated system with interconnected porosity filled with stagnant waters in which chemical reactions are mediated by the slow diffusion of chemical species.

Cementitious materials comprise much of the EBS

Cementitious materials (tunnel liner segments, backfill, buffer, waste conditioning matrices) dominate by volume in each section of the GDF - up to 98% in the case of the tunnels containing vHLW waste packages. In OPERA, these materials were conservatively assumed to have no physical containment role after closure of the GDF, but in reality they fulfil an important safety function, by controlling the movement of water, by creating highly alkaline conditions in porewaters and by providing mineral surfaces that can interact with radionuclides in solution. During COPERA, several cementitious backfill materials and COVRA waste conditioning matrices have been investigated. Analysis of new experimental results confirm that the permeabilities of these types of concrete are lower than those of Boom Clay measured at 225 m depth. The lack in observed chemical or mechanical changes for small concrete specimens that were exposed to synthetic clay pore water or air for several years is a result of these low permeabilities, since ingress of little or no gaseous and dissolved species can take place. The enhanced understanding of the mechanisms of leaching allows a proper choice of the type of cement to be used to manufacture concrete, so that degradation of the mechanical strength can be prevented over the period of concern in the safety assessment. The cementitious materials can also provide an important chemical buffer that enhances chemical containment of many radionuclides by reducing their solubilities and promoting ion exchange.
In particular, the type of ion exchange (cation or anion) is known to be pH dependent, which means that taking into account this favourable property requires knowledge of the evolution of the pH of concrete at different positions in the EBS.

The tunnel liner provides mechanical support for the tunnels during the operational phase. After closure, this support function is no longer assumed to function and overburden stresses can be transferred from the surrounding geological formations through the liner onto the mass of the EBS materials in the tunnels. The foamed concrete tunnel backfill is a porous backfill which can accommodate gas and reduce the gas pressure but in which microbial activity is feasible and may enhance the corrosion of metals. For this reason, the outer stainless steel envelope surrounding the concrete buffer of the waste package for HLW in earlier EBS designs is not included in the current EBS.

How will the waste packages behave in the multibarrier system?

Conservatively, only HLW waste packages have been assigned a post-closure containment role. The carbon steel overpack prevents water accessing the inner waste canister for a period determined by the ability of the concrete buffer to provide the chemical conditions to minimize steel corrosion. This prevents access of porewaters to the waste for as long as the overpack can sustain mechanical and early thermal stresses and resist failure through corrosion. It is designed to provide complete containment for thousands of years, beyond the early 'thermal period' of 1200 years when temperatures in the EBS are significantly elevated due to heat emission from the vHLW. Thermal calculations of the current design show that the heat emissions of SRRF are too low to significantly heat the clay host rock.

In the normal evolution scenario (NES), corrosion will eventually result in loss of integrity of the overpack safety function; this takes place at the so-called 'failure time' used in the safety assessment. Four cases for the lifetime of the overpack were studied in OPERA: 1,000 years, 35,000 years (the base case value), 70,000 years and 700,000 years. In COPERA, these calculations have been repeated when calculating releases of the long-lived radionuclide selenium-79, since this radionuclide was calculated to contribute most to the radiation dose rate in OPERA. Additional calculations were performed while taking into account only the low permeability of the backfill but, conservatively, not the permeability of the concrete buffer which is much lower. Mechanical analysis in COPERA shows that the thickness of the carbon steel overpack was sufficiently optimised in OPERA, so that no changes are proposed.

The Konrad Type II containers used for depleted uranium are assumed to have a failure time of 1,500 years. The 200 and 1,000 litre steel and cement LILW packages will contribute to chemical containment, but the conservative assumption in OPERA is that radionuclides are released instantaneously into the EBS porewaters after closure of the GDF, so an effective zero 'failure time' for LILW packages is used in the safety assessment.

Waste materials and gas production

The long-term behaviour of the solid waste forms, in particular how they react with and dissolve in pore waters in the EBS, influences the delay and attenuation of releases of radioactivity by limiting and spreading in time the release of radionuclides. The chemical reactions involved always consume water. Hydrogen gas can be produced by the corrosion of the metallic containers and from the waste forms, if they include metals. If the gas generation rate is larger than the capacity for migration out of the system as a dissolved gas, a free gas phase will be formed. This might result in gas-driven movement of radionuclides present in pore waters.

During COPERA, calculations have been carried out on rates of gas generation and its potential behaviour. These calculations show that the hydrogen evolved from the EBS in the case of vHLW does not exceed the hydrogen solubility in the porewater of the clay host rock. The hydrogen solubility would be exceeded for corrosion of aluminium in SRRF, assuming the hydrogen generation rates of solid pieces of aluminium exposed to aqueous solutions to be representative for the aluminium in SRRF surrounded by the low permeable concrete and clay.

In OPERA, the vHLW glass was conservatively assumed to dissolve either very rapidly, within 260 years, or else (still conservatively) over 20,000 years. These high glass dissolution rates were obtained from alteration rates of glass in which solid pieces of non-radioactive vitrified waste are exposed to a relatively large volume of an aqueous solution. During COPERA, more experimental results, in which the solid to liquid ratio is higher, have become available. Lower glass alteration rates are measured with these higher ratios. The calculated water consumption rates show that only the experiment with the highest ratio is representative for the vitrified waste form encapsulated in the concrete buffer. Also, the silicon concentration in equilibrium with the evolved cementitious minerals increases with reducing pH, further reducing the alteration rate of glass.

The radionuclide release rate from the waste form was assumed to depend on an alteration rate only for vHLW; for other wastes instant radionuclide release rate was assumed, after the so-called supercontainer 'failure time'. For LILW, an instant release rate was conservatively assumed to occur immediately upon closure of the GDF, except for depleted uranium. Depleted uranium, generated by URENCO during the uranium enrichment process, is the largest waste family by volume. Depleted uranium is also encapsulated in carbon steel, but with a smaller thickness than HLW. The uranium release rate into the clay host rock is constrained by an assumed uranium solubility. The assumed solubility is orders of magnitude larger than the measured concentrations of uranium in the clay pore water of Boom Clay.

How will the multibarrier system evolve over time?

The information available to quantify the performance of the multibarrier system is subject to different types and levels of uncertainty. OPERA allowed for this by making conservative simplifications, assuming poor performance, using pessimistic parameter values and omitting potentially beneficial processes. The results of the OPERA safety assessment are thus pessimistic forecasts of the performance of the multibarrier system. However, it is also essential for system engineering optimisation purposes to make best estimates of how we expect the multibarrier system to behave, acknowledging uncertainties along the way. This allows a balanced view that will inform later decisions on GDF design optimisation and, eventually, on acceptable site characteristics. This best estimate approach avoids over-engineering system components, allows waste to be disposed of according to their hazard potential, and prevents rejecting otherwise acceptable GDF sites.

OPERA presented a comparison (in different time frames) of the best estimate of the expected behaviour of components in the multibarrier system, based on the simplified assumptions of the safety assessment. This comparison is also done in COPERA. The expected behaviour is summarized below.

From closure to 1,000 years

The clay host rock is completely saturated at the start of the post-closure phase. The pores in concrete are partly filled with water and partly with the gases present in ordinary air: nitrogen, oxygen and traces of carbon dioxide. Clay pore water initially enters the backfill mainly through the joints between the concrete segments of the tunnel liner. The surface area of backfill exposed to water increases as the concrete in the tunnel segments become further saturated. Oxygen and carbon dioxide are consumed by reactions with minerals present in the cementitious phase of concrete, and nitrogen dissolves further as the saturation degree of concrete increases. The rate of corrosion of the carbon steel overpack is controlled by the amount of available water, the diffusion rate of dissolved iron away from the overpack surface, and the reducing (Eh) and alkaline conditions (pH) in the concrete buffer in the vicinity of the overpack. The small hydrogen generation rate during the anaerobic corrosion of the steel overpack ensures that only dissolved hydrogen can enter the clay host rock.

Up to 1,200 years, the temperature of the clay host rock in the vicinity of the EBS will be higher than its natural temperature at GDF depth, due to heat emission by the vHLW. There will be a thermal gradient from the HLW through the buffer, backfill and liner that will counteract inward flow of water from the clay host rock.

The lithostatic load of the geological formations overlying the tunnels has been taken up only by the tunnel liner in the operational phase. In the post-closure phase, there can be some additional support from the backfill and other EBS materials. The dissolved species in clay pore water entering the backfill and tunnel segments, and react with cementitious minerals leading to decalcification of these minerals, forming calcite, siliceous hydrates, some magnesium hydrates and magnesium siliceous hydrates. Leaching, which increases the porosity of concrete , has been minimized with a proper choice of the type of cement used to manufacture the concrete materials. An increase in porosity leads to a reduction in compressive strength. The porosity of concrete may however decrease by the formation of calcite. If a reduction in compressive strength is caused by decalcification of cement minerals, this reduction is localised to a few tens of mm at the edges of the concrete segments.

At the end of the thermal period, it is expected that the properties and geometry of the tunnels and other EBS materials will have changed very little, there will be limited chemical interaction between concrete and clay, and the carbon steel overpack will be mechanically and physically intact, corroding at a very low rate. The initial high radiotoxicity of vHLW and SRRF will have reduced considerably during this period of total containment.

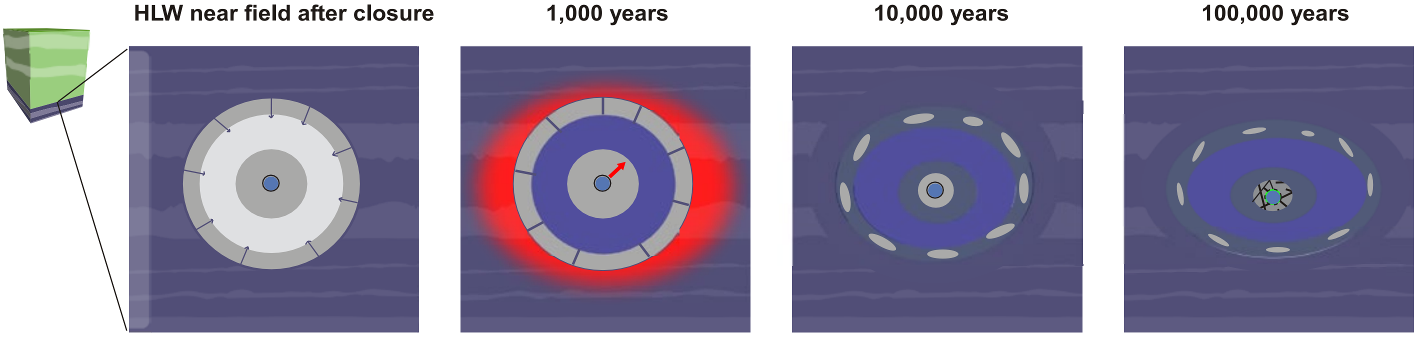
Elsewhere in the GDF, anaerobic corrosion rates of the steel on the outer surfaces of the LILW waste packages are controlled by the availability of water, the diffusion rate of dissolved iron and the reducing and alkaline conditions, but also by the microbial activity of the backfill in the vicinity of these outer surfaces. The porous cementitious backfill allows dispersion of gas so that the formation of a free gas phase is minimized. The degree of saturation of the waste package concrete increases so that the alteration rates of the waste forms become larger than the rates under dry storage conditions. Dissolved radionuclides slowly diffuse through waste package concrete and other engineered barriers and can enter the clay host rock.

A **simplified** behaviour is modelled in the OPERA safety assessment. In the base case, nothing happens for HLW since all the carbon steel overpacks fail by complete corrosion, exactly at 35,000 years. For depleted uranium also nothing happens since the Konrad containers fail at 1,500 years. Other LILW containers 'fail' at time of closure of the GDF and all radionuclides are assumed to dissolve instantaneously in the EBS and are free to enter the clay host rock.

From 1,000 to 10,000 years

All pores in concrete barrier materials are almost completely saturated, a steady state of water consumption rate by the

HLW near field after closure | **1,000 years** | **10,000 years** | **100,000 years**



The disposal tunnel has a concrete liner for mechanical support. After emplacement of the waste package in the tunnels, the void space between the package and the liner is backfilled with foamed concrete. Cementitious materials dominate the overall volume of the materials in the EBS. The low permeability of the concrete liner prevented drying of the clay host rock in the operational phase so that excavation-induced fractures are closed. Initially in the post-closure phase, clay pore water can access the backfill through the joints between the concrete segments in the liner. Later, the saturation degree of the concrete segments increases so that a larger surface area of the backfill is wetted by the clay pore water migrating through the liner.

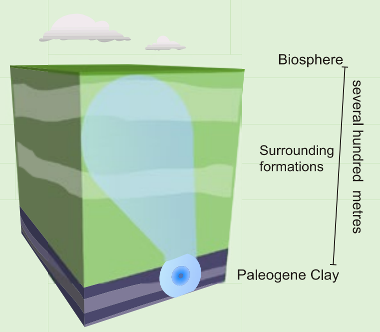
The waste heats the clay host rock in the vicinity of the EBS. Reactions between cement minerals and dissolved species from clay pore water have altered some of the backfill and liner concrete. This clay affected concrete has lost its strength and deforms under the load of the overburden. The slow anoxic corrosion rate ensures that the carbon steel overpack has not been breached.

The waste has cooled down and no longer heats the clay host rock. Reactions between cement minerals and dissolved species in the incoming clay pore water have led to a reduction in pH in the vicinity of the overpack. The anoxic corrosion rate of the overpack has increased. The vitrified waste form has not come into contact with water.

The radiotoxicity of the waste is lower than that of the original uranium ore. Fracture of the overpack allows contact between pore water and the vitrified waste. The majority of this waste becomes covered by a passivating film of hydrated glass and iron-phyllosilicates have been formed in the vicinity of the corroded steel.

1,000,000 years

Immobile, long-lived radionuclides will remain within the degraded EBS. Most other nuclides migrate very slowly through diffusion and retardation processes in the clay and eventually decay. Due to sorption, dispersion and dilution only extremely small concentrations of non-sorbing, long-lived nuclides reach the biosphere.



anaerobic corrosion process of the carbon steel overpack has been achieved. Dissolved iron and hydrogen diffuse further into the concrete buffer. The small connecting pore throat restricts diffusion of dissolved iron so that cementitious minerals start to react with dissolved iron, forming iron-affected concrete, a transformed medium. The mechanical strength of this medium is small so that the thickness of concrete buffer with a high strength, reduces. The highly alkaline conditions of the concrete buffer persist, so that the generation rate of the transformed medium is controlled by precipitation and not by ion exchange of dissolved iron with calcium.

Ingress of dissolved species in concrete pore water (bicarbonate, magnesium, sulphate) have further decalcified the cementitious minerals. If this decalcification leads to a decrease in compressive strength, the circular shape of the EBS starts to change slowly into an oval shape by creep of concrete and anisotropy of mechanical loads in the clay host rock.

By 10,000 years, vitrified HLW has almost achieved the same radiotoxicity as the uranium ore from which the fuel was originally manufactured. SRRF is more radiotoxic than uranium ore, due to the presence in the spent fuel of plutonium, which decays at a lower rate than the fission products and americium in vitrified HLW. During the production of vHLW, uranium and plutonium are removed from the waste and re-used to make fuel again. The content of uranium and plutonium in vHLW is therefore negligible compared to SRRF.

A **simplified, conservative** behaviour was modelled in the OPERA safety assessment. In the base case, there is no contact between and HLW and pore water since the carbon steel overpack is not breached. For depleted uranium, the steel containers fail by a combination of corrosion and lithostatic load, at an assumed time of 1,500 years. The reason for a earlier failure time of 1,500 years, compared to the 35,000 years for HLW overpack, is that the thickness of steel in the HLW overpack is larger than in the Konrad containers. The release of uranium into the clay host rock is limited by the solubility of uranium.

From 10,000 to100,000 years

The movement of dissolved species from the concrete materials into the clay host rock is very limited, since the concentration of dissolved calcium in the saline clay pore water is higher than, or similar to, the concentration in concrete pore water so that there is little concentration gradient to drive diffusion. The clay host rock itself will be little different from its original state. The continued ingress of dissolved species (e.g., magnesium, bicarbonate) from the clay host rock into the concrete materials further decalcifies these materials. The liner, backfill and buffer begin to lose their individual identity, to form a continuous mass of clay-affected concrete. Modelling studies show that the concrete buffer in the vicinity of the carbon steel overpack will retain its high pH.

It seems probable that the majority of the waste packages for heat-generating HLW would retain their containment function throughout this period. However, loss in compressive strength of the backfill and buffer, combined with reducing pH in the vicinity of the carbon steel overpack by further decalcification, and resulting in an increase of the corrosion rate of the overpack leading to an insufficient thickness with strength, may lead to breaching. At lower pH, cation exchange with cement minerals becomes dominant. The generation rate of the iron-affected concrete becomes controlled by ion exchange of dissolved iron, since ion exchange is a faster process than precipitation. The loss in compressive strength of concrete also increases the size of its connecting pore throats, so that diffusion values for dissolved iron become larger - perhaps larger than their values in the clay host rock. Both processes enhance a faster dissipation of dissolved iron in the vicinity of the carbon steel overpack, so that the corrosion rate increases. The corrosion rate can then become controlled by the permeability of the clay host rock instead of the (initial) lower permeability of the concrete materials.

After 20,000 years, the radiotoxicity of vitrified HLW has become lower than the radiotoxicity of the uranium ore from which the fuel was originally manufactured. The cross-over time for SRRF is towards the end of this period, at around 100,000 years.

The base case was **conservatively** modelled in the OPERA safety assessment by assuming that all waste packages for HLW fail at 35,000 years. At that time, there is contact between the waste and pore water, and the radionuclides becomes dissolved in the EBS. The high pH of the concrete buffer in the vicinity of the carbon steel overpack is conservatively modelled to remain for 80,000 years. Assuming a maximum anoxic corrosion rate in alkaline aqueous solutions of 2 μm per year to be representative, it takes 15,000 years to corrode 30 mm of steel. Assuming a maximum of 0.2 μm per year to be representative, it takes 150,000 years to corrode this thickness of steel. In COPERA, based on water consumption constraints, it has been calculated that the maximum possible corrosion rate is 1 μm per year for steel interfacing with the thick concrete buffer, using the updated design of the vHLW package and a permeability of concrete equal to COVRA's waste package concrete.

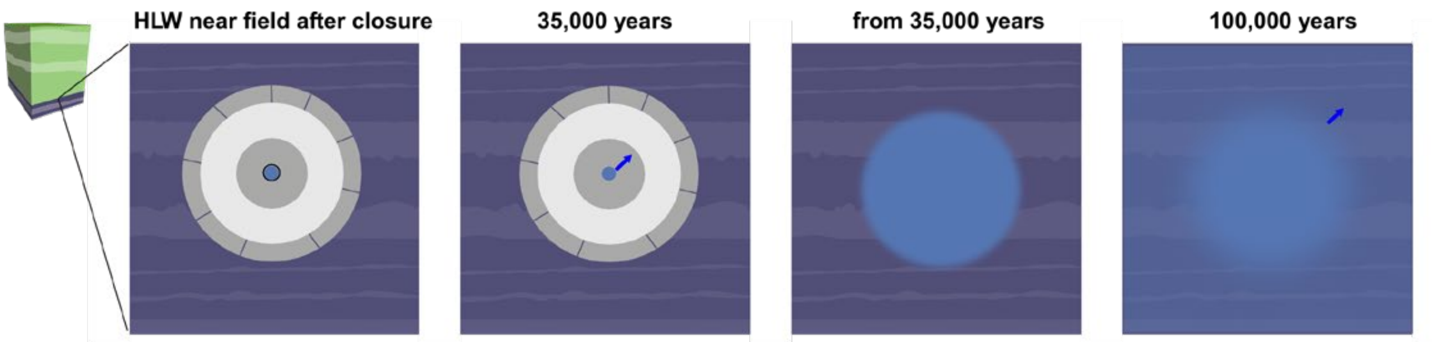
In OPERA, the vHLW is assumed to dissolve quite quickly in the base case: within 20,000 years. Modelling work in COPERA, based on water availability, estimates that this period would be much longer - at least 200,000 years. The release rate of dissolved radionuclides into the EBS is assumed to depend on the alteration rate of glass. For the SRRF, all the radionuclides are assumed to enter solution instantaneously. The approach developed during COPERA can also be used to estimate more realistically how radionuclides can be released into the EBS from SRRF as a function of the geometry of the multibarrier system and the low permeabilities of concrete and clay.

From 100,000 to one million years

At the start of the post-closure phase, there is some void volume within the vHLW stainless canister on top of the vitrified waste form. Because of this void volume, the overpack and the canister will crack as the thickness of the overpack decreases, its initially high strength reduces and the lithostatic load of the geological formations overlying the tunnels comes onto the supercontainer. This will be a progressive process over the 100,000 to one-million year time scale, with the formation of cracks staggered over many tens of thousands of years, so that access of pore waters to the vitrified waste form would be spread over long periods in time.

The initial alteration rates of glass are controlled by how fast pore water can enter the fractured canister. A passivation layer of hydrated glass is formed. In the vicinity of the corroding steel canister, the passivation capacity of this layer is not as strong as on a pure glass surface. The passivation layer is a mixture of clay minerals and zeolites. In the long-term, the properties of such a layer are similar to those of the rims found on basaltic glass (a natural volcanic glass). Estimated alteration rates of basaltic glass are about 0.1 µm per year. Uranium and plutonium from the degrading glass will be taken up by the clay minerals formed. Some radionuclides are not taken up by clay, for example selenium-79 with a half-life of 327,000 years. However, with the expected low alteration rates of glass, most of the selenium-79 is expected to decay within the vitrified waste form and not to be released to the surroundings. The small fraction of selenium-79 released diffuses slowly through clay-affected concrete, clay host rock and the surrounding rock formations. Dispersion and the large delay and dilution in space and time, implies that this mobile radionuclide can reach the biosphere only in very small concentrations.

After a million years, immobile and long-lived radionuclides will still remain within the clay-affected and iron-affected concrete of the EBS. Uranium-238, the main component of depleted uranium, will remain in the EBS until inexorable processes of geological erosion over hundreds of millions of years disperse it into new sediments and rocks. The residual uranium within the degraded EBS will behave like a naturally occurring ore body.



Up to the time of failure of the carbon steel overpack, which is assumed to take place after 35,000 years, the system remains effectively unchanged, with only slow corrosion of the overpack occuring. The failure time of the carbon steel overpack is determined by the corrosion rate and steel thickness.

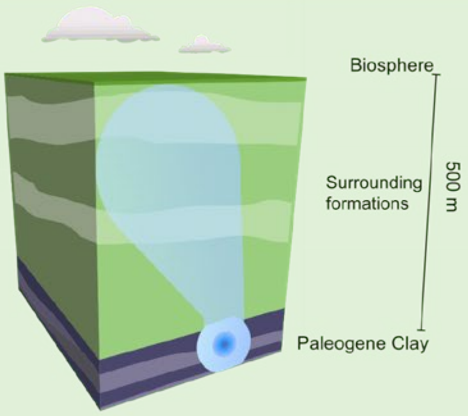
At this point, it is assumed that all the overpacks fail and the total glass surface area, including fractures in the vitrified waste form, becomes available for dissolution. The dissolution rate of radionuclides is determined by the cracking factor and the alteration rate of glass.

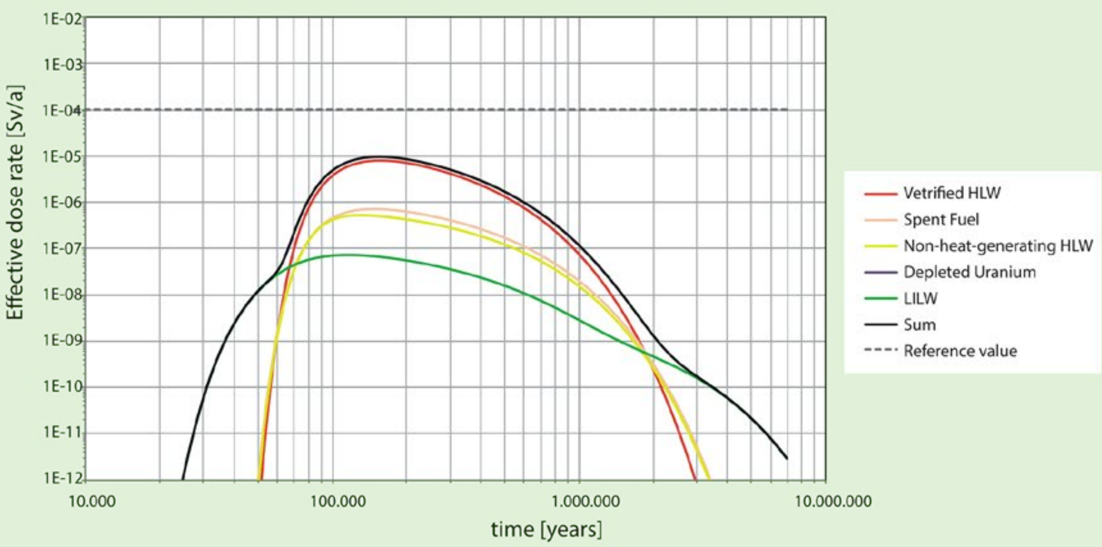
In the SA model used in OPERA, all the components in the disposal tunnels, including the waste packages and tunnel liner are modelled as a single homogeneous volume within which radionuclides are generated uniformly, at a rate determined by the dissolution rate of vitrified waste.

More and more radionuclides are released into the EBS due to gradual dissolution of the waste form. Mobile radionuclides are dispersed further into the clay host rock and then into the aquifer system, which can result in uptake in the biosphere.

100,000 - 1,000,000 years

The wastes that dominate the calculated exposures are vitrified HLW and SRRF, even though the volumes of these waste are relatively small compared to other wastes. The calculated peak exposure is about 10 µSv per year, at about 200,000 years into the future. This peak is ten times lower than the reference value of 0.1 mSv per year and about 150 times lower than average natural background radiation exposures.





The simplified geometry used in the safety assessments assumes that the full surface area of the vitrified waste form is in contact with pore water after the failure lifetime of the overpack. The illustration above summarises the simplified behaviour modelled in the OPERA safety assessment over each of the periods discussed above and can be compared with the previous illustrations of expected behaviour. Dissolved radionuclides from the EBS enter directly the clay host rock. The containment function of the clay host rock is for non-sorbing chemical elements such as selenium limited to its small permeability and reducing conditions. People are eventually exposed to releases, the highest contribution to the dose rate was calculated to be from selenium-79 released from the vitrified HLW. In COPERA, there is the waste form releasing radionuclides to concrete materials. All concrete materials have been assumed conservatively to have the same permeability as the porous cementitious backfill. If the permeability of the backfill is taken into account, the calculated contribution of the dose rate from vitrified HLW decreases by more than an order of magnitude. The second largest contribution comes from iodine-129 in spent fuel (SRRF), since this is also a mobile long-lived radionuclide. The contributions from non-heat generating HLW and LILW come from radionuclides that are cations, which were assumed in OPERA not to be retarded by the clay host rock. However, analysis of experimental measurements from clay host rocks, as well as available literature, shows that some retardation of these cations is likely. In the base case, despite the assumed high solubilities of uranium, the contribution to the dose rate from depleted uranium is negligible and therefore not visible at the scale used in the release calculations. This negligible contribution to the dose rate from depleted uranium is judged to be realistic.

How safe is the multibarrier system?

The safety assessment calculates the potential radiological impacts of the multibarrier system on the environment over the timescales discussed. The results are compared with indicators and reference values used for judging acceptable levels of safety. The assessment model splits the geological disposal system into compartments, evaluates radionuclide behaviour within each and calculates transfers between them.

The biosphere acts as the receptor for any radioactivity that moves upwards from the geosphere. Reference biospheres developed by the IAEA are used to determine how people might be exposed to radionuclides from the multibarrier system. A uniform temperate climate is assumed for the whole period of the OPERA calculations. This is considered adequate for the present preliminary safety assessment in this phase of the Dutch geological disposal programme.

The radiological impacts (radiation exposure or dose) of ingestion, inhalation and external radiation by radionuclides entering a well, surface water bodies (rivers, lakes, ponds) and wetlands is included in the reference biospheres. The modelled well is small, at shallow depth and supplies a family with all its drinking and other water, including water used for crop irrigation and livestock.

The calculated potential radiation dose to an individual is compared with a reference dose. In Dutch legislation, no dose constraints are yet defined for geological disposal, so the reference value has been set at 0.1 mSv per year, a value used in most other national programmes. The flux of radiotoxicity from the multibarrier system into the biosphere is another useful reference value; it can be compared to the flux from radionuclides naturally present in the overburden.

The bulk of the calculated total radiotoxicity in the system remains in the EBS and the clay host rock. About a tenth of the total radio-toxicity results from the depleted uranium, which remains within the EBS, where its low solubility and mobility continue to contain it. Only a tiny fraction of the radiotoxicity enters the overlying geological formations; by the time of peak releases to the biosphere at 200,000 years, this fraction represents only about one millionth of the activity that is contained within the multibarrier system. As expected in this disposal concept, the low permeability clay host rock and concrete in the multibarrier system represent the most effective barriers. In summary, within a few hundred thousand to a million years, almost all the radioactivity initially in the GDF has either decayed within the EBS or the clay host rock, only a tiny fraction has migrated out to be diluted and dispersed in the overlying formations and biosphere, and the multibarrier system has effectively performed its isolation and containment task.

The exception to this picture is depleted uranium. This comprises more than half the mass of the waste materials in the GDF but contains only about 0.2% of the total radioactivity at the time of disposal. However, its principal radionuclide (naturally occurring U-238) has a half-life of 4.5 billion years, which means that it does not decay perceptibly within tens of millions of years. In calculations run out to the very far future, uranium series radionuclides are the only significant contributors to exposures, but in the Normal Evolution Scenario (NES) these exposures occur only after some tens of million years into the future. A further key observation is that it is not possible to mitigate these exposures by any realistic optimisation of disposal system engineering. However, they are a minute fraction of natural background radiation doses and arise from what is effectively a natural material that, owing to its low mobility, is expected to remain within the geological environment. Investigations of the natural uranium already contained in the Paleogene clays are expected to elucidate the key processes for uranium migration and derive representative parameter values for the post-closure safety assessment.

Overall, even using pessimistic approaches, the performance assessment calculations for the NES show that potential radiation exposures to people in the future are orders of magnitude below those currently experienced by people in the Netherlands due to natural sources of radioactivity. Also, they would not occur until many tens or hundreds of thousands of years into the future. The calculated impacts for the NES are also well below typical, internationally accepted, radiation protection constraints for members of the public.

The NES represents the most likely evolution of the disposal system and remains the focus for calculations in the future, with other scenarios that address climate changes to be included in future post-closure safety assessments. Alternative evolution scenarios are less likely but need to be assessed, because they illustrate the redundancies in the multibarrier system, also in extreme climate states. What-if scenarios are not likely but contribute to the testing of individual barriers in the multibarrier system. Human intrusion scenarios have so far only studied radiological exposure to people working at drilling sites and not any public exposures. To minimise the probability of public exposures by human intrusion, COPERA made the choice to focus on saline Paleogene clays overlain and underlain by saline Paleogene sandy formations where there is little incentive for extraction of groundwater.

Can the GDF be optimised for post-closure safety?

Optimising the radiological protection provided by the multibarrier is an important objective for the future. In OPERA, all types of waste were proposed to be disposed of at the same disposal depth, but, as explained earlier, a multilevel design of the GDF has been developed. This depth segregation means that long-term interactions between degradation products of the different types of waste are less likely, and the sandy formations between the clay formations also enhance dissipation of degradation products such as gases.

The multilevel GDF also reduces the footprint occupied by the waste, so that the likelihood of human intrusion is further reduced. In any case, extraction of core samples of EBS and waste materials would lead any competent company or organisation to cease drilling, at least temporarily, and report to the relevant authority.

Conclusions

What is the feasibility of constructing the GDF?

The disposal concept is based on the well-developed Belgian GDF design for Boom Clay, but its construction in the Netherlands could utilise other available Paleogene formations with suitable properties over a wide depth range. While there is certainly flexibility in choosing an appropriate host formation (or formations), the detailed knowledge of the geotechnical properties of Dutch Paleogene clays at relevant disposal depths that we need to refine our designs and safety assessments is currently poor. During COPERA, good quality Paleogene clay cores have been obtained at 400 m. These cores are currently being investigated in SECUUR, a research project led by Delft University of Technology. More needs to be known about the nature and variability of Paleogene Clay properties and about the in-situ stress regime on a regional basis across the Netherlands in order to refine the current GDF layout concept, in which one transport tunnel intersects all the disposal tunnels at the same disposal depth. Existing tunnelling techniques using a tunnel-boring machine can be used to excavate the clay host rock.

The range in disposal depth for HLW considered is from 200 m to 1,000 m. The minimum disposal depth is sufficient for isolation of HLW. As the temperature of the clay host rock increases with depth, 1,000 m is currently expected to be a maximum, for an acceptable working temperature. Costs increase with increasing depth, predominantly due to the greater required thickness of concrete segments in the liner of disposal tunnels; this reduces the disposal volume so that the tunnel length needs to be larger. For mechanical stability, the spacing between the disposal tunnels also needs to be larger, requiring a larger transport tunnel. The disposal depth at which the costs of the GDF become prohibitive is yet to be calculated.

What is the feasibility of siting a GDF?

Siting studies are currently foreseen after 2050, but there is confidence today that suitable locations for a GDF in Paleogene clays with appropriate thickness and depth are available, but the data on their characteristics need to be improved. Significant uncertainties in depth-thickness distributions of Paleogene clays are present since most of the data originate from oil/gas exploration wells, where there has been little interest in characterising the clays.

A siting programme will need to avoid certain geological structures and features, and guidelines and criteria for doing this will need to be developed. Factors that will need to be taken into account include natural resources, variability of Paleogene clay properties, levels of seismic activity and evidence of past deep glacial erosion.

Future development of the concept will also depend on obtaining better data on regional hydrogeological and geomechanical properties of the formations overlying and underlying, the Paleogene clays. This will require access to boreholes and cores from relevant disposal depths. At the current programme phase, data from boreholes are required, not for commencement of a siting programme, but rather for achieving broader validation of some of the geoscientific assumptions.

Other potential GDF host rocks exist in the Netherlands, some of which have been evaluated in the past and all of which will be studied in more detail in the future. These include Zechstein rock salt, for which a COPERA Salt Safety Case has been developed, in parallel to this COPERA Clay Safety Case.

It is recognised by COVRA that siting a GDF involves considerably more than evaluating technical factors. Any future siting programme will need to take account of societal requirements and will be staged, progressive and consensual in nature.

Does the multibarrier system provide adequate safety?

The multibarrier system provides complete containment and isolation of the wastes during the first few hundreds to a few thousand years during which the hazard potential of the wastes is at its highest, but is decaying rapidly. Beyond 10,000 years, we expect that any residual radioactivity that escapes the degraded EBS will be contained by the clay host rock for hundreds of thousands to millions of years. A minute fraction of highly mobile radioactivity will move into surrounding geological formations on this timescale, but will be diluted and dispersed in deep porewaters and groundwaters, resulting in concentrations that cause no safety concerns and are well below natural levels of radioactivity in drinking water.

Other evidence underpinning safety

Natural and archaeological analogues of the preservation of materials in clays show that all degradation processes can be much slower than typically modelled. The preservation of ancient woods for millions of years in Neogene clays in Italy (see image next page) and Belgium is a good example of how the absence of groundwater flow and the presence of anoxic conditions contribute to very long-term preservation, even of fragile organic material. The 2,000 year preservation of Roman iron objects in similar anoxic conditions (see image next page) supports the assumptions on the minimum longevity of the carbon steel overpack of the waste package for heat-generating HLW. Roman cements and concretes show that the massively concrete-dominated engineered barrier system can maintain its physical properties and structural stability for thousands of years.

Natural radioactivity, present in all rocks, soils and waters around us, provides a useful yardstick against which to compare the impacts of any releases from the multibarrier system. The unavoidable natural radiation exposures to which we are all subject are higher than those from even our pessimistically calculated releases. We live in, and human-kind has evolved in, a naturally radioactive environment.

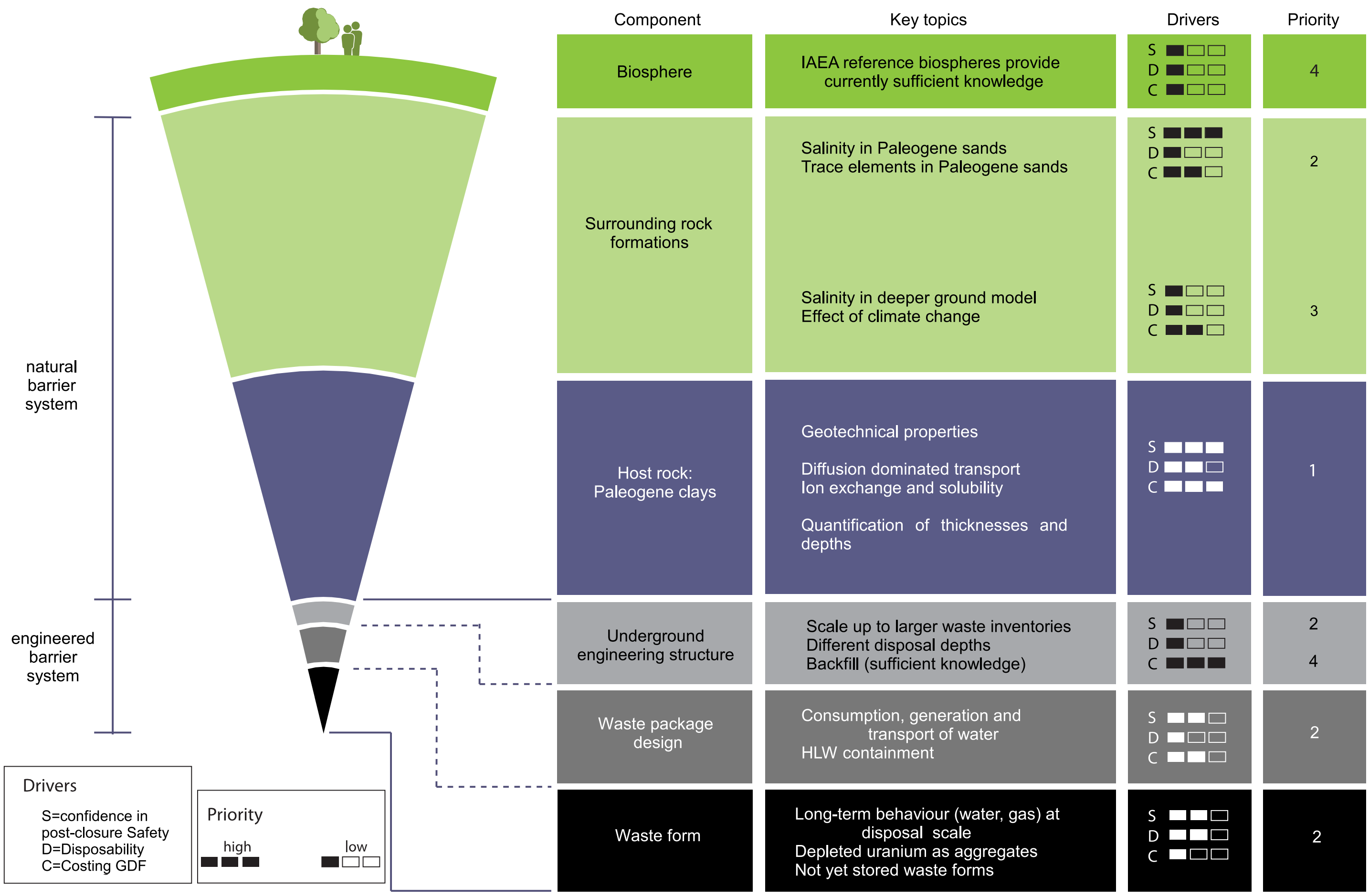
Confidence in the reliability of the OPERA safety assessment calculations is also enhanced by the fact that they are broadly similar to those estimated independently for a wide range of wastes and host rocks, in other national programmes.

Improving the design and Safety Case

A number of processes and scenarios that could affect or alter the NES have not yet been treated and thus constitute open issues that will require further R&D and safety assessment. The principal uncertainties that have been identified will be addressed by future studies. Not all of the work is required in the next decades; some will be staged over several iterations in COVRA's long-term research programme. A roadmap that starts with the identification of the key topics that need to be addressed in future work has been developed for this future RD&D. The illustration below shows these key topics for the main components in the disposal system, along with the drivers for carrying out further work and the priorities currently attached to each component. The highest priority is associated with obtaining further information on the Paleogene clays.

Awareness of the GDF design concept and its requirements in terms of depth, area and geological conditions will facilitate fitting this facility into national planning policies and priorities for the use of underground space. At present, there are good prospects for disposing Dutch radioactive waste within the Paleogene clays, but more data need to be collected on its properties and their variability at relevant depths.

The existence of COPERA and its findings are important contributions to satisfying the Netherlands' obligations under both EC Directive 2011/70/EURATOM and the IAEA Joint Convention, showing that substantial progress has been made on the national programme. The project also supports the Netherlands' position of carrying out a dual-track (national and potential multinational) policy for radioactive waste management. The results can be used as the Netherlands' contributions to the development of multinational projects.



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RADIOACTIE

How COPERA can increase confidence, enhance knowledge and guide future work

1. Introduction

1.1 Purpose and context of the present report

The principal objective of this report is to present an overview of results and conclusions of the on-going work in the Netherlands on developing safety cases for a Geological Disposal Facility (GDF). One of the options being examined is for a GDF in a Paleogene Clay formation. A major milestone in the clay studies was reached in 2017 with the publication of an initial Safety Case based on the R&D work completed in the OPERA research programme.
It is foreseen that safety cases will be made by COVRA every 10 years and updated after 5 years. The present work is part of COVRA's on-going COPERA programme and the report updates and expands the OPERA clay safety case, taking into account progress in the Netherlands and elsewhere in the intervening years. The progress made in the clay studies is mostly related to improved understanding of the physical and chemical processes determining disposal safety. However, significant effort has also been put into examining more closely the practicability and efficiency of construction and operation of a GDF; this explains why the present report title refers to both safety and feasibility. Our intent to ensure that the report can be read as a stand-alone document means that information that remains the same as in the 2017 OPERA Safety Case has been brought forward from that report and amended with updated information only as necessary.

In addition, a parallel study on the safety and feasibility of disposal in a GDF in salt has been prepared. Again, this is a stand-alone document, so information common to both clay and salt studies (e.g. on the Dutch waste inventory) is included in both reports.

1.2 Why do we need geological disposal of radioactive waste?

Radiation and radioactive materials occur naturally. The human population is continuously exposed to ionizing radiation from cosmic rays incident on Earth and from radioactive nuclides that were generated during Earth's formation and are still present in Earth's crust. However, human activities associated with the use of nuclear technologies can generate more radioactive materials or can concentrate those already present in nature. All these radioactive materials must be properly managed whilst they are in use, in order to protect people and the environment - and they must be safely disposed of, if and when they become radioactive wastes.
A radioactive material becomes radioactive waste when no future use is foreseen for it. Radioactive wastes can arise during the generation of nuclear power and the production of medical isotopes, or from activities using radioisotopes in research, education, diagnostics for human health and examination of the integrity of engineered constructions. The wastes must be managed in a way that ensures safety and security at all times. Radioactivity naturally decays over time, so that safety can be achieved by ensuring that the wastes are isolated from the human environment until they no longer pose a hazard. The period of time for which the wastes must be isolated depends on the radioactive half-lives of the radionuclides in the waste. It can range from a few days, for wastes containing only very short-lived radionuclides, to more than 100,000 years for some long-lived wastes.

The necessary levels of isolation are initially achieved by containing the radioactive waste in safe and secure storage facilities. Storage of radioactive waste in surface facilities for periods up to many decades is a proven safe technology and is applied globally. Nonetheless, this storage method is not a long-term or final solution for wastes that remain radioactive for very long times. Over such long periods the necessary continued active monitoring, inspection, security and maintenance cannot be assured. For waste that remains hazardous for thousands to hundreds of thousands of years, the acknowledged approach to ensuring passive long-term isolation and containment is disposal in a stable geological environment, deep enough beneath Earth's surface of to exclude disruptions due to near-surface processes and events. The waste is emplaced and sealed in a GDF. Geological processes in the deep underground occur at slow and predictable rates over very long periods of time. At the current state of science and technology, geological disposal is the only solution that can ensure no radioactive elements will ever return to the human environment in concentrations that can be harmful.

1.3 Dutch policy on Radioactive Waste Disposal

The Dutch disposal programme is at an early conceptual stage, but COVRA can learn from waste management organisations that already, after decades of work, have selected a design concept and a site for geological disposal of waste. Their experience teaches us that technical progress alone will not result in successful implementation of a disposal project. Progress also depends on projects being accepted and embraced at the societal level, as being necessary and appropriate. Disposal projects run for many decades with several generations involved. The most successful national programmes have transparent schedules and activities, and they publish the key dates for policy decisions and the justification for these. Figure 1-1 shows the key decisions and dates in Dutch policy related to disposal of waste, including the requirements for disposal concepts. Since 1982, COVRA has been charged with the task of implementing this policy. The figure also shows the different disposal research programmes that have been carried out in the Netherlands. Details are given in Appendix 1. Here, we look at the highlights and explain the context in which the current report has been produced.

In 1984, a formal Dutch policy for the storage of both chemically toxic waste and radiologically toxic waste was established based on the following three objectives: Isolation, Control and Surveillance/ Monitor (In Dutch: Isoleren, Beheersen en Controleren: the so called IBC-principle). At that time, disposal of radioactive waste in the Netherlands was proposed to be in rock salt formations (Winsemius and Kappeyne van de Coppello, 1984). The first national research programme, OPLA, thus considered only rock salt formations and it included disposal concepts without controlled emplacement of waste packages and closure of the GDF (OPLA, 1989, 1993). A policy requirement for the retrievability of waste was later introduced (Alders, 1993) and, accordingly, the second national research programme, CORA, was focussed on disposal concepts allowing retrieval of waste packages[1].

At the request of the Dutch government, the research programme was extended to include clay formations in addition to rock salt formations (CORA, 2001). OPLA and CORA were coordinated by the Dutch Geological Survey, but COVRA was tasked with coordinating the third national programme, OPERA. This programme focused on Boom Clay, a Paleogene clay formation, and ran from 2010 to 2017. It was aimed at attaining a level of knowledge of disposal of waste in a clay GDF equivalent to that resulting from earlier work on salt. In 2017, OPERA produced the initial Dutch safety case for disposal of radioactive waste in clay (Verhoef et al., 2017).

1. In order to comply with the EU Waste Directive (EC, 2011), the definition of disposal of waste was changed into: the emplacement of waste in a facility without the intention to retrieve the waste (Kamp and Teeven, 2013). Nevertheless, in the operational phase of a GDF, retrievability of waste packages can be facilitated by design.

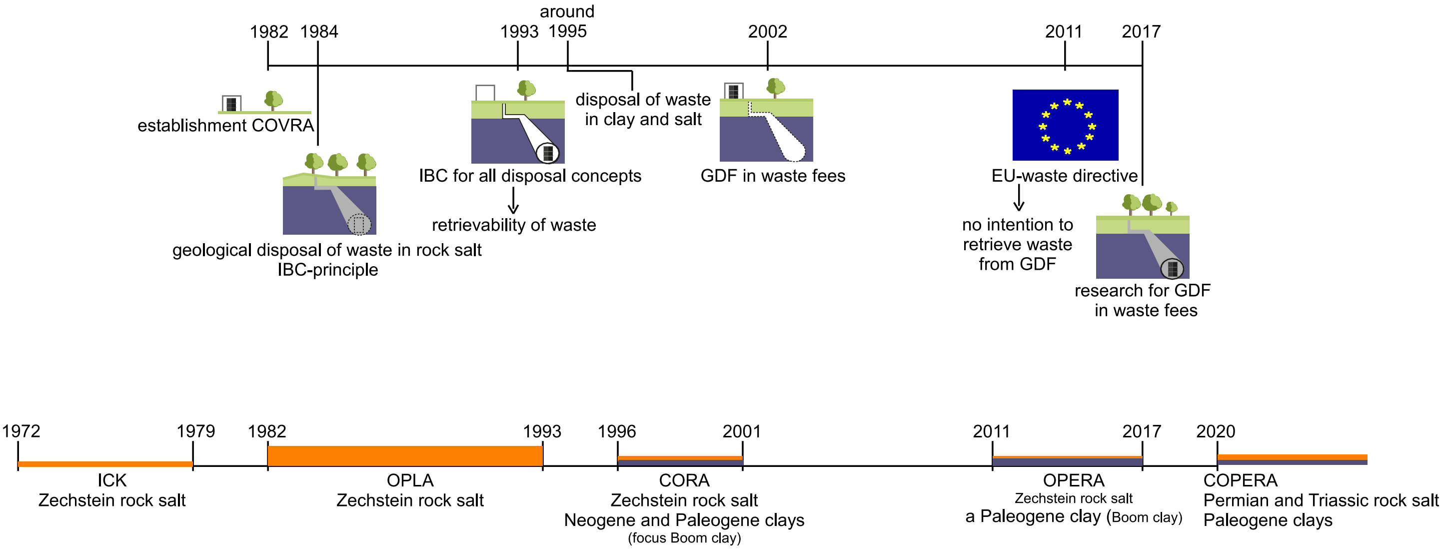


Figure 1-1: Key dates in Dutch policy (top) and research programmes (bottom) for geological disposal of radioactive waste. A more detailed image of the handling of radioactive waste in the Netherlands can be found in Berkers et al. (2023).

The Netherlands have now chosen to structure future work on disposal of waste through the national radioactive waste management programme prepared in the framework of the European Council Directive for safe management of spent fuel and radioactive waste in the EU (van Veldhoven-van der Meer, 2018). COPERA is the acronym for COVRA's on-going research programme into geological disposal of waste. The safety and feasibility studies presented in this report are key components of COPERA and they will be updated at intervals of five years. The following three drivers are used to prioritize the specific research for a GDF in the Netherlands (Verhoef et al., 2020; Verhoef et al., 2017):

- Demonstrate the feasibility and safety of disposal of the different types of radioactive waste;
- Enhance confidence in the post-closure safety provided by the multibarrier system of engineered and natural barriers;
- Improve scientific understanding and optimise costs for a GDF.

1.4. Role of the GDF Safety Case in the Dutch Programme

1.4.1 What is a safety case?

'Safety Case' is a common term applied in many industries where potential hazards to workers and the public must be assessed. In studies on the geological disposal of radioactive wastes, it has been used widely for over two decades, both in national programmes and in the documents of the International Atomic Energy Agency (IAEA), the European Commission (EC) and the Nuclear Energy Agency (NEA). Development of the safety case concept for geological disposal of waste has been documented in NEA and IAEA publications describing the nature and purpose of safety cases (NEA, 2013) (IAEA, 2011a, b, 2012). It has also been addressed in international safety standards and guides and there have been major symposia devoted to the topic.

The currently most widely accepted description of a safety case for geological disposal was formulated by the IAEA in 2011 and reproduced in a 2013 NEA update. The concise definition used in COPERA is from the IAEA Safety Standards for Disposal of Radioactive Waste (IAEA, 2011a).

"The safety case is an integration of arguments and evidence that describe, quantify and substantiate the safety, and the level of confidence in the safety, of the geological disposal facility".

In the context of the present report, several key additional generic points concerning safety cases can be made and their relevance to the Dutch case explained:

- Safety cases are made at various stages in a programme for disposal of waste, reflecting the progress towards implementation and the improvements in scientific understanding, so that an iterative process is necessary. OPERA was the first iteration of a safety case for a GDF in the Boom Clay of the Netherlands. In COPERA, all Paleogene clay formations are considered.
- At earlier stages, key data may be incomplete or not yet sufficiently accurate. This can be seen clearly in the present report, where many data are not yet available - sometimes because they will be obtained through future work, sometimes because they will be site specific, and no location is intended to be selected for the Dutch GDF for many decades.
- In this situation, a safety case can make conservative, well-founded assumptions and then show that these still allow safety goals to be met. As was the case for OPERA, this approach is also followed in COPERA. Later work may therefore lead to higher estimated levels of safety and/or to design modifications.
- Accordingly, the assumptions currently made by COVRA must be clearly stated and the approaches to confirming their validity laid out. For this reason, the final Chapter of the present report outlines the roadmap for future work on radioactive waste disposal in the Netherlands.
- A safety case made under these conditions can be characterised as a 'conditional safety case.' The OPERA Safety Case was labelled as such. The present COPERA Clay Safety Case and feasibility studies are also conditional, and explore the impact of improved system understanding and newly acquired data on the OPERA Safety Case.

1.4.2 Principle objectives of COVRA safety case and feasibility reports

Because these studies mark major milestones in the Dutch radioactive waste management programme, they cover a wider and somewhat different scope from the safety cases in other more advanced disposal programmes. Their principal objectives are:

- To propose practical conceptual designs for a GDF and to examine their engineering feasibility;
- To assess post-closure safety based on these designs;
- To use the design information to provide a basis for estimating future costs and therefore the level of financial provisions to be made by COVRA;
- To use the experience gained in producing the report to strengthen the national competences in scientific and technical areas related to geological disposal;
- To use the finding of the reports to select the R&D activities to be carried out in the Dutch disposal programme over the coming years;
- To inform decision-makers, the public and the scientific/ technical community about the progress of geological disposal planning in the Netherlands.

1.4.3 Why has the present safety case been produced?

In current Dutch policy, a final decision on the disposal method will be taken around 2100[2] and the start of disposal is expected around 2130 (I&E, 2016). This planning provides time to learn from experience in other countries, to carry out research and to a ccumulate the knowledge needed to make well-founded decisions. COVRA will iterate its conditional safety cases during the next decades. Site-specific safety cases are currently foreseen after 2050 (Verhoef et al., 2020).

The Dutch programme is in the conceptual phase of geological disposal development, with different disposal concepts and

2. The currently assumed period of 30 years between the decision on the disposal concept to be implemented and the start of emplacement of waste packages in a disposal facility may be too ambitious, based on experience in the most advanced programmes today (e.g. Finland, Sweden, France). The Advisory board of the ANVS therefore recommended that the decision on the disposal method should be taken earlier than around 2100, in order to start emplacement of waste in a facility in 2130 (de Vries, 2019). In addition, many experts and stakeholders in the dialogue sessions held in the framework of the Dutch national programme questioned the societal and ethical justification for postponing decision-making to 2100 (van Rooijen et al., 2023).

potential host rocks being considered. The government lays out the boundary conditions for geological disposal of waste. The implementer (COVRA) establishes the safety strategy using the boundary conditions and carries out preliminary assessments of post-closure safety. Post-closure safety should be provided by a system of natural and engineered barriers. Regulatory review of the work at this stage should guide the implementer on the likelihood of achieving the necessary demonstration of safety (EPS, 2016).

The implementation of the European Directive on radioactive waste management (EC, 2011) requires an evaluation of the national programme every decade. The last Dutch national programme was published in 2016 and is currently being evaluated. A revision of the national programme needs to be completed in 2025 (Heijnen, 2022). This safety case and the broader Requirements Management System (RMS) in which it is embedded have been developed to help this evaluation.

1.5 What's new or different in COPERA 2024 compared to the OPERA 2017 safety case?

1.5.1 Wider range of disposal concepts

OPERA focussed on a GDF concept based on disposal in a tunnel system constructed in the Paleogene Boom clay formation in the Netherlands. In COPERA, other Paleogene clay formations are also considered. This increases the range of potential siting regions that might be available in the Netherlands[3]. The greater choice of geological formations also implies that alternative disposal facility designs become feasible, including the multi-level option considered in the present report.

1.5.2 Design changes to enhance feasibility

The emplacement of waste packages in the GDF considered in the OPERA Safety case would be difficult to implement unless currently available techniques are improved. Tunnels with a larger diameter are therefore considered in this COPERA study. Detailed consideration of emplacement techniques is important in the safety case in order to provide confidence that waste packages can be efficiently emplaced and retrieved if necessary. The infrastructure requirements at the surface as well as the configuration of tunnel intersections in the GDF, have been updated to ensure feasibility. In addition, radiation protection calculations have been initiated to demonstrate that operational safety can be provided during the emplacement of waste packages.

1.5.3 New data on clay cores

The geotechnical properties of clay formations constrain the design of the tunnel structure. With the current state of knowledge, these properties need to be determined from freshly cored clay or well-conditioned clay cores. During OPERA, Boom clay cores extracted from the underground at depths between 63 and 80 m (PCR, 2013) were studied (e.g., Behrends et al. (2015). In COPERA, in the framework of research for geothermal wells, Paleogene (Ypresian and Landen) clay cores were obtained at a depth of about 400 m in Delft (Vardon et al., 2022).

3. The parallel COPERA Salt report further expands the scope of future regional siting studies.

15.4 Expansion of the COVRA RMS System

COVRA is developing a Requirements Management System (RMS) that will structure all of its activities from waste conditioning, through temporary waste storage to disposal operations, including ensuring that safety is provided after closure of the GDF. The OPERA Safety Case gave some information on the RMS at its highest hierarchical levels. Further levels are defined in the COPERA study, taking into account the need to be compatible with the parallel safety case in salt, and also with COVRA's waste storage programme. The requirements on the chosen geological host rocks and engineered barriers are defined in the RMS and all of the changes between OPERA and COPERA will be recorded in the system.

1.5.5 Revised waste inventory and packaging assumptions

The waste inventory has been updated using the latest estimation in the national programme (Burggraaff et al., 2022). The current Dutch nuclear power policy implies that several waste inventory scenarios are possible, and these are described in COPERA. In addition, new packaging assumptions have resulted in a significant decrease in the space required for disposal of some wastes. The disposal volume for HLW, for the same waste inventory scenario as in OPERA, has been reduced by one third in the COPERA study. Resulting cost estimates for disposal of the inventory in a waste scenario similar to OPERA are included in this safety case.

1.5.6 Progress in the assessment basis for post-closure safety

The safety assessment shows if and how releases of radionuclides may take place and to what extent these releases can be harmful. This assessment is therefore the backbone of a safety case. The following aspects have been considered in more depth in the current COPERA Clay Safety Case:

- **Enhanced understanding of transport and retention of radionuclides in clay host rock**
  Clays contain natural radionuclides and also non-radioactive isotopes of radioactive elements that are present in the waste. In this study, the impact of these naturally occurring nuclides on potential releases is considered, using detailed studies on transport and retention mechanisms. This work includes initial studies on how the natural radionuclides and non-radioactive elements are distributed throughout the deep clay formations. These analyses aim to provide some validation of the models and data used in the safety assessment and give a perspective on the transport of radionuclides released from the waste (see section 5.1.6.4).
- **Detailed study of degradation of the engineered barriers**
  The potential degradation of barriers and the releases of radionuclides from the waste are controlled by the availability of water in the multibarrier system and by the reactions taking places at interfaces. The impacts of surface films and of the availability of water is assessed in this safety study in order to determine whether these may limit the alteration rate of the waste forms and the resulting radionuclide release rate from the wastes. These studies are described in sections 5.1.6.2 and 6.1.3.
- **More use of natural and archaeological analogue data.**
  Information on the longevity of buried man-made materials and on the behaviour of natural systems that are analogues of barriers can enhance confidence in the analyses of their

future evolution. Reference is made to relevant analogues when assessing the key processes determining the impact of the geological and engineered barriers in the multibarrier system (see Chapter 5, 6, 7 and 8).

1.5.7 Embedding of the research in a continuous programme

Previous Dutch national disposal programmes OPLA, CORA and OPERA were all carried out with specified durations. COPERA is an on-going programme envisaged to run for decades. Consequently, research to improve system understanding and to gather data will continue. The long-term research programme has a structure that can be used for several programming periods, each of which will result in an iteration of the safety case. Each Work Package contains a set of tasks. Figure 1-2 shows how the work packages are functionally related to each other.

Work package 0 concerns all tasks related to programme management and coordination. Work packages 1 and 2 have a more strategic and integrative character. WP1 is related to strategic aspects, such as estimating costs and exploring shared solutions and other strategic options. WP2 covers the integration of the knowledge obtained through the research programme and the production of safety cases in rock salt and poorly indurated clay.

Work packages 3 to 6 are structured around key topics that need to be studied in order to produce these safety cases; these are related to the components of the multibarrier system and are different for the specific host rocks, poorly indurated clay and rock salt.
The tasks (projects) in these work packages differ in each programming period and they contain the main research activities of the programme. Appendix 2 shows all work packages, tasks and publications related to clay as a host rock, to which COPERA (2020-2025) has contributed by the time of production of the current report, in 2024.

The last work package (WP 7) covers all interactions with the wider public, including education, communication and public participation in the long-term research programme. An example is COVRA's contribution to the recently updated Geology of the Netherlands which is frequently used by university students. For COVRA, transparency about the long-term research programme is very important. Results have been and will be published online. People interested, but not involved in the research programme, are able to access documentation about it online. The knowledge generated in the research programme and consolidated into reports is published on COVRA's website to foster the dissemination of results. COVRA has also encouraged researchers to publish their work in scientific journals (preferably Open Access), as well as in popular scientific magazines. This allows the non-academic community to freely access the scientific knowledge that has been obtained by the research (co)funded by COVRA.

1.5.8 Relationship of COPERA to European Joint Programming

Some important aspects of COVRA's research programme are carried out in an international framework. The preparation of COVRA's long-term research programme started around the same time as the European Joint Programming initiative of the EC (Gaus et al., 2019). A EURAD Consortium was established in which 51 organisations from 23 EU Member States were mandated as participants by their official National Programme Owner. In the Dutch case, COVRA was mandated as a Waste Management Organisation. Dutch research projects were mainly performed in the framework of work on clay host rock. In EURAD-1; COVRA focussed on research on the engineered barriers system embedded in clay host rock. That work concerns the following work packages: ACED (**A**ssessment of **C**hemical **E**volution of ILW and HLW **D**isposal cells) and MAGIC (chemo-**M**echanical **AGI**ng of **C**ementitious materials). EURAD-1 also included further clay host rock studies: GAS (mechanistic understanding of **GAS** transport in clay materials) and FUTuRE (**Fu**ndamen**t**al **u**nderstanding of radionuclide **re**tention).

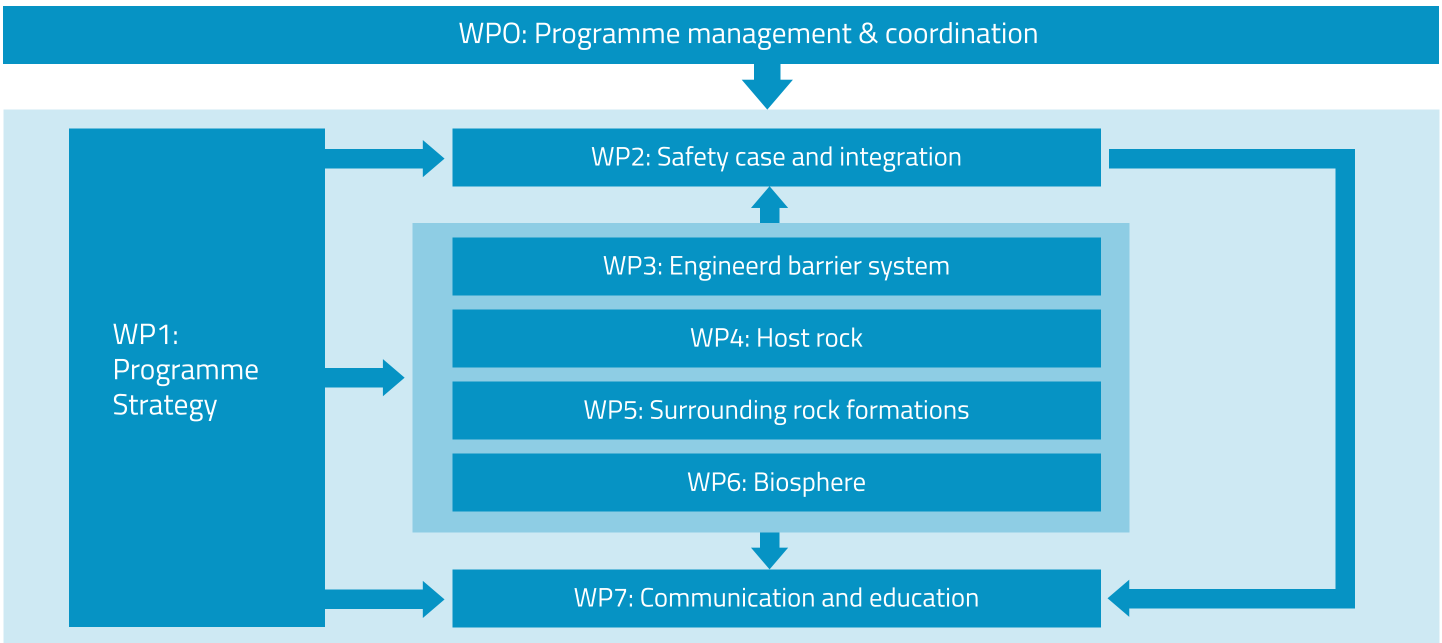


Figure 1-2: Overview and relation between the programme's work packages (Verhoef et al., 2020)

Figure 1-3 shows the coherence between the tasks related to the clay disposal system and the research and development (R&D) effort for the first programme period of 5 years (Verhoef et al., 2020).

1.6 Structure of this safety case report

This is a stand-alone document describing the COPERA safety case work, the feasibility studies, and the associated research that has been, and is currently being carried out.

- Chapter 2 summarises the concept of geological disposal of waste and provides an international perspective on the current status of waste disposal, specifically in clay host rocks.
- Chapter 3 describes the approach used to assess post-closure safety, the structure of the safety case, the different requirements for geological disposal of waste and the contributions to safety of different components in a multibarrier system with clay host rock, and how these change over time, with the details reserved for Chapter 7.
- Chapter 4 describes the Dutch waste inventory to be disposed of and summarises the disposal concept. The estimated costs of GDF implementation are also presented.
- Chapter 5 discusses the natural barrier system composed of the clay host rocks and the surrounding rock formations. It covers the selection of clays that are currently investigated, the evidence for their properties and their past evolution.
- Chapter 6 gives a description of the engineered barrier system and how the requirements for this system have been derived.
- Chapter 7 describes the processes affecting the evolution of the multibarrier system and the scenarios that are considered in assessing the post-closure safety.
- Chapter 8 reproduces some of the calculated safety assessment results that were obtained in OPERA and then discusses how taking into account updated parameter values and also additional processes considered in COPERA could impact on the calculated doses.
- Chapter 9 provides a synthesis of the COPERA clay studies and draws conclusions.
- Chapter 10 provides a description of the research work that is currently being performed and will be performed in the future.

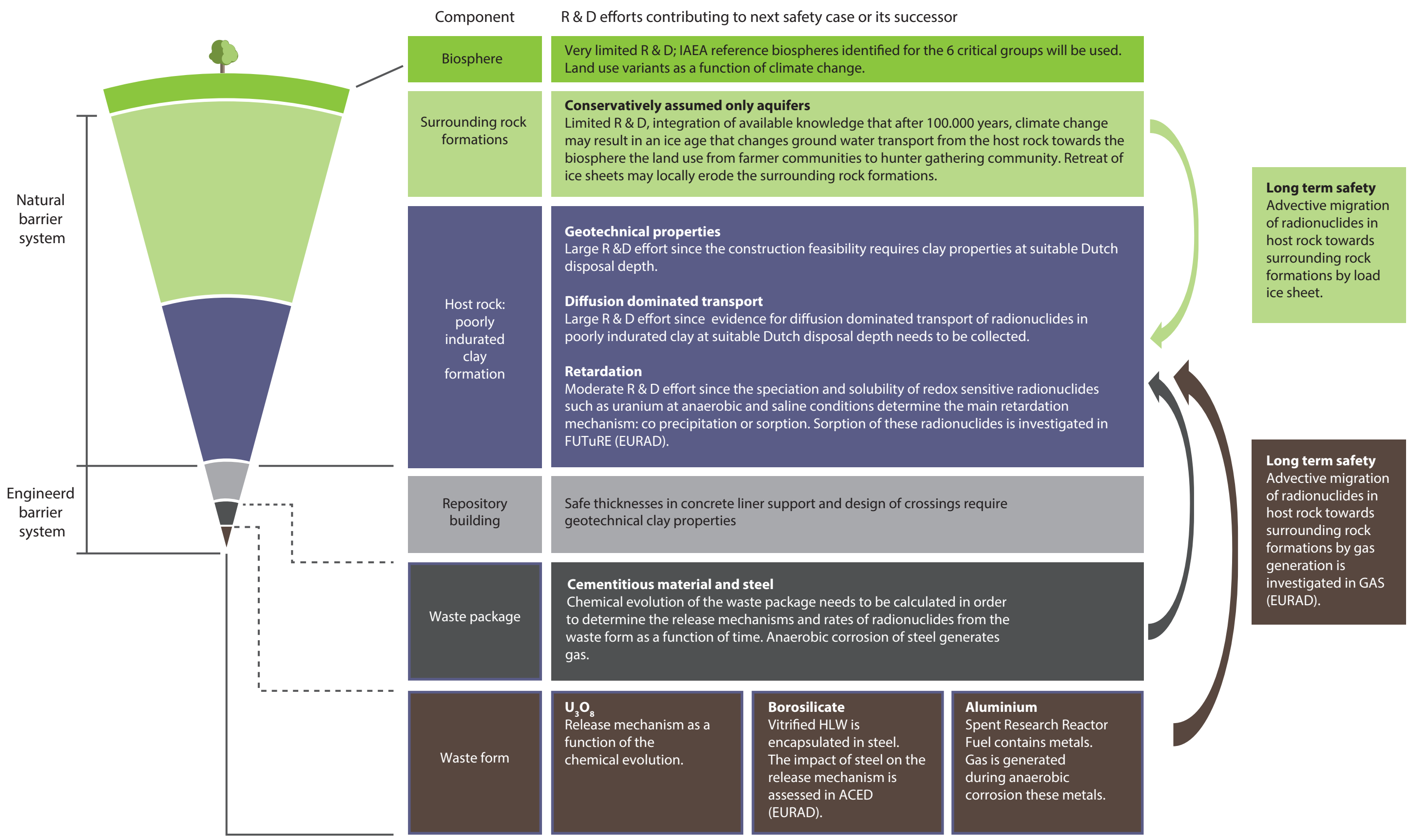


Figure 1-3: Coherence between tasks, the disposal system and the updated safety case for poorly indurated clays (Verhoef et al., 2020)

2. Geological Disposal of radioactive waste

This chapter describes the concept of geological disposal of radioactive waste, covering its objectives, showing how components in a multibarrier system contribute to post-closure safety and describing the practical activities to be carried out throughout the long period from planning through to closure of the Dutch GDF - a period that may last from several decades to a century or more. The international perspective on the current status of geological disposal of waste that was included in the OPERA Safety case has been updated, primarily to focus on disposal of waste in clay host rock, as new developments have occurred in Switzerland and France since the publication of the OPERA Safety case in 2017. At the request of the Dutch Ministry of Infrastructure and Water Management, the Rathenau Institute is working on societal issues in waste disposal in the framework of the national programme, and their publications since 2017 have been integrated in this Chapter as well.

2.1 Disposal objectives

Geological disposal aims to remove hazardous materials from the immediate human and dynamic, natural surface environment to a stable geological environment deep underground where they will be protected from disturbance by natural or human processes. Waste packages are emplaced in a deep underground facility constructed in a suitable host rock and this facility is closed and sealed. The concept of geological disposal of waste has been firmly established internationally for more than 40 years, based on the use of a so-called 'multi-barrier system', in which a series of engineered and natural barriers acts in concert to isolate the wastes and enclose the radionuclides that they contain (IAEA, 2011a):

- ISOLATION: removes the wastes safely from direct interaction with people and the environment. In order to achieve this, locations and geological environments identified for a GDF must be deep, inaccessible and stable over long periods (for example in formations where rapid uplift, erosion and exposure of the waste will not occur) and should be unlikely to be drilled into or excavated in a search for natural resources in the future.
- CONTAINMENT: means retaining the radionuclides within the multibarrier system until natural processes of radioactive decay have reduced the potential hazard considerably. For many radionuclides, a multibarrier system can provide total containment until they decay to insignificant levels of radioactivity within the waste packages. However, the engineered barriers in a multibarrier system will degrade progressively over hundreds and thousands of years and eventually lose their ability to provide complete containment. Because some radionuclides decay extremely slowly and/or are mobile in water, their complete containment for all times is not possible in groundwater-bearing formations. Assessing the safety of geological disposal involves evaluating the mobilisation and transport of these radionuclides and their potential impacts if they eventually reach people and the surface environment, even in extremely low concentrations and many thousands of years into the future.

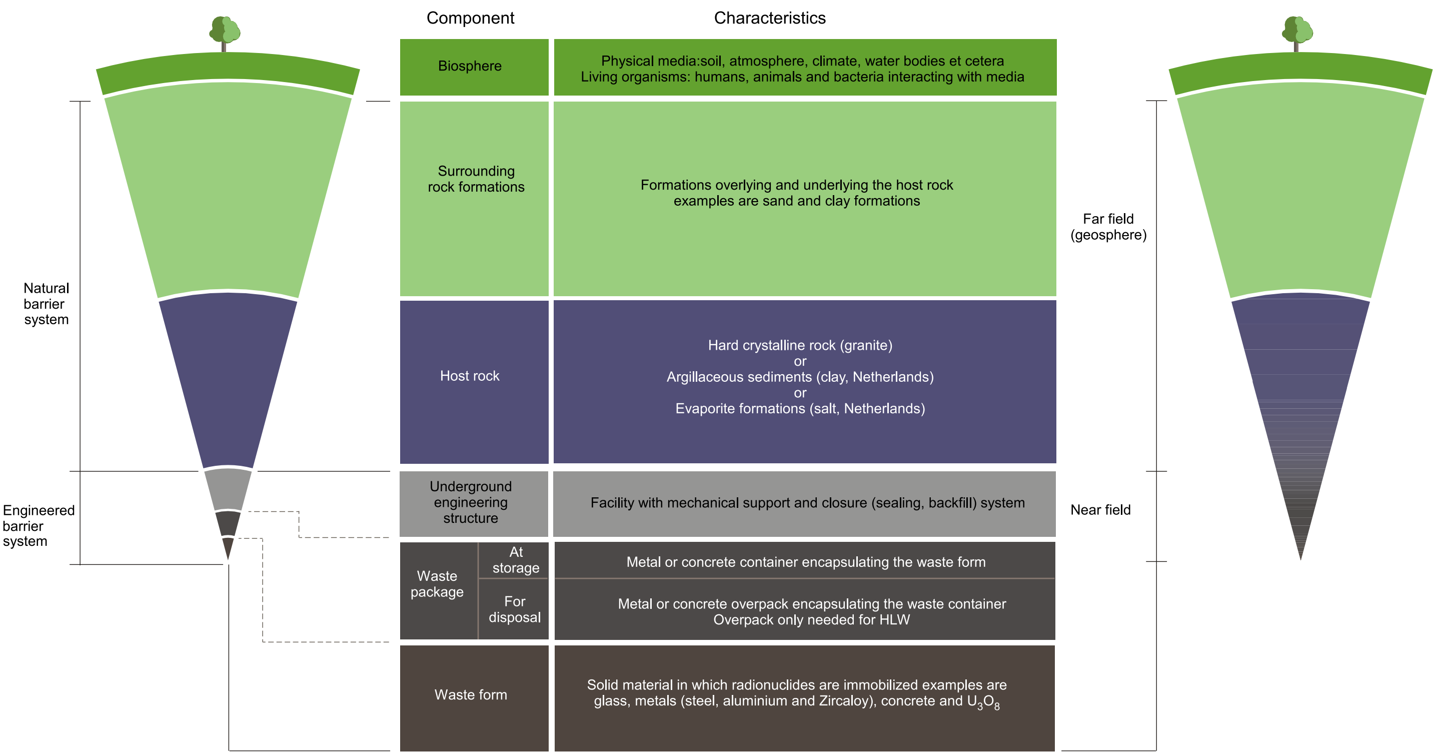


Figure 2-1: Components of multibarrier systems at the time of completion and closure of the geological disposal facility, adapted from the OPERA Safety case.

Over the very long-term, the safe performance of a multibarrier system thus depends on the balance between the rates of radioactive decay and the rates of processes involved in radionuclide mobilisation and transport through the rocks and groundwaters of the natural barrier system.

Six components are distinguished in a multibarrier system (see Figure 2-1). The three, inner components are engineered barriers and lie within the so-called 'near-field' of the multibarrier system. The 'far-field' is comprised of the natural barriers: the host rock and the surrounding rock formations. Each of the components in the multibarrier system contributes to ensuring isolation and containment. A generic set of such contributions to post-closure safety, adapted from Chapman and Hooper (2012), is shown in Table 2-1.

The relative contributions to safety of the various barriers at different times after closure of a disposal facility and the ways that they interact with each other depend upon the design of the multibarrier system. The design itself is dependent on the geological environment in which the facility is constructed. Consequently, the multi-barrier system can function in different ways at different times in different disposal concepts. The safety concept for a multibarrier system with clay host rock is described in Chapter 3. The multibarrier system shown in Figure 2-2 distinguishes between the engineered barrier system (EBS) and the surrounding natural barriers: clay host rock (blue) and surrounding rock formations (green). Box 2-1 discusses the declining radiotoxicity of wastes as a function of time, showing that this radiotoxicity reduces by factors of many thousands over a period of some hundreds to a few thousands of years, depending upon the waste type. Providing safe isolation and containment over this 'early' period of the highest hazard potential is perhaps the most important role of a multibarrier system. It is expected that the operational life of the GDF would be many decades, depending on how much waste already exists in storage when the facility becomes operational, and how much is to be produced in the future. An essential aspect of the multibarrier system is that it provides protection and safety in a completely passive manner. After a GDF is completed and closed, no further actions from people are required to manage the wastes. Over immensely long times, the engineered barriers and the wastes become part of the deep, natural environment, with conditions in the clay host rock returning to those of the natural, undisturbed environment before the GDF was constructed (see Chapter 7).

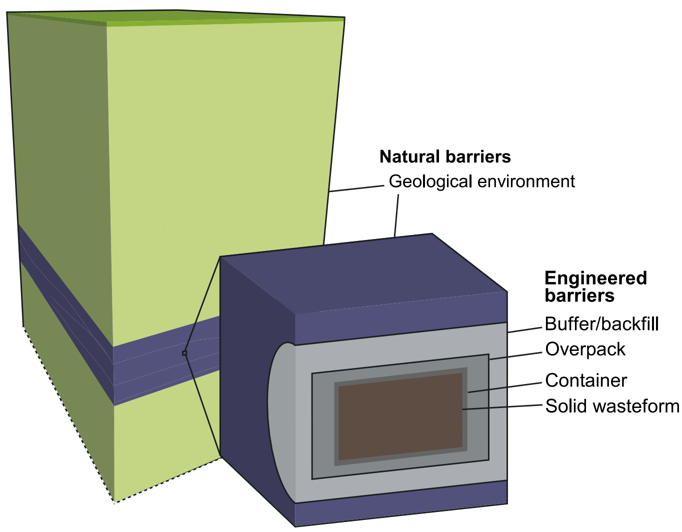


Figure 2-2: The general concept of the multiple barrier system for geological disposal of radioactive waste in a clay formation: adapted from Chapman and Hooper (2012) and adapted as presented in the OPERA Safety case.

Barrier component	Generic contributions to post-closure safety
Waste form: the solid waste material	▪ provide a stable, low-solubility matrix that limits the rate of release of the majority of radionuclides by dissolving slowly in groundwaters that come into contact with it
Waste container: generally metal or concrete: for higher activity wastes the container might have an outer metal overpack	▪ protect the waste form from physical disruption (e.g. by movement in the bedrock) ▪ prevent groundwaters from reaching the waste form for a period of time ▪ act as a partial barrier limiting the movement of water in and around the waste form after corrosion has breached the container ▪ control the redox conditions in the vicinity of the waste form by corrosion reactions, thus controlling the solubility of some radionuclides ▪ allow the passage of any evolved gases from the waste form out into the surrounding engineered barrier system
Buffer or backfill: material around the waste container, separating the package from the rock. In many designs, a natural clay buffer (bentonite) is used	▪ protect the waste container from physical disruption (e.g. by movement in the bedrock) ▪ control the rate at which groundwaters can move to and around the waste container (e.g. by preventing flow) ▪ control the rate at which chemical corrodents in groundwaters can move to the waste container ▪ condition the chemical characteristics of groundwater and pore water in contact with the container and the waste form so as to reduce corrosion rate and/or solubility of radionuclides ▪ control the rate at which dissolved radionuclides can move from the waste form out, into the surrounding rock ▪ control or prevent the movement of radionuclide-containing colloids from the waste form into the rock ▪ suppress microbial activity in the vicinity of the waste ▪ permit the passage of gas from the waste and the corroding container out into the rock
Mass backfill: filling material for access and service openings. Various natural materials and cements in different parts of the GDF, chosen to be compatible with the geological environment	▪ restore mechanical continuity and stability to the rock and engineered barrier region of the facility so that the other engineered barriers are not physically disrupted (e.g. as a clay buffer takes up water and expands) ▪ close voids that could otherwise act as groundwater flow pathways within the facility ▪ prevent easy access of people to the waste packages
Sealing systems: emplaced locally in tunnels and shafts at key points in the system	▪ cut off potential fast groundwater flow pathways within the backfilled facility (e.g. at the interface between mass backfill and rock) ▪ prevent access of people into the backfilled facility
Natural geological barrier: the host rock in which the waste emplacement tunnels or caverns are constructed and all the overlying geological formations, which might be different to the host formation	▪ isolate the waste from people and the natural surface environment by providing a massive radiation shield ▪ protect and buffer the engineered barrier system from dynamic human and natural processes and events occurring at the surface and in the upper region of the cover rocks (e.g. major changes in climate, such as glaciation) ▪ protect the engineered barrier system by providing a stable mechanical and chemical environment at depth that does not change quickly with the passage of time and can thus be forecast with confidence ▪ provide hydrogeological rock properties that, together with low hydraulic gradients, limit the rate at which deep groundwaters can move to, through and from the backfilled and sealed facility, or completely prevent flow ▪ ensure that chemical, mechanical and hydrogeological evolution of the deep system is slow and can be forecast with confidence ▪ provide properties that retard the movement of any radionuclides in groundwater - these include sorption onto mineral surfaces and properties that promote hydraulic dispersion and dilution of radionuclide concentrations ▪ allow the conduction of heat generated by the waste away from the engineered barrier system so as to prevent unacceptable temperature rises ▪ disperse gases produced in the facility so as to prevent over pressures leading to mechanical disruption of the engineered barrier system

Table 2-1: Contributions to post-closure safety of the principal barriers in multibarrier systems: adapted from Chapman and Hooper (2012), as presented in the OPERA Safety case.

2.2 Different options for GDF host rocks

Over the last 45 years, geological disposal of waste has developed from a concept to reality. The world's first GDF for spent nuclear fuel is currently being licensed for operation in Finland, and others are in advanced stages of siting and development in France, Sweden, Switzerland and China. In that period, most countries have focussed their attention on three broad groups of rocks as host formations:

- **Hard 'crystalline' rocks** such as granite, gneiss and other metamorphic or plutonic rocks can be extremely stable, especially with respect to future erosion (e.g., by ice sheets) and are generally easy to construct in, allowing large, stable underground openings to be used for waste emplacement. Extensive worldwide studies have been performed on hard crystalline rocks of varying compositions and ages, including ancient Pre-Cambrian shield rocks (e.g., in Canada, Sweden and Finland).
- **Argillaceous sedimentary rocks** such as clays, mudstones and marls can provide a high level of physical containment owing to their low permeability, which can lead to their pore-waters remaining essentially immobile, with little or no groundwater flow occurring through them on timescales of interest for post-closure safety. This characteristic has been demonstrated in the Jurassic and Paleogene clay formations being targeted in France, Switzerland and Belgium, using environmental isotopic and chemical compositional profiles of their pore waters (Mazurek et al., 2011).
- **Evaporite formations** are principally dome and bedded salts, with the principal host rock of interest being halite.
These formations, although they can be structurally and compositionally complex in the case of domal salts, are often cited as providing ideal containment properties. In homogeneous regions of either bedded or dome formations, there is essentially no fluid that is sufficiently mobile to transport radionuclides to the surrounding rock formations.
These formations were the first to be identified as potential hosts for radioactive waste disposal, as long ago as 1950 (NRC, 1957), and have been studied in the Netherlands as well as several other countries, including Germany and the USA. A parallel study describing COVRA's current preliminary safety case for dome salt as a host formation for the Dutch waste inventory is published together with this report.

Each of these groups has its own strengths, advantages and challenges with respect to containment and isolation. There is also a wide range of variability of these strengths within any one group and between specific sites that have been investigated for disposal internationally. It is recognised that safety can be achieved by different balances of these characteristics and strengths of the safety functions of the natural, geological barrier, so that there is no unique solution that is the 'best rock' or the 'best environment'.

Over the last 45 years, a range of generic, but host rock specific, GDF designs has been developed around the world and a range of materials proposed for various components of the EBS. Both the design and the materials selected depend upon the category of waste to be disposed of and the geological environment under consideration. In some countries, there is a preference for a single GDF for all wastes that require geological disposal, with separate sections that have different designs to accommodate the different wastes. Many further design considerations are involved in fitting a generic concept to a specific site, including the ability to be flexible and adapt design, depth and geometry to local conditions in order to exploit the best volumes of rock or to avoid certain geological features. This provides scope for optimising operational procedures and costs, accommodating local community requirements and minimising environmental impacts of construction and the operation of surface facilities and the GDF.

2.3 Activities through the lifecycle of a GDF

The major phases of activity through the lifecycle of a geological disposal facility (Figure 2-3) are site selection, construction, operation and closure. There is relevant international experience on design and/or implementation for each of these phases. The role of a safety case at each stage is described in Chapter 3.

2.3.1 Site selection

Selecting a suitable location for the Dutch GDF is an activity that lies decades in the future. At the request of the Dutch Ministry of Infrastructure and Water Management, the Rathenau Institute is developing policy advice on how the Netherlands can best organise the decision-making process for siting the GDF (Cuppen, 2022); work will be informed by extensive European experience on radioactive waste governance (van Est et al., 2023). The institute has already concluded that aiming for decision-making only around 2100 negatively impacts people's perception of the need for actions today, making public participation a complex challenge. They advise establishing criteria allowing reservation of potential locations for a GDF and emphasise the importance of embedding the role of public participation within research and various national and decentralized political decision-making processes (Dekker et al., 2023).

COVRA assumes that the siting strategy will be based on a volunteering model, incorporating stakeholder involvement at all

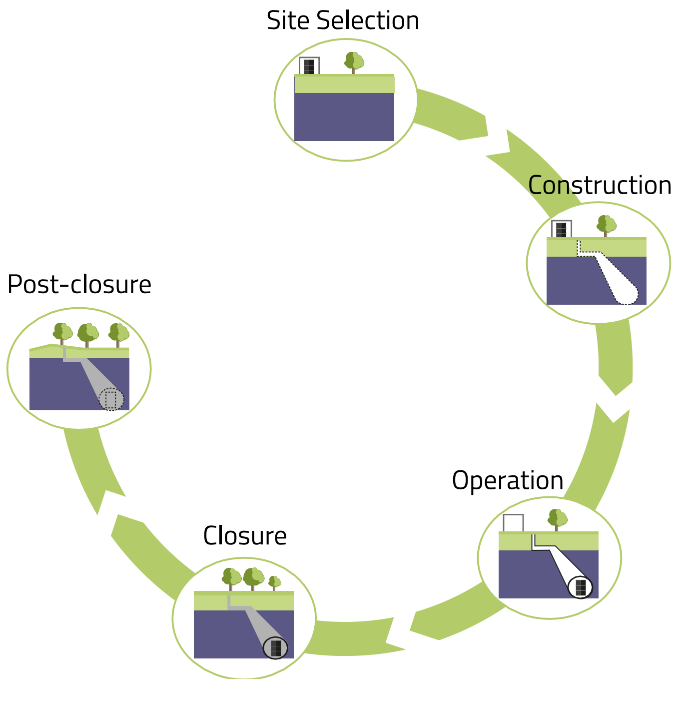


Figure 2-3: Phases in the lifecycle of a geological disposal facility, as presented in the OPERA Safety case.

stages. It would be technically guided at the outset only insofar that clearly unsuitable regions are excluded at the start. For example, a relevant geological criterion could be that candidate sites should have a potential host rock formation that shows no evidence of past local, deep glacial erosion, because potential similar future events could impact post-closure safety. It is considered important today that the eventual siting strategy will incorporate the flexibility to evaluate objectively any proposal that might emerge from volunteer communities or regions. A visualisation of what a site selection process might involve is described below, but no process has yet been established in Dutch policy. We use examples from disposal programmes in clay host rocks, including the most recent international developments and their associated time frames.

Many national geological disposal programmes have suffered setbacks and delays because their GDF siting projects have proved difficult or impossible to implement. In general, this is because it has proved hard for implementers to prepare and present the appropriate mix of technical, societal and political inputs that is required to achieve consensus amongst the stakeholders. However, the recent success of several national programmes indicates that this problem can be overcome, largely by recognizing that siting needs to be an open and inclusive process for all parties concerned.

Gathering technical information to help identify suitable regions and, eventually, specific locations, involves iterative programmes of data evaluation and site investigation to characterise the geological environment in sufficient detail. At each stage, information is generated in progressively more detail, to refine the design of the GDF and to improve the system modelling that is central to the post-closure safety assessment. Generally, GDF design and safety evaluation will go through several cycles of development, as more, and more specific, information becomes available. The basic geological and geotechnical characteristics of the host rock and surrounding formations must be adequately understood and, for the safety case, an integrated picture must be built up of the dynamic evolution of the deep environment during tens and hundreds of thousands of years.

This requires the compilation and interpretation of information gathered by many field, laboratory and remote sensing techniques, at a wide range of spatial scales. This will involve the use of data available from other geotechnical, survey and exploration activities in the Netherlands, plus dedicated deep drilling, testing and sampling in boreholes. Identifying, scoping and managing technical uncertainties will be a key activity within the siting programme. Underground Research Facilities (URFs) can be involved in a site characterisation process. Such a research facility can greatly increase confidence that radioactive waste can safely be disposed of at the URF site.

For disposal of waste in clay host rocks, following pioneering work in Belgium, major developments leading to siting have been made in the past 6 years in France and Switzerland. An Underground Research Facility (URF)[4] has been constructed at Bure (France) in Callovo-Oxfordian clay at a depth of 490 m and has been operating since 2002 (see Figure 2-4). Sufficient confidence had arisen that the subsurface around Bure is suitable to construct, operate and

4. ANDRA's preference is to name the underground laboratory in Bure a URL instead of a URF in order to distinguish the current and past activities from the future activities. The IAEA has a URF network for several decades in which the progress in the implementation is shared. We use the IAEA term URF in this safety case.

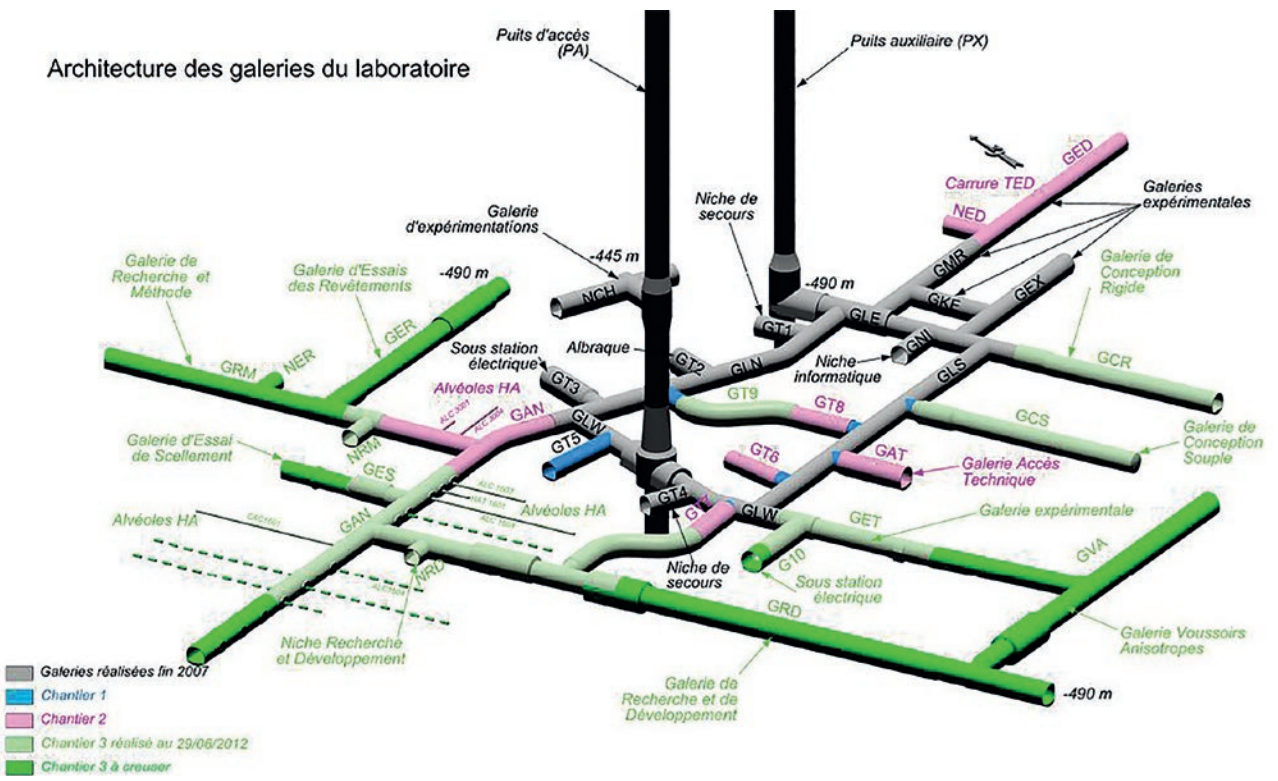


Figure 2-4: The underground laboratory in Bure in northeast France (source: French Waste Management Organisation ANDRA) from which characterisation work and research have provided sufficient confidence to select Bure as a site for geological disposal of waste.

close a disposal facility at a depth of around 500 m. The disposal facility (called Cigéo) will be constructed close to the URF at Bure and was declared an 'infrastructure of public utility' in 2022. This declaration allows ANDRA to adapt, as needed, other infrastructure in the immediate vicinity of Cigéo (new railways, electricity, roads). The start of construction is currently foreseen for 2027.

After two decades of work on both granitic and clay disposal concepts, the formal government-led Swiss site selection process started in 2008 with a Sectoral Plan. Preliminary site characterisation activities to dispose of waste in Opalinus Clay finished in 2022. All characterisation activities were performed after gaining public acceptance. Three regions were investigated, including drilling boreholes and surface geophysical surveys. Out of these three regions, NAGRA, the Swiss waste management organisation, proposed one region, with the justification that it has:

- the least geological deformation (the fewest major faults) and thereby has the most available space to dispose of waste;
- the highest effective containment characteristics in the Opalinus clay formation and therefore provides the best quality as a natural barrier. All regions had a similar clay content and thickness, but one region has the largest proportion of very old pore water, implying a less dynamic deep geological environment.
- a deeper clay formation that provides better protection against glacial erosion.

The submission of a general licence application by NAGRA is foreseen in 2024. It is expected that the process of consideration, review and granting of this licence application will take about ten years. The initial construction work for underground geological investigation is therefore foreseen in 2034. The envisaged on-site URF includes demonstration and confirmation experiments (NAGRA, 2021).

2.3.2 Constructional phase

Construction work starts after a construction license has been obtained from the relevant authority. For EU Member States, the procedure also includes consultation with and obtaining an opinion from the European Commission, as required by the Euratom Treaty (Carbol et al., 2022). Globally, there are currently no geological disposal facilities being constructed in clay host rocks, although France is close to implementation, as discussed above.
However, the experience gained in construction of URFs in clay host rocks is expected to provide a sound technical basis for planning construction of a GDF. The first of these URFs in clay were constructed in the previous century, including those at Mont Terri (Switzerland) in Opalinus Clay since 1996 and at Mol (Belgium) in Rupel (Boom) Clay since 1980. URFs provide understanding of how clay host rocks behave at depth and direct experience of which techniques to construct, operate and close a disposal facility are feasible. The considerable knowledge gained over many years means that the pioneering role of these generic laboratories is lessening and future URFs are likely to be mainly part of the deep rock characterisation stages at actual GDF sites.

Various techniques are available for the mechanised excavation of tunnels and other underground openings in clay, including road headers and tunnel boring machines, and there is considerable experience worldwide in their use in sedimentary rock formations over a range of strength properties and depths. For construction in the poorly indurated Palaeogene clays that we are considering in the Netherlands, a tunnel supported by a lining similar to that already used in Boom Clay in Mol (Belgium) is envisaged (see Figure 2-5). The Netherlands has considerable experience with this tunnel construction technique, in which concrete segments are installed immediately after excavation of the clay (see Chapter 4), although experience is limited to traffic tunnels, where there is fast convergence of the poorly indurated clay at their relatively shallow depths.

In indurated clays, such as the Callovo-Oxfordian clay selected as the host formation in France, immediate emplacement of concrete liner segments is not necessary for short-term support. A cementitious support of sprayed concrete (shotcrete) is sufficient, but decades of experience in using shotcrete of different compositions shows that is does not provide sufficient stability in indurated clays for the very long period over which a GDF might remain operational. The French waste management organisation has therefore decided to construct tunnels using concrete segments for both accessways and for the tunnels in which Intermediate Level Waste is to be disposed (ANDRA, 2016): see Figure 2-5).

2.3.3 Operational phase

At present, there are no GDFs operational in clay host rock, but there is growing international experience in operating underground disposal facilities in other rock formations and this is applicable to any type of GDF. This includes experience from the operational Waste Isolation Pilot Plant (WIPP) facility in rock salt in the USA for ILW, and from disposal facilities for short lived low and intermediate level waste (SL-LILW) in granitic host rocks in Hungary, Finland and Sweden. SL-LILW is also currently disposed of in surface facilities in many European countries such as France, Spain, Bulgaria and the Czech Republic. Underground disposal facilities, surface disposal facilities and storage facilities at the surface (such as COVRA's own storage facilities at Nieuwdorp) use similar approaches to waste handling, techniques for the emplacement of waste packages, and overall active facility operational management.

2.3.4 Closure phase

The disposal tunnels with emplaced waste may either be backfilled directly after emplacement of waste in the operational phase or backfilled in the closure phase. Plugs in shafts and other accessways may be required to prevent ingress of water into the disposal facility and seals may be needed to maintain the isolation of the waste. In plastic rocks, the requirements for seals are less demanding than in hard rocks (Carbol et al., 2022). Methods for the closure of a disposal facility have also been demonstrated in trials in URFs. One example is the casting of a high-strength concrete plug in granitic rocks in the Aspö URF (Sweden), to produce a barrier that would minimize the possibility of human intrusion.

Depending on the design concept, some of the installations in the underground and surface facilities (e.g., cranes used for package handling) need to be dismantled and removed before closure and some components may need to be either decontaminated or disposed of as active waste. Site remediation activities allow the site to be returned to normal use (Carbol et al., 2022). There are currently no closed GDFs; but there are closed surface disposal facilities, such as the Manche disposal facility in France, which operated for 25 years (ANDRA, 2020).

2.3.5 Post-closure phase

The post-closure phase begins with a period of active institutional control over access to and activities at the disposal site. This is primarily intended to increase the level of confidence in the isolation and containment provided by the multibarrier system. Active institutional control is not required to assure long-term safety, as the multiple barriers function as an entirely passive system at all times after closure, but it does help to prevent or minimise the probability of inadvertent human intrusion into the multibarrier system, so long as monitoring is maintained. The active institutional control period may extend over several decades, depending on the national regulations and licence requirements in place. Eventually, at some point in time to be agreed by future generations, active institutional control will be terminated.

Passive institutional control primarily consists of record keeping and preserving knowledge on the waste, the disposal facility and the site. Propagating this knowledge into the future will require a range of provisions to be made with local, national and international organisations. The longer that knowledge of the GDF can be preserved and communicated, the greater the reduction in the hazard potential of the wastes by decay and the lower the likelihood and consequences of inadvertent human intrusion.

The scope and duration of institutional control must be defined in national regulations and requirements set out by the regulatory body, but in many countries such requirements have not yet been fully defined (Carbol et al., 2022).

Closed Dutch chemical waste disposal facilities, such as Volgermeerpolder nearby Broek in Waterland (Berkers et al., 2023; SBV, 2023), are useful analogues or examples of long-term institutional control.

Figure 2-5: Constructed parts of underground facilities with segmented concrete liners in poorly indurated clay (left; source EURDICE) at 225 m depth in Mol (Belgium) and in indurated clay (right) at 490 meters depth in Bure (France).

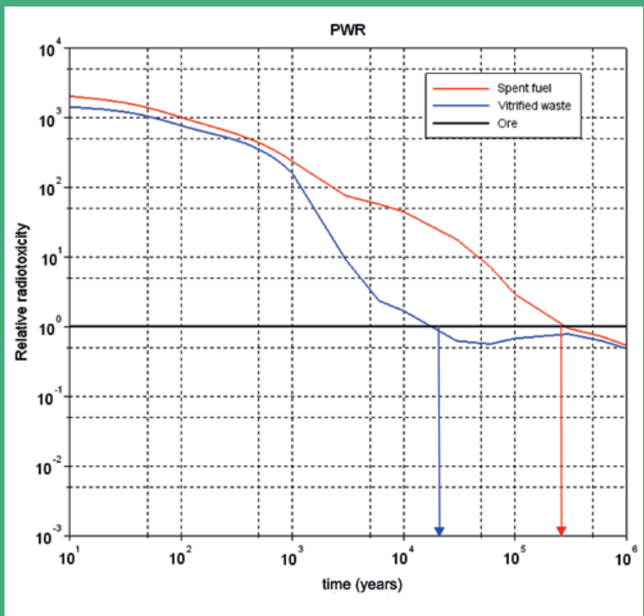
Box 2-1: Addressing the long-time scales in the safety case

There is a commitment among those managing radioactive wastes to ensure safety at all times to levels at least as protective as those provided today. This has meant looking farther into the future than has been attempted for any other engineering project - not just a few generations (the design life of most engineered structures), but tens of thousands of generations. Typical safety assessments model potential impacts on future generations out to a million years - a timescale that is hard to imagine for most people. However, even such an immense period of time is relatively short for a geologist used to considering how our natural environment has evolved over hundreds of millions of years. The long times over which we wish to provide protection are put into a different perspective when we consider our ability to characterise and understand natural geological processes occurring deep below the surface over much longer periods. This is what underpins the concept of geological disposal and provides confidence in the achievable safety.

Of course, forecasting the future behaviour of a disposal system for such long times brings with it increasing uncertainty as we look farther into the future. The level of uncertainty depends on the particular geological environment being studied, the materials used in the multibarrier system and the physical and chemical processes being evaluated. For some materials or processes, we can only be confident in our predictions of behaviour for thousands of years. For others, particularly many geological processes, we can have confidence in our predictions for hundreds of thousands, or even millions of years.

Furthermore, radioactive wastes exhibit one key characteristic that sets them apart from many other hazardous materials and that puts the issue of the long timescales in a different perspective - owing to the natural process of radioactive decay, their radioactivity reduces with time. If the multibarrier system prevents radionuclides returning to the human biosphere for sufficiently long, they will no longer pose hazards for humans. The rate and scale of reduction in radioactivity is known exactly and depends on the radionuclides contained in the wastes. Because much of the original activity in the most radioactive categories of COVRA's waste is due to radionuclides that decay relatively quickly (e.g., Sr-90 and Cs-137, whose activity halves every 30 years), most of the activity disappears within the first thousand years. This early decay in radioactivity significantly reduces concerns about the long timescales that are being considered. However, the potential impacts of longer-lived radionuclides must also be taken into account - and this is a central aspect of the safety assessment in Chapter 8. It is important, therefore, to consider in more detail how the total radioactivity of the wastes changes with time.

In practice, when considering the potential impacts of radionuclides on people, it is their 'radiotoxicity' rather than their radioactivity that is more relevant, since the radiation dose (in Sv) from ingesting a given amount of a radionuclide (in Bq) differs between radionuclides. The radiotoxicity of a given amount of waste is thus a measure of the radiation doses that would result if all the radionuclides in a given amount of waste were to be dissolved in water which was then drunk by a person (Hamstra, 1975; Hamstra and van der Feer, 1981). This situation is entirely hypothetical, but it does allow comparison of how hazardous different types of radioactive materials can be. For example, it allows comparisons between the radiotoxicity of spent fuel or HLW and the radiotoxicity of the natural uranium ore from which the fuel was produced. An example of the calculation of the relative radiotoxicity of spent nuclear fuel from a pressurized water reactor (PWR, such as the Borsele nuclear power plant) is shown in the figure below, which also shows the radiotoxicity of vitrified HLW resulting from reprocessing such spent nuclear fuel (Gruppelaar et al., 1998). The figure plots the declining radiotoxicity of spent fuel and vitrified HLW as a function of time after the fuel has been taken out of the reactor or, for vitrified HLW, after it was manufactured, following the reprocessing of the equivalent quantity of spent fuel. These curves are shown normalised to the radiotoxicity of the amount of uranium ore that was originally used to make the fuel (the horizontal line). To determine the radiotoxicity of each radionuclide Gruppelaar et al. (1998) used slightly different dose conversion factors to determine the radiotoxicity of each radionuclide than those used by Hamstra (1975) and Hamstra and van der Feer (1981), since the International Commission on Radiation Protection (ICRP) has since then updated the radiotoxicity data.



With a burn-up of 47,500 MWd/t, spent fuel is more radiotoxic than the uranium ore[5] from which it was manufactured for a period of about 200,000 years. At present, direct disposal of spent fuel from power reactors is not considered in the Netherlands, so the more relevant curve in the figure is that for vitrified HLW (the principal part of COVRA's higher activity waste inventory). In the reprocessing process, the long-lived uranium and plutonium are removed and recycled to manufacture more nuclear fuel. It can be seen that the resulting HLW is more radiotoxic than the uranium ore only for a period of for around 20,000 years. By this time, the large reduction in hazard potential that has occurred means that the primary functions of the multibarrier system have largely been achieved, by isolating and containing the waste until it presents a hazard potential similar to materials found in nature and, specifically, to those materials from which it was originally manufactured. Of course, it must also be acknowledged that uranium ores themselves can present hazards and that the wastes are now in a different location from the original ores. Accordingly, the safety case still needs to consider the possible impacts on people and the environment of the residual radionuclides that do not decay for very long times. These are predominantly radioisotopes of the heavy elements such as uranium, neptunium and plutonium, and of fission products such as I-129, Tc-99 and Se-79. However, the former group is strongly retarded in the clay host rock, and the fission products, although mobile in groundwaters, have low radiotoxicities (Chapman and Hooper, 2012).

This illustrates that, in the design and safety assessment of a multibarrier system, it is essential to ensure that complete isolation and containment are achieved over the first hundreds of years after closure. In the early period after closure, it is appropriate to judge possible health impacts on people using normal radiological protection standards. In the longer term, the hazard potential is much less, and in the very long-term we are dealing with something similar to naturally radioactive materials. Consequently, as the timescale increases beyond a few tens of thousands of years and out to a million years, it becomes more appropriate to assess hazards using other measures, more related to our daily exposure to natural radioactivity.

5. Note that the radiotoxicity of U-238 is central when comparisons are made to uranium ore. The calculations shown here use 2.4 × 10^{-7} Sv/Bq for the dose conversion coefficient for uranium-238, from ICRP-68 (1994). ICRP-74 (1996) proposed 1.2 × 10^{-6} Sv/Bq: i.e., uranium-238 was then believed to be more radiotoxic. The impact in cross-over time for spent fuel with a burn-up of 50 MWd/ton is a reduction from 170,000 years to 130,000 years (Magill et al., 2003). An even higher dose conversion coefficient for uranium-238 was used by NAGRA (2002): 2.5 × 10^{-6} Sv/Bq. NAGRA estimated a cross-over time for vitrified HLW relative to uranium ore at about 2000 years. The latest ICRP-119 (2012) report proposes again 2.4 × 10^{-7} Sv/Bq, as proposed previously in ICRP-68 (1994). This change makes the calculations by Gruppelaar et al. (1998) with a cross-over time of about 20,000 years for vitrified HLW, currently the most relevant one, and this is different from the one presented in the OPERA Safety Case from Chapman and Hooper (2012), which is closer to the value estimated by NAGRA. Nevertheless, the main message that the hazard potential diminishes over many thousands of years and should match requirements for containment, is unaffected.

How the role of the safety case changes in different stages of GDF development

3. Approach to showing post-closure safety

As explained in Chapter 1, demonstration of the safety of a multibarrier system is achieved through the preparation of a series of safety cases that are assembled sequentially, at key phases of programme development. The present Chapter explains in more detail the safety strategy for disposal, the structure of the safety case prepared by COVRA and the roles the evolving safety case will play throughout all phases in the lifecycle of a geological disposal facility. The safety strategy is designed to satisfy national and international requirements. Since the publication of the OPERA Safety case in 2017, the COVRA requirements management system has been made compatible with the parallel safety case in salt and also with COVRA's waste storage programme.

The principal safety-relevant impacts of the multibarrier system are calculated in terms of radiation doses that might be received by people in the distant future. To put this into context, the following section describes the permissible dose targets or limits that have been laid down in regulations.

3.1 Required levels of safety

In order to establish that a multibarrier system will not give rise to unacceptable impacts on people, agreed limits for such impacts must be defined. Calculating the consequences of potential releases of radionuclides from a multibarrier system is, in principle, a purely technical challenge. Judging whether the calculated releases would be acceptable to people is, however, also a societal issue. The most common metrics for quantifying radiological impacts are calculated radiation doses or risks. To assess whether adequate safety has been achieved, these doses and risks are then compared with regulatory limits or targets. As yet, no regulatory criteria have been defined explicitly for the implementation of a GDF in the Netherlands. However, European radiation protection criteria and standards have been established by Council Directive 96/26/Euratom and Member States must comply with this Directive (EC, 2014).

The EU radiation protection criteria and standards are derived from the recommendations made by the International Commission on Radiological Protection (ICRP), in particular those made in 2007 in ICRP Publication 103 (which sets down a limit of 1 mSv per year for the total dose to any member of the public from any regulated source) and in 2013, in Publication 122, which proposes a lower constraint of 0.3 mSv per year for a GDF (ICRP, 2013). In OPERA, a still lower limit of 0.1 mSv per year was proposed (Hart and Schröder, 2017), since this has been adopted in various national programmes.

To give some perspective on these numbers, it can be noted that the average total natural radiation exposure to a person living in the Netherlands is much higher, averaging 1.7 mSv per year (Smetsers and Bekhuis, 2021). This total radiation exposure in the population is monitored and periodically updated by the National Institute for Public Health and Environment (RIVM), which analyses prevalent exposure pathways and uses radionuclide-specific dose conversion coefficients set, and periodically updated, by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), and by ICRP. As a consequence, calculated values vary a little, periodically - for example, a value of 1.6 mSv per year was estimated at the time of the OPERA Safety case. In fact, natural exposures of people living in the Netherlands are significantly below the global average value of about 2.6 mSv per year.

Figure 3-1 shows the different average contributions to radiological exposure by natural radionuclides, with a short description of exposure pathways, as well as the additional average exposure to radiation from medical diagnostics, which raises the average exposure from 1.7 mSv per year (natural background only) to a total value of 2.8 mSv per year.

Radionuclides with a primordial origin have an important contribution to our natural background radiological exposure:

- Uranium-238, uranium-235 and thorium-232 occur in various concentration in all rocks and minerals and these all decay to radionuclides that generate radioactive radon, a noble gas. Radon is emitted from building materials containing these radionuclides and can subsequently be inhaled by people. The resulting radon dose is the largest single contributor to our average radiological exposure by natural radionuclides: see Figure 3-1;
- Potassium-40[6] also occurs in various concentration in all rocks and minerals and is mainly responsible for the external gamma-radiation from soil and building materials at home, and also provides the largest contribution to doses from the ingestion of food (Cinelli et al., 2019; Smetsers and Bekhuis, 2021).

The multiple barrier system for geological disposal of waste is designed to contain the artificially generated radionuclides in the wastes to such an extent that any additional radiological exposures that might arise for people in the future are negligible compared to the natural radiation exposures to which they will be subject.

6. Potassium-40 is the largest source of natural radioactivity in animals including humans. A 70 kg human body contains about 140 g of potassium, hence about 140 g × 0.0117% ≈ 16.4 mg of ^{40}K.

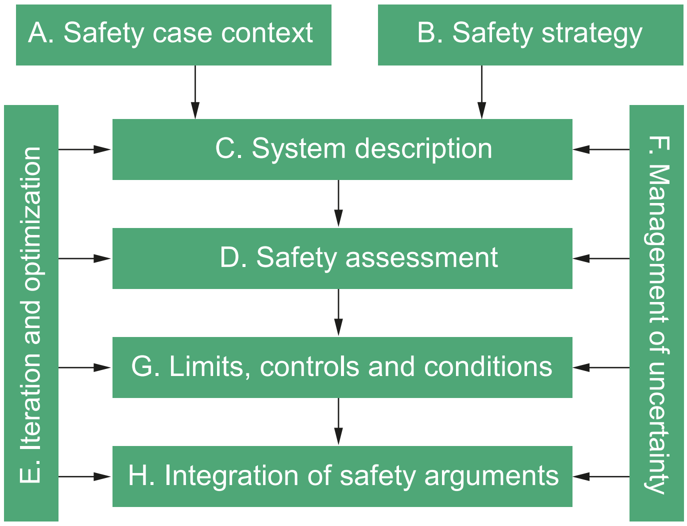


Figure 3-2: Components of a safety case (IAEA, 2012)

3.2 Structure of a safety case

Expanding upon the concise definition of a safety case given in Chapter 1, the IAEA (2012) and NEA (2013) draw attention to the following key points. The safety case has to:

- provide the basis for understanding the multibarrier system and how it will behave over time;
- address site aspects and engineering aspects, providing the logic and rationale for the design;
- be supported by a safety assessment that includes quantitative estimates of the behaviour and evolution of the multibarrier system;
- identify and acknowledge unresolved uncertainties that may exist at the specific stage of the geological disposal development programme, along with their safety significance and approaches for their management;

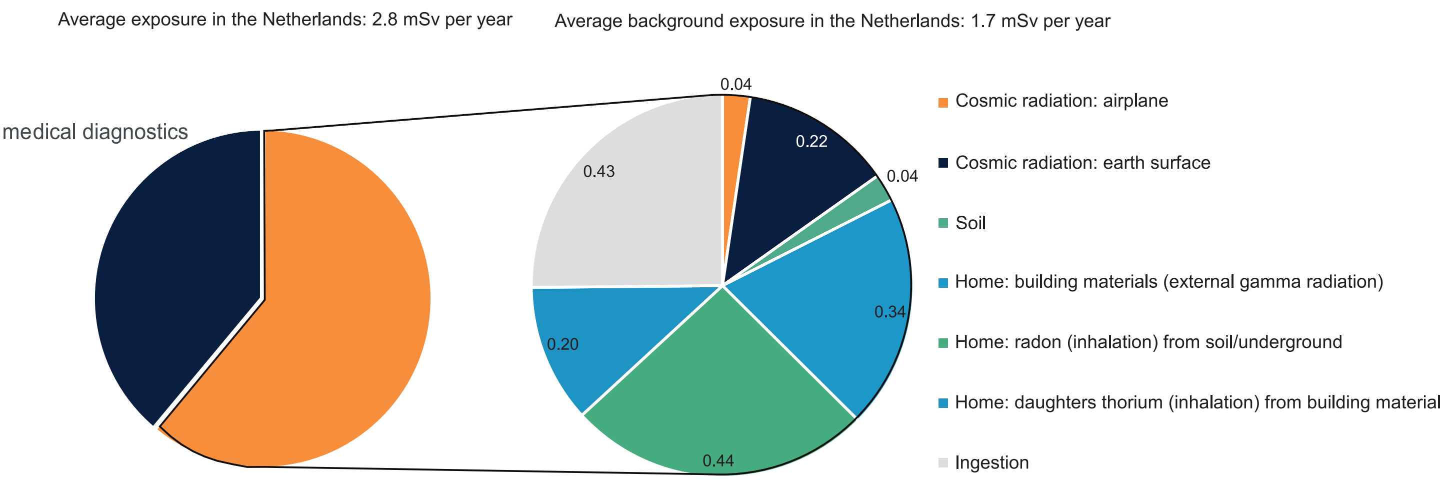


Figure 3-1: Average radiological exposure to members of the public in the Netherlands with a total exposure of 2.8 mSv per year as estimated by the National Institute for Public Health and Environment, as estimated in 2021. Radiological exposure by natural radionuclides and cosmic radiation from Smetsers and Bekhuis (2021).

- include additional information and evidence that supports the safety assessment and provides reasoning on the robustness and reliability of the multibarrier system;
- present, if required, more general arguments and information to put the results of safety assessment into perspective.

The components of the safety case as defined by the IAEA are portrayed graphically in Figure 3-2. Each of these items is addressed in the present report, as follows:

- The **safety case context** was mentioned already in Chapter 1; to increase confidence in post-closure safety, assess the disposability of the waste and assure adequate funding for disposal. The Dutch national radioactive waste management programme is currently being evaluated in the framework of the EU Waste Directive (EC, 2011) and this safety case has been written to help this evaluation.
- The following Section 3.3 gives more details on the overall **safety strategy**.
- A high-level **system description** of the multiple barrier system for geological disposal of radioactive waste is covered in Chapter 2 and an overview description of COVRA's current concept for a multibarrier system with clay as a host rock is described at the end of this Chapter. The system description is covered in more detail in Chapters 4, 5 and 6. Chapters 5 and 6 also discuss current design requirements and specifications, showing how the components of the multibarrier system contribute to safety.
- **Uncertainties** and gaps in data can arise because not all of the information required for a safety assessment is available. This can be addressed by using alternative models of processes and behaviour or by making assumptions. When assumptions need to be made, these are generally chosen to be conservative, i.e., pessimistic, so as not to overestimate the performance of the multibarrier system. However, a best estimate of the expected evolution can also be made, and this provides a perspective on how conservative the assessment assumptions are. Chapter 7 discusses the realistically expected evolution of the engineered barriers in clay host rocks.
- Chapter 8 shows the system evolution assumed for the **safety assessment** and the numerical results of safety assessment calculations that were derived in OPERA, and complements these with some waste-specific assessments based on developments in understanding of the behaviour of the multibarrier system.
- Chapter 9 is an **integration** of the previous work to formulate conclusions. Discussion of uncertainties has not been allocated a specific section; instead, the uncertainties associated with each of the important processes described, or with the data employed, are addressed at the appropriate section. In addition, the final Chapter summarises uncertainties and open questions.
- **Design iterations**, as indicated in the IAEA structure, have been performed and are presented in Chapters 4 and 6.

3.2.1 Safety strategy

According to both IAEA and NEA guidance documents, one of the initial components of the safety case should be a safety strategy (IAEA, 2012; NEA, 2013), which is defined as the high-level approach adopted for achieving safe and acceptable disposal of radioactive waste. The implementer (i.e., COVRA) should develop the safety strategy. In the current phase of work in the Netherlands, the strategy should provide for a systematic process for developing, testing and documenting the present level of understanding of the performance of a GDF and for building and maintaining the necessary knowledge and competences through successive research programmes. It is important to note that the safety strategy is presented in the form of a living document; both the strategy and the disposal concepts based on the strategy will develop iteratively over the whole implementation period, which in the Netherlands is currently planned to last almost a century (van Gemert et al., 2023).

The safety strategy also includes the identification of the overarching national and international requirements to be satisfied and the definition of the more detailed requirements made by the programme implementer to accomplish this. National and international requirements are derived from relevant national (Dutch Decree on radiation protection) and international regulatory frameworks (EURATOM). COVRA's requirements are more specific derivations of the general international and national requirements. Both are used to define further requirements to be satisfied by the multibarrier system and its components and lead to the identification of the design requirements and design specifications for the implementation of the GDF. The safety strategy has been chosen to focus on a design-driven basis by developing a hierarchical set of different levels of requirements in a requirements management system (RMS), as described below in section 3.4.

3.3 Roles of the safety case

The role that the safety case will take throughout the conceptual, site selection, constructional, operational, closure and post-closure phases is different. The iterative nature of the safety case is apparent when one considers Figure 3-3. This figure shows the common steps or stages in the decision-making processes leading to geological disposal and indicates the key stakeholders involved, as well as the planned timing in the Netherlands, as laid down in the first national programme made in the framework of the EU Waste Directive (I&E, 2016). At each decision point, the safety case has to provide the safety related information that allows a judgment on whether to proceed to the next stage.

The nature of the decisions to be made and the characteristics of the safety case for each of the stages in the disposal of waste are described in IAEA (2011b) and the regulatory expectations of the safety case are periodically updated in the European Pilot Study (EPS, 2016). The design basis for each phase has a different set of objectives, requirements, constraints, inputs and outputs (IAEA, 2020).

3.3.1 Need for action stage

When a country starts generating radioactive waste, there is a need for action by the government, which has to define a policy to meet its responsibility for managing all of the necessary steps, from collection to eventual disposal. Commonly, the government nominates or establishes an organisation responsible for developing and implementing the disposal strategy. The Netherlands already passed this stage in 1982, with COVRA being the nominated agency to manage Dutch radioactive wastes. The decision for geological disposal of waste was made in 1984.

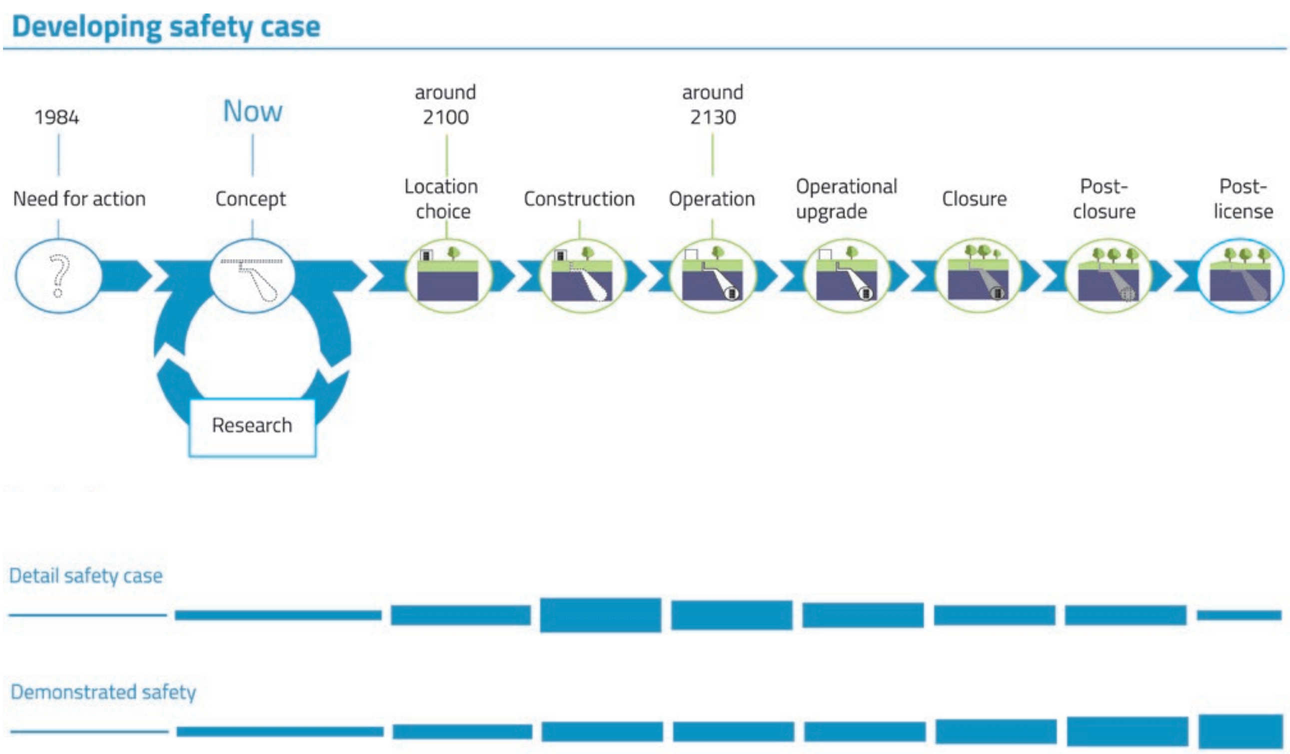


Figure 3-3: Developing the safety case with the planning as laid down in the first national programme made in the framework of the EU Waste Directive (I&E, 2016).[7]

3.3.2. Disposal concept stage

The Dutch programme is in the conceptual phase of geological disposal development, in which different disposal concepts and potential host rocks are being considered. These generic designs allow definition of associated generic, non-site-specific safety cases (how the concepts being considered would provide safe disposal for the waste inventory) and they provide a starting point for programme planning and the estimation of duration, costs and project risks (IAEA, 2020). The government lays out the requirements for and framework within which geological disposal of waste should be implemented. The implementer (COVRA) establishes the safety strategy, incorporating these requirements into the high-level tiers of its RMS, and carries out preliminary safety assessments for post-closure. Post-closure safety should be provided by a system of natural and engineered barriers. Regulatory review of the work at this stage should guide the implementer on the likelihood of achieving the necessary demonstration of safety (EPS, 2016). This is effectively the present stage of the Dutch programme. COVRA has established a research programme (for non-site specific safety cases) for the next decades (see section 1.5.7). In this, disposal concepts are developed for two host rocks (poorly indurated clays and rock salt) and other techniques emerging from international collaboration are investigated. COVRA makes safety cases - including the post-closure safety assessments - aligned with the review cycles of the national programmes in the Waste Directive (EC, 2011), which implies that a safety case is to be made each decade (Verhoef et al., 2020; Verhoef et al., 2017). COVRA also makes the cost estimates and research programmes for disposal of waste every 5 years and publishes updates of safety cases 5 years after publication of its safety cases.

3.3.3 Site selection stage

The government, together with the implementer, must develop a national framework for decision-making on site selection. For successful projects, this must be widely supported, and adhered to, by the relevant actors, whose roles and interrelationships must be clear. The national framework should support participatory, flexible and accountable decision-making processes. For example, the implementer identifies potentially suitable sites that are compatible with the disposal concept(s) and characterises these sites to the extent that a decision can be made on a preferred site (EPS, 2016). In the Netherlands, it is not yet decided who will identify potentially suitable sites but, in any case, a key element of the basis for this decision should be a safety case, including at least an outline of the operational safety case together with a comprehensive post-closure safety case. One of the most important inputs of this post-closure safety case is the Site Descriptive Model (SDM). The SDM can be seen as a synthesis of the descriptions of the site geology, rock mechanical properties, thermal properties, hydrogeological properties and parameters, hydrogeochemistry, transport and flow properties and surface environment. The SDM represents an integrated suite of information and understanding of the natural systems. The SDM is not static but is continuously updated as the

7. A period of 30 years between the decision on GDF location and the start of emplacement of waste packages in a disposal facility has been recognised to be too ambitious, based on experience from available current practices - for example in Finland. The Advisory board of the ANVS therefore advised the decision on the disposal method to be taken earlier than around 2100 in order to start emplacement of waste in a facility in 2130 (de Vries, 2019). In addition, in the dialogue sessions held in the framework of the national programme, many experts and stakeholders expressed the opinion that postponing decision making until around 2100 is irresponsible (van Rooijen et al., 2023).

Box 3-1: Institutional arrangements for radioactive waste management

The institutional arrangement of actors with responsibilities in the management of radioactive waste can be viewed as a triangle in which the authorities, waste management organisation and waste generators must each fulfil clearly defined roles that are described below and must exhibit independence from the others.

The full range of stakeholders in the Dutch programme also includes the public - both nationally and internationally. The Dutch public is currently kept informed about progress in geological disposal of waste through websites, governmental workshops, guided tours at COVRA's premises and lectures upon invitation by members of Dutch society.

Dutch authorities

The authority that prepares policy and establishes laws governing the generation and management of radioactive waste is currently the Ministry of Infrastructure and Water Management. More specifically, the Directorate-general for Environment and International Affairs within this Ministry is responsible for policy development with regard to nuclear safety, security and radiation protection (I&W, 2020). The Ministry prepares the national programme to comply with the European Commission Waste Directive (EC, 2011) and prepared the last (seventh) national report for the IAEA Joint Convention (I&W, 2020). The Ministry is also responsible for developing a disposal policy aimed at arriving at a publicly accepted disposal facility.

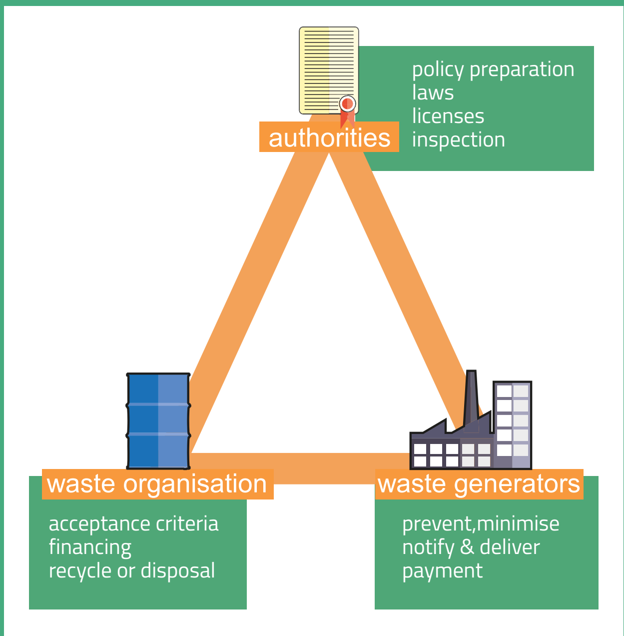
The authority that grants licences and carries out inspections is the Authority for Nuclear Safety and Radiation Protection (ANVS), which was established in 2015. Its responsibility for policy development was transferred to the Ministry of Infrastructure and Water Management in May 2020 (I&W, 2020). ANVS focusses on the safety aspects related to the geological disposal facility. Reviewing COVRA's safety cases is therefore an important part of ANVS work. ANVS has a legal responsibility for informing the public about nuclear safety and radiation protection. The Ministry of Infrastructure and Water Management allocates the financial resources for ANVS to carry out its duties.

Waste generators

Waste generators are nuclear power plants, nuclear research reactors, hospitals and research organisations. These generators are required to minimise the generation of radioactive waste as much as is reasonably achievable. Radioactive materials for which no use, re-use or recycling is foreseen, are transferred to COVRA. The waste generators pay COVRA waste fees and notify COVRA of the types and amounts of wastes being produced. Each generator prepares the waste according to the waste acceptance criteria set by COVRA and submits documentation on the characteristics of the waste. This documentation has become more detailed, in order to ensure sufficient characterisation of the waste to be disposed of and also to treat the wastes more safely. Discussions with some waste generators are on-going, to allow COVRA to confirm that all relevant details have been provided.

Waste management organisation

The government founded COVRA in 1982 to manage radio-active waste in the Netherlands, from collection to an end-point management technique. This technique can be recycling of sufficiently decayed waste or geological disposal for other types of waste; no near surface disposal is planned in the Netherlands. COVRA takes ownership of the radioactive wastes when they are delivered and is responsible for development and implementation of the disposal facility. COVRA is charged with implementing all necessary steps to ensure safety at all times into the future. These include waste collection, treatment and storage, conducting research on geological disposal and implementing final disposal. COVRA is also responsible for ensuring that the fees paid by waste generators ensure that sufficient funding is available for all future radioactive waste management steps. Every 5 years, COVRA updates its cost estimate for the GDF, taking into account national developments and also the inter-national-state-of-the art, in order to ensure that the GDF costs will be covered by COVRA's waste fees.



site knowledge base grows through further investigations (IAEA, to be published). The Environmental Impact Assessment (EIA: in Dutch, Milieu Effect Rapportage) is based on this safety case. The realisation of the SDM will require the drilling of boreholes to characterise the sub-surface environment. The implementer is expected to require a licence for drilling boreholes or for implementing site specific underground facilities. A licence application for drilling may already require an EIA. This EIA includes the impact on local and regional stakeholders of any environmental disturbances, including potential harmful emissions, noise, dust, traffic etc, during the drilling.

A monitoring programme and the organisations that, in addition to the implementer, monitor properties of the site, must be identified and agreed. The aim of the monitoring carried out by the implementer is to obtain reference values for a wide range of environmental parameters. Monitoring is therefore to be started at an early stage, with one of its aims being to quantify, against a pre-construction baseline, any additional (radiological) exposure from the construction and operation of the geological disposal facility. The National Institute for Public Health and Environment monitors radioactivity in the Dutch environment and has sensors within nuclear facilities in order to perform their own independent monitoring. All countries in the European Union are required to perform these measurements in their national environments, under the terms of the Euratom treaty of 1957. It is therefore expected that this National Institute will also have a role in the independent monitoring of the GDF.

Local and regional stakeholders have an important role during the lifecycle of the GDF, especially during the site selection process and onwards. Public information, consultation and/or participation in environmental or technological decision-making should represent current best practice and must take place at different geographical and political scales. Large-scale technology projects are more likely to be accepted when local and regional stakeholders have been involved in making them possible and have developed a sense of interest in, or responsibility for, their success. For the Netherlands, this stage of site selection lies far in the future, probably not beginning until the second half of the 21st Century. However, the approaches to be used and the decision processes that will be applied must be proposed, discussed by all stakeholders and agreed at an earlier phase in the disposal programme.

3.3.4 Construction licensing stage

The reference design (and application for construction) phase, is the period in which the implementer adapts the conceptual design to the site properties, substantiates and finalises the design of the disposal facility and develops the safety case to support the implementer's application to construct, operate and close the facility. Based on the review of the safety case, the licensing body decides whether to grant a licence for the implementer to construct the facility. This is a crucial milestone in the development of a GDF (EPS, 2016). Depending on the licensing approach adopted, licensing may be the basis for going underground to enable more detailed and direct characterisation of the site than can be accomplished from the initial boreholes, or it may be the basis for extending from a URF that has already been used for underground characterisation purposes into volumes of rock that will be used for disposal. The reference design uses information from the SDM and the EIA includes assessment of the impacts on the local and regional environment, as well as identifying the impacts on stakeholders of any environmental disturbances resulting from GDF construction activities (IAEA, to be published). The EIA will also address mitigation measures to reduce such disturbances, developed in agreement with the hosting community.

3.3.5 Construction and operational licensing stage

When a construction license is granted, underground access, characterisation and testing excavations can be extended into a progressive programme of GDF construction, including any surface facilities such as a waste encapsulation plant that may be required.

During construction, the implementer demonstrates that the facility is being built as planned in the safety case and in accordance with the conditions of the construction licence. Towards the end of this phase, the implementer will present its final approach for operation and a concept for closing the facility and then submit an application for an operating license. In preparing for operation, the implementer will need to demonstrate safety during operation, including radiation protection of workers and members of the public (EPS, 2016). Commissioning tests are envisaged to be required to provide final assurance that the GDF will operate safely. These might include tests of the transportation and emplacement of waste packages using dummy waste containers of the same weight and shape as the final waste packages (IAEA, to be published).

3.3.6 Operation and closure phases

The operational stage is the period in which the implementer emplaces waste packages in the disposal facility. During this phase, the implementer may excavate new disposal tunnels or caverns, and possibly backfill and seal underground openings, either temporarily or permanently. Late in this phase, the implementer also develops an application to close and seal the facility, and prepares a plan for post-closure institutional controls, monitoring and surveillance. At the termination of operations, the regulator will decide whether to grant a licence for the implementer to close and seal the facility. When the licence is granted, the implementer proceeds to the closure of the facility (EPS, 2016).

3.3.7 Post-closure stage

The post-closure phase, is the period in which the implementer provides evidence to demonstrate that it has closed the disposal facility in accordance with safety and license requirements, presents a firm plan for institutional controls, and continues monitoring and surveillance as long as is required by the national legal and regulatory framework (EPS, 2016).

3.3.8 Post-licensing period

At some point after closure, the GDF will cease to be a licensed nuclear facility in the ownership of the implementer. The national government takes over responsibility for the GDF. International nuclear safeguards requirements (with respect to any fissile materials contained in the GDF) might then be satisfied by remote surveillance means (e.g., satellite monitoring, aerial photography, micro-seismic monitoring). All relevant information about the nature and location of the GDF is expected to be accessible, as obligated by implementation of the European Directive for the establishment of Infrastructure for Spatial Information in the European Community (EC, 2007). It is likely that the national government will introduce measures to regulate any monitoring, surveillance or safeguarding activities and to control or prohibit activities, such as exploration drilling, in the vicinity of the GDF.

3.4 Requirements Management System (RMS)

3.4.1 What is an RMS?

There is a broad agreement in systems engineering that the identification and management of requirements for complex systems and their components is essential, if the purposes and objectives of the final system are to be achieved. In the field of systems engineering, a requirements management system (RMS) is a hierarchical set of requirements establishing a design basis for a process or a manufactured object, such as a piece of machinery, a building or a major piece of infrastructure. An RMS should provide the logic and the rationale of the design and provide a structured framework for checking that all requirements are being met.
For a GDF, this framework facilitates management of requirements that are placed on the multibarrier system and its component parts and ensures that the inevitable changes in requirements and specifications that will occur over the lifetime of a disposal project are properly addressed and documented. In addition, it will help to identify knowledge gaps and potential optimisations. A key goal of an RMS is to ensure that the completed component or structure fully meets all the requirements that drove the design.

3.4.2 Why do we need an RMS for disposal

Implementing geological disposal of waste is a lengthy process that has to cover waste collection, treatment, processing, storage and disposal (see Figure 3-4). Throughout this long and complex series of activities, an RMS provides a tool to identify and manage requirements, to provide traceability and transparency, and to communicate between professionals with different expertise.

Each step in the management of waste involves facilities and activities that must be linked. Each procedure and facility is a sub-system within the overall waste management system that COVRA has to manage, and each sub-system places requirements on one or more of the others. Managing these systems involves a broad range of disciplines including civil, electrical and chemical engineering, worker health and safety, security, geology, physics, chemistry, microbiology, and project and cost management. Requirements management takes advantage of the existing information in all these areas and the corresponding work-processes available and integrates them into an overall structure to ensure successful implementation of the management of the waste. International experience has shown that the necessary integration of requirements for disposal of waste is best addressed by the early development of a requirements driven design basis (IAEA, to be published).

3.4.3 Current RMS at COVRA

COVRA initiated its RMS in OPERA (Verhoef et al., 2017). This RMS has been expanded from an RMS specific for disposal of waste into an RMS that includes all the steps in the management of waste. This updated RMS, consisting of six levels (Figure 3-5), is currently in a development stage (COVRA, 2017, 2022). The first two levels of the current RMS are requirements that hold for disposal as well as for pre-disposal activities (collection, treatment and storage). These two levels contain requirements that are applicable to all the steps in the management of waste: for example, isolation of the waste from people and the accessible biosphere. Isolation is foreseen as an active measure during the pre-disposal phase: i.e., isolation has to be provided by security of the buildings where the waste is stored. Considering the long-time scales involved for disposal of waste, as explained in Chapter 2, this isolation must be provided by passive means in the post-closure period: i.e., isolation has to be provided by natural barriers. From the third level down in the hierarchy, requirements become specific for pre-disposal and disposal activities. For the purposes of this study, where we look specifically at disposal, the GDF can be treated as a system in its own right.

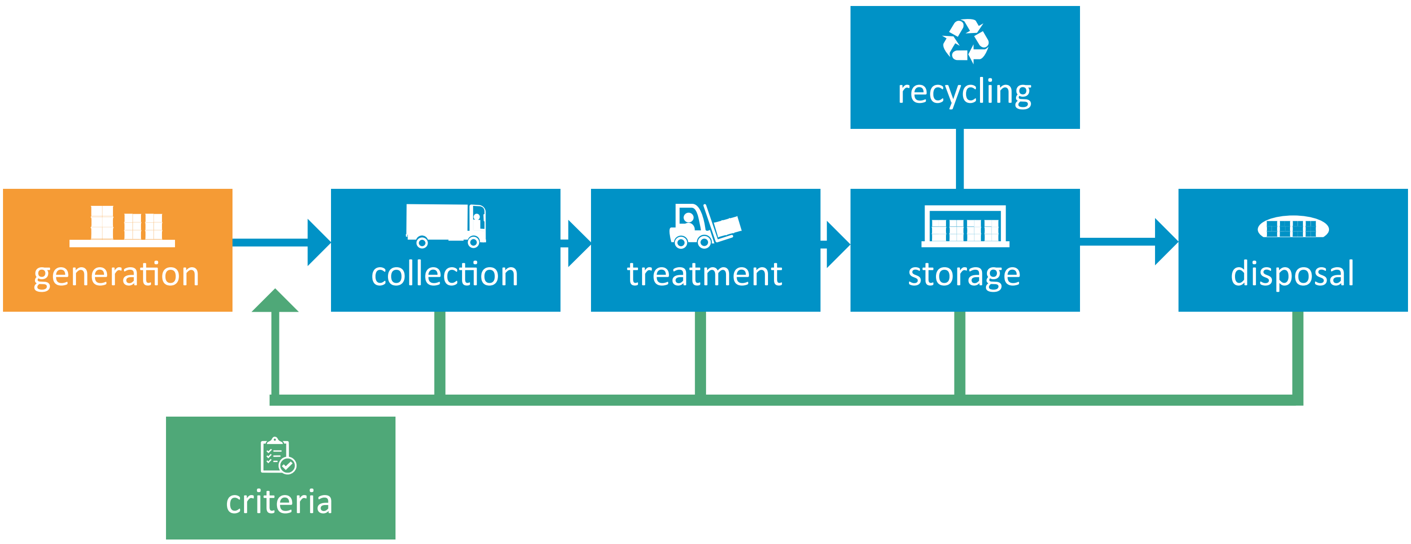


Figure 3-4: All the steps of radioactive waste management involve requirements on linked systems that must be managed (adapted from COVRA (2017)).

Lower levels of the hierarchy, contain the requirements on components or sub-systems; these can be expressed as safety functions, with associated design requirements and performance targets. Each subsystem or component can have one or multiple safety functions. Design requirements are derived from the safety functions and the lowest levels of the hierarchy are the detailed design specifications for components which are required to ensure that safety functions are fulfilled.

Unlike COVRA's pre-disposal systems that are already operational, our disposal systems with the host rocks clay and salt are currently generic and conceptual so our GDF designs are at an early stage. Nevertheless, we can define the safety concepts and their subsystems in sufficient detail to develop an RMS for these disposal systems that already acts as a useful guide to our work. The safety concept for passive safety with its safety functions that meets the high-level requirements (isolation and containment) is described at the end of this chapter for a multibarrier system with clay host rock. Design requirements and design specifications are described in Chapter 4, 5 and 6. COVRA's current RMS is at an early and incomplete stage and will certainly be extended, adapted and updated in the future. For the present study, an approach has been adopted where, for each sub-system or component of the GDF and multibarrier system, we consider shielding, isolation, containment, handling and monitoring when we define the safety and operational functions, design requirements and design specifications.

3.4.4 Structure of the requirements

Requirements must be clearly and unambiguously defined, without duplication. In COVRA's current RMS, each requirement consists of three parts namely: (1) a UIN (Unique Identification Number), (2) the requirement itself and (3) a short description of the requirement. As the RMS develops in future, other attributes are expected to be assigned to each requirement, such as measures of effectiveness in meeting the requirement, current status, and the responsible 'owner' within COVRA.

The UIN is a unique identifier needed to keep track of the requirement and its inter-relationships in the RMS. It is also used to identify its location in the RMS (e.g. the level and the (sub) system it belongs to) and the source of the requirement. An abbreviation for the source of a requirement is included in the identifier for the Levels 1, 2 and 3. Figure 3-6 shows the defined requirements for those three levels and their connections, with the UIN indicating the level in the RMS, the origin and the requirement number.
The short descriptions of these requirements are in Appendix 3. The UIN in Level 3 also includes an indicator of which part of COVRA's overall RMS the requirement lies in: 'D' for disposal, in the case of GDF-specific requirements. The requirements in Figure 3-6 have been selected from the Dutch Decree on radiation protection (DCRE), the National Programme Radioactive waste (NPRA), documents written by COVRA (COV) and the Specific Safety Requirements for disposal of waste (SSR-5) by the International Atomic Energy Agency (IAEA).

3.4.5 L1 requirements for all steps in the management of waste

Level 1 requirements are applicable to all the steps in the management of waste. International requirements to which COVRA adheres can be identified in documents written by international organisations, for example EURATOM regulations and EU directives. National requirements are selected from documents written by the Dutch government and the Dutch regulatory body. The Dutch Decree on radiation protection is a General Administrative Measure giving content to the Nuclear Energy Act. The provisions of the European radiation protection criteria and standards established in Council Directive (EC, 2014) have also been implemented in Chapter 10 of this Decree, which is devoted to the management of radioactive waste. This requires the national programme (I&E, 2016) established in the framework of the European Waste Directive (EC, 2011) to contain the points of departure for the safe management of radioactive waste. Many Level 1 requirements (see Figure 3-6) have therefore been identified from the Decree and from the national programme (I&E, 2016).

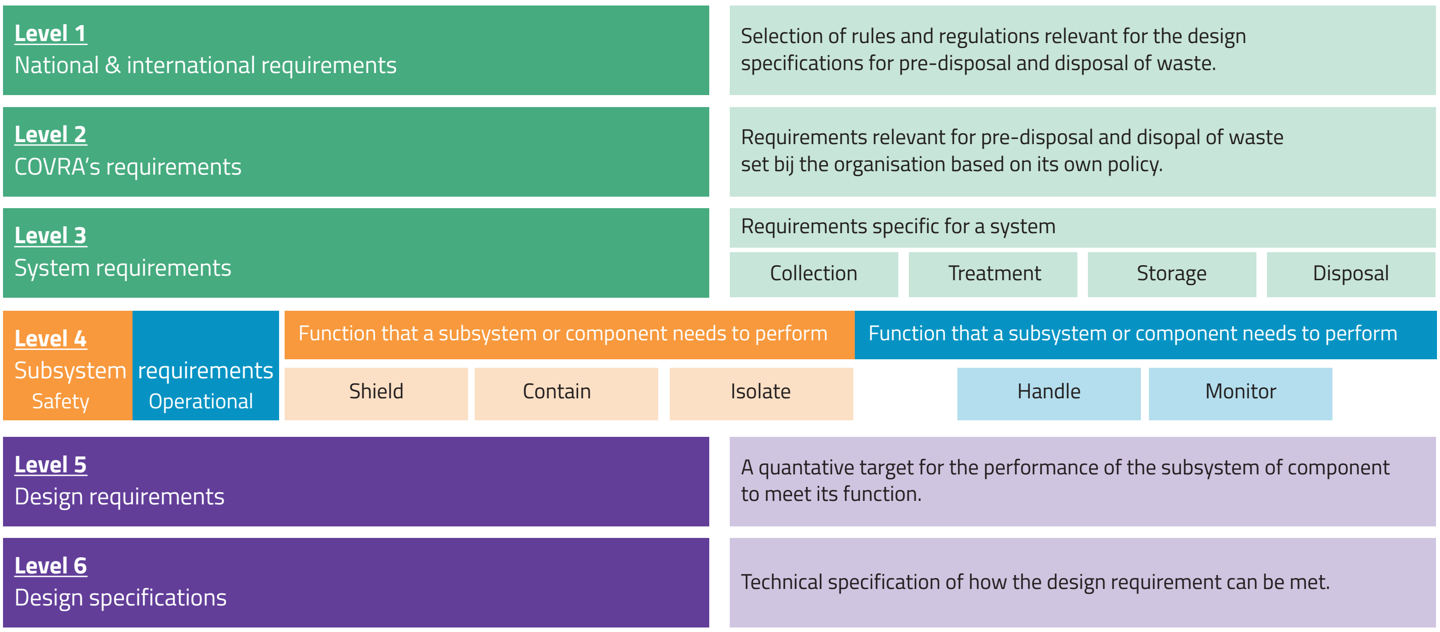


Figure 3-5: Current hierarchical arrangement of requirements: adapted from COVRA (2022).

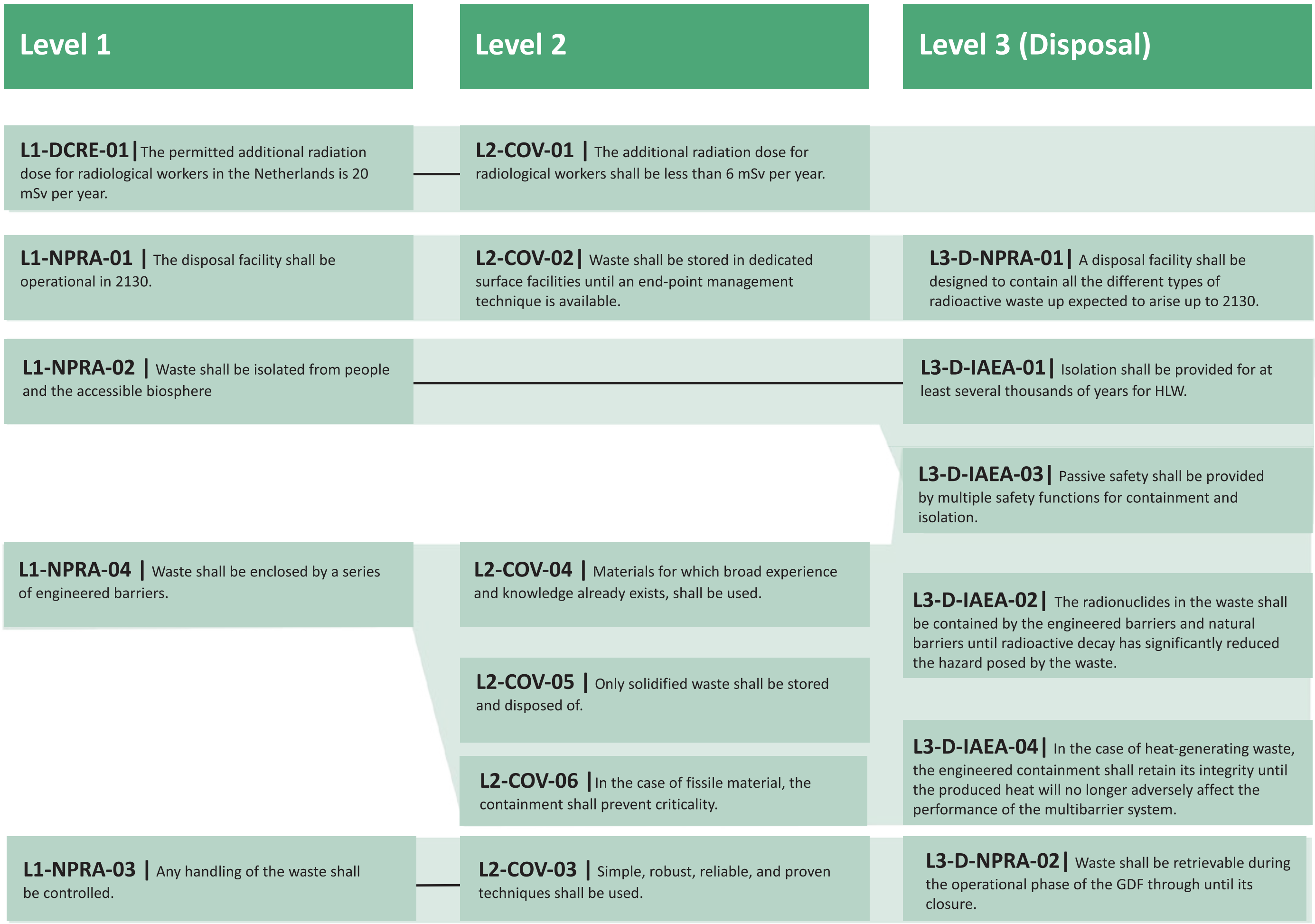
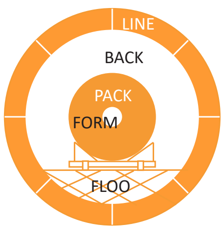


Figure 3-6: Connection between the requirements described in this Chapter. The black line between some requirements means that the lower-level requirement is a refinement of a higher level requirement.

L4-CD-CLAY-CONTA-01: The low permeability of the clay host rock shall minimise the access of water to the EBS by which the alteration rates of components in the EBS are minimized and the pore water is stagnant allowing radionuclides, if released from the waste form, only to leave the clay host rock through diffusion.

L4-CD-CLAY-CONTA-02: The reducing conditions in the clay host rock shall limit the solubility of redox sensitive elements by which corrosion rates of redox sensitive metals in the EBS are minimized and the dissolved amount of redox-sensitive radionuclides that can leave the clay host rocks, is minimised.

L4-CD-CLAY-CONTA-03: Solid and immobile phases in the clay host rock shall limit the dissolved amounts of radionuclides that can leave the clay host rock, especially radionuclides that are dissolved as cations or cationic dissolved complexes.



L4-CD-LINE-CONTA-01: The thickness and strength of the liner shall ensure sufficient mechanical support to minimize mechanical disturbance of the clay host rock until the waste has sufficiently decayed till non-hazardous levels.

L4-CD-LINE-CONTA-02: The thermal properties of the liner shall ensure sufficient heat dissipation until the clay host rock is no longer heated by the waste.

L4-CD-BACK-CONTA-01: The thermal properties of the backfill shall ensure sufficient heat dissipation until the clay host rock is no longer heated by the waste.

L4-CD-BACK-CONTA-02: The strength of the backfill shall ensure sufficient mechanical support to minimize mechanical disturbance of the clay until the waste has sufficiently decayed till non-hazardous levels.

L4-CD-BACK-CONTA-03: The low permeability of the backfill shall minimize the access to water to the waste package in order to limit its alteration and minimize further spreading of radionuclides, if released from the waste form, into the clay host rock.

L4-CD-PACK-CONTA-01: The physical and chemical properties of materials used for the package shall prevent contact between the waste form and pore water until the clay host rock is no longer heated.

L4-CD-PACK-CONTA-02: The low permeability of the package shall limit the access of water to the waste form by which the radionuclide release rate is minimized and minimize further spreading of radionuclides, if released from the waste form, into the backfill.

L4-CD-FORM-CONTA-01: The low solubility of the waste form minimizes the radionuclide release rate into the package.

Figure 3-7: Post-closure safety functions of the different components of a multibarrier Disposal system with Clay (CD) host rock and heat generating HLW . Clay host rock (CLAY), Overburden (OVER), Liner (LINE), BACK(backfill), FLOO(floor), PACK (package) and waste form (FORM).

3.4.6 L2 COVRA's requirements for all the steps in the management of waste

Level 2 requirements are also extracted from the national and international requirements, but have been specifically developed into a form that reflects COVRA's policy. These level 2 requirements are also applicable to all the steps in the management of waste; they can be further restrictions of level 1 requirements or additional requirements (see Figure 3-6).

3.4.7 L3 system requirements specific for disposal of waste

Level 2 requirements are also extracted from the national and international requirements, but have been specifically developed into a form that reflects COVRA's policy. These level 2 requirements are also applicable to all the steps in the management of waste; they can be further restrictions of level 1 requirements or additional requirements (see Figure 3-6).

3.4.8 L4 requirements specific for disposal of waste in clay host rock

The safety concept with safety functions assigned to components of the multibarrier system gives an integrated picture of how the engineered and natural barriers provide safety after closure of the GDF. Each safety function, providing passive safety in the post-closure phase, has an assigned time frame, as described in paragraph 3.4.8.1. In this chapter, this time frame is indicated qualitatively. The quantification of the time frame is described in chapters 4, 5 and 6. A safety concept with safety functions also needs to be made to provide operational safety, as described in section 3.4.8.2.

3.4.8.1 Passive safety

The engineering and containment properties of the engineered barriers and clay host rock are sensitive to temperature. The host rock provides ideal containment at its natural ambient temperature. The safety concept for passive safety of heat-emitting waste (e.g. vHLW) therefore includes a period of engineered containment that prevents release of radionuclides from the waste form until the clay host rock is no longer heated by the waste.

Five main components of the multibarrier system were identified in chapter 2: waste form, waste package, underground engineering structure, host rock and surrounding rock formations. Using a similar approach to that used in the Finnish project for geological disposal of spent fuel (Posiva, 2021), an abbreviation for the component is included in the unique identifier for the safety functions.

The clay host rock(s) and the engineered barriers should provide containment until the hazard potential of the waste has decayed sufficiently. Three structural components are used in the multi-barrier system with a clay formation: tunnel liner, backfill and floor (which becomes part of the backfill in the post-closure phase).

The overburden (the rock formations above the host rock) provides isolation. Figure 3-7 shows the safety functions of the different components with their associated time frame. As described in Chapter 2, the components and their safety functions assure that the GDF system provides passive safety in the post-closure period, as the hazard potential of the waste progressively declines.

3.4.8.2 Operational safety

Operational safety in the GDF includes the physical protection of workers in the disposal facility from accident hazards and the radiological hazards involved in handling waste packages. A stable underground working environment is required. The clay host rocks investigated in the Netherlands are poorly indurated and require substantial tunnel support in the form of the tunnel liner to provide a safe and practical working environment.

HLW is remotely-handled in surface facilities at COVRA's premises. In the underground facility, there is less space and hence more difficulties in monitoring the movement of waste packages and inspecting the quality of shielding material. It is therefore envisaged to emplace only contact-handled waste packages in the disposal facility in order to contribute to a safe and practical working environment. In the current disposal concept, LILW as well as HLW are disposed of in a single disposal facility. The constructional activities and the emplacement of waste packages are separated by performing these at different times: i.e., sections of the underground facility are completely constructed before waste emplacement commences.

Figure 3-8 shows the safety and operational functions of some of the components that are present in the operational phase. Other features that contribute to the safety of the working environment are, for simplicity, not included.

3.4.9 L5 requirements specific for disposal of waste in clay host rock

A design requirement is a quantitative target for the performance of the subsystem or component to meet the safety or operational function (see Figure 3-5). Chapter 4 shows the derivation of the design requirements that any handling of waste shall be controlled (CONTR). The derivation of the design requirements to contribute to containment (CONTA), that sufficient radiation protection (RADPR) is provided e.g. by shielding and that any handling of waste shall be controlled (CONTR) are presented in Chapter 6. The backfill (BACK) and floor (FLOO) have the same design requirement to contribute to containment (CONTA) in the post-closure phase.

3.4.10 L6 requirements specific for disposal of waste in clay host rock

The design specification is a technical specification of how the design requirement can be met (see Figure 3-5). Chapter 4 and Chapter 6 show the derivation of the design specifications with a multicriteria analysis. Each technical specification is a solution for a single or several design requirements. This solution consists of the requested material property and in many cases also the dimensions of the material. The technical specifications with dimensions in Figure 3-10 are all for the thicknesses of cylindrical shaped geometries: liner and in the package for heat generating HLW: overpack and buffer.

L4-CD-LINE-CONTR-01: The lining shall ensure sufficient mechanical support in the operational phase of the facility.

L4-CD-LINE-CONTR-01: The lining shall limit water inflow into the tunnel.

L4-CD-DUCT-CONTR-01: The air entering the disposal tunnel through the duct shall ensure sufficient oxygen for the workers and cooling, if needed.

L4-CD-DEVI-CONTR-01: The waste package shall be emplaced under human control. If needed, a device shall be emplaced as well in order to be able to retrieve the waste in the operational phase of the facility.

L4-CD-PACK-RADPR-01: Contact handled waste packages are foreseen to be emplaced in the GDF.

L4-CD-FLOO-CONTR-01: The floor shall ensure sufficient mechanical support when loaded. If needed for the equipment used to emplace a package, the floor shall have a finishing off.

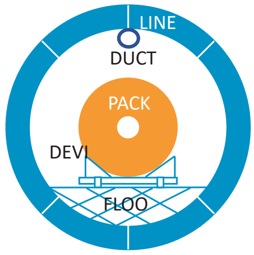
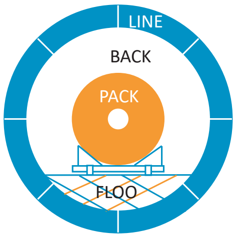


Figure 3-8: Pre-closure safety and operational functions of the different components for a disposal system with clay (CD) host rock with heat generating HLW.



L5-CD-LINE-CONTR-01: Mechanical support by the liner as a function of the radius of the tunnel and thickness of the liner should be larger than the pressure as a function of the radius of the tunnel and the properties of clay at the depth of the facility.

L5-CD-LINE-CONTR-02: The liner shall be 'water-proof' i.e. impermeable in engineering terms.

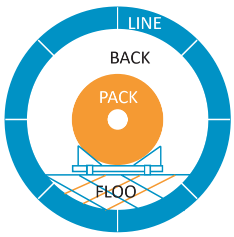
L5-CD-PACK-CONTA-01: The package shall sustain a mechanical load of 10 MPa (500 metres) for 1200 years.

L5-CD-PACK-RADPR-01: Gamma and neutron contact dose rate for each package shall be less than 0.12 mSv per hour (max. 15 minutes handling per day) and at 1 metre less than 0.0075 mSv per hour (max. 4 hours handling per day).

L5-CD-FLOO-CONTA-03: The material used to construct the floor shall have a smaller diffusion value for water than the clay host rock.

L5-CD-FLOO-CONTR-01: The floor shall ensure sufficient mechanical support for the axle load of a loaded transport device.

Figure 3-9: Design requirements to meet some of the safety and operational functions in Figure 3-7 and Figure 3-8.

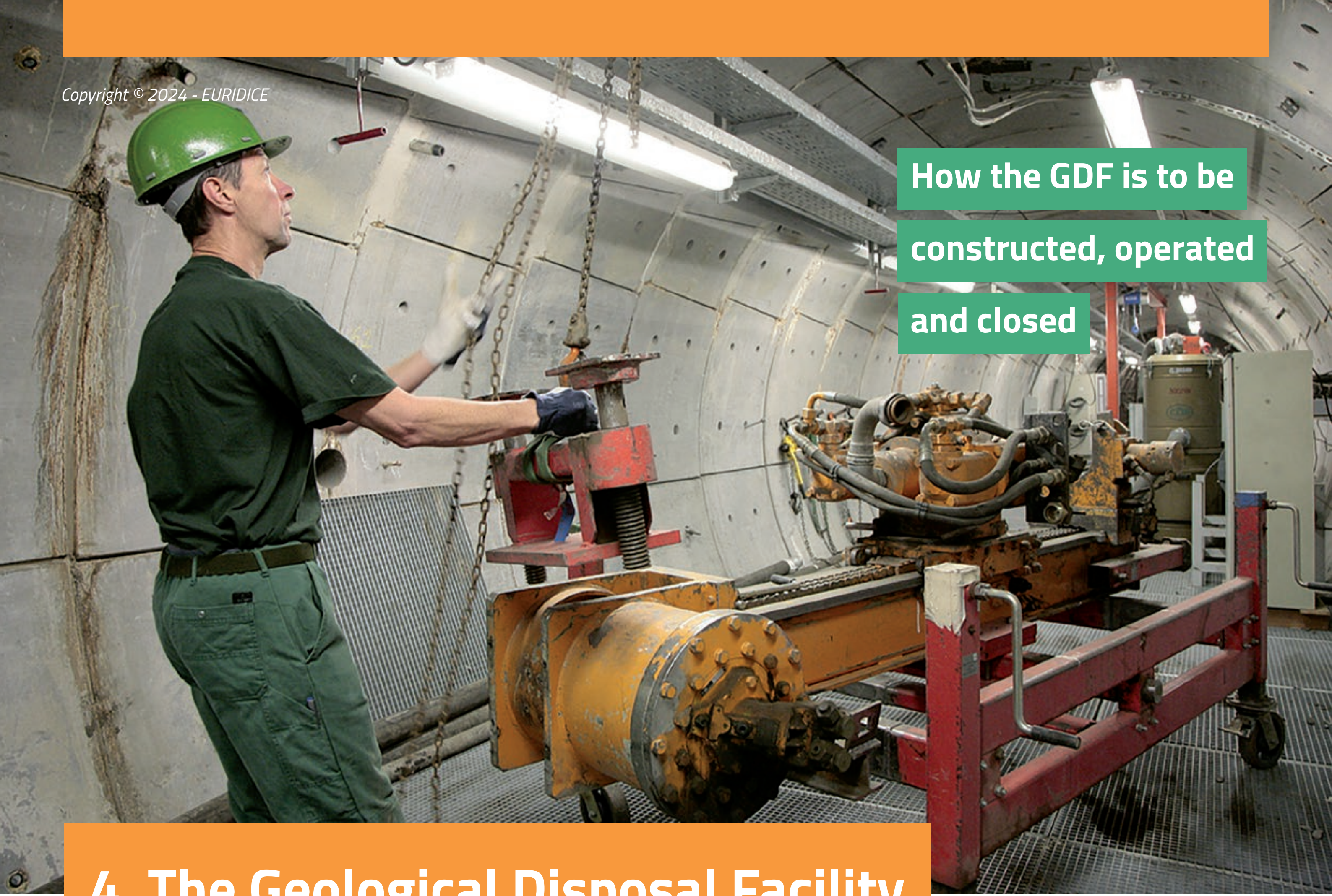


L6-CD-LINE-CONTR-01&2: A thickness of 0.5 m for concrete with a compressive strength of at least 80 MPa after 28 days hardening at a depth of 500 metres and an internal diameter of 4.0 meter.

L6-CD-PACK-RADPR-01&CONTA-01: For HLW cooled for 130 years
1) the carbon steel overpack shall have a thickness of 30 mm (density 7850 kg m^{-3} & yield strength 600 MPa & uniform corrosion rate of 0.1 µm per year)
2) the outer diameter of the concrete buffer shall be 2 m (thermal conductivity > 0.1 W/mK density 2350 kg m^{-3})

L6-CD-FLOO-CONTR-01&CONTA-03: A material with a compressive strength of at least 4 MPa (e.g. foamed concrete) and saturated diffusion value for water of 5×10^{-14} m^2/s.

Figure 3-10: Design specifications to meet some of the safety and operational functions in Figure 3-7 and Figure 3-8.



4. The Geological Disposal Facility

This chapter introduces the different types of Dutch waste that are destined for geological disposal and describes two potential underground layouts for a conceptual design of a GDF in poorly indurated clay host rock. The detailed waste characteristics are described in Chapter 6. Further considerations of constructional and operational feasibility have resulted in some aspects of the conceptual design of the GDF being updated from those originally proposed in OPERA.

The functions of the subsystems and components in the GDF were described in the Requirements Management System at Level 4 in Chapter 3. The definition of conceptual design layouts allows the derivation of preliminary design requirements and specifications of these subsystems and components for inclusion in the RMS. This Chapter also shows how the conceptual design complies with some of the requirements at Level 1, 2 and 3 of COVRA's current RMS. In the present study, increased emphasis has been put on the rationale for the layout and on the description of the different stages of the disposal facility implementation: construction, operation and closure. Two layouts are considered: one in which High Level Waste (HLW) and Low and Intermediate Level Waste (LILW) are disposed of at the same depth and another one in which HLW is disposed of at a greater depth than LILW. The latter is motivated by the possibility of using several Paleogene clay formations that can be present at different depths at a site so that wastes with significantly different hazard potentials, requiring different degrees of isolation, can be emplaced at appropriate depths. At this early stage, and with limited information on the potential clay host rocks, the GDF design is inevitably only developed at a conceptual level, and research on construction, operation and closure methodologies will certainly be necessary when site-specific safety cases begin to be made (i.e., after 2050).

4.1 Waste packages for disposal

The inventory of waste that will eventually be emplaced in a GDF depends mainly on the future utilisation of nuclear energy and production of medical isotopes in the Netherlands. Three waste generation scenarios have been developed in the framework of the national programme (Burggraaff et al., 2022):

- **Waste Scenario 1:** Operation of Borssele Nuclear Power Plant (NPP) until 2033 and replacement of the High Flux Reactor (HFR) by Pallas for the production of medical isotopes;
- **Waste Scenario 2:** Borssele NPP operation until 2043,i.e. the operational time is extended by 10 years;
- **Waste Scenario 3:** Waste scenario 2, with additional wastes from two new nuclear power plants with each a capacity three times higher than the Borssele NPP.

Table 4-1 shows the inventory for Waste Scenario 1, which is the scenario used in OPERA. The wastes from Waste Scenarios 2 and 3 are described in Appendix 4. Production of medical isotopes and performing research generate Spent Research Reactor Fuel (SRRF). Under the current recycling contracts with France, spent nuclear fuel from the Borssele NPP is reprocessed to produce vitrified HLW (vHLW) in CSD-v canisters and compacted hulls and ends in CSD-c canisters. The assumed number of vHLW canisters has not changed since OPERA. The amount of heat generating HLW has been estimated in more detail than other types of waste since special structures (double sided wells) need to be constructed for safe storage in the future. Non-heat generating waste can be held in a large flexible storage space, so uncertainties in estimating the number of canisters have much smaller planning implications. The estimated number of CSD-c canisters has been reduced from 600 (Verhoef et al., 2017) to 502, in agreement with the waste inventory determined by Burggraaff et al. (2022).

A category of 'legacy waste' was included in the OPERA Safety case (Verhoef et al., 2017) but improved characterisation of the waste has allowed the majority of the legacy waste volume to be re-classified as LILW, rather than HLW.

A separate section on dismantling and decommissioning waste is now included in the waste inventory for the national programme (Burggraaff et al., 2022); this was not included in the OPERA Safety case. Its volume has been estimated to be 3814 m^3.

Depleted uranium is the main type of Technically Enhanced Naturally Occurring Radioactive Material (TE-NORM) to be disposed of. Depleted uranium is also the largest volume of waste in the waste inventory under Waste Scenario 1. After OPERA, the waste generator notified COVRA of new waste arisings, and a new storage facility for depleted uranium to store all the waste was opened on 13 September 2017. The inventory in the national programme is determined by linear extrapolation of the receipt rate of containers over the past 20 years from the currently stored uranium inventory up to 2050 (Burggraaff et al., 2022). This has resulted into a volume of 49.360 m^3 compared to the 34.000 m^3 estimated in OPERA.

LILW arises from activities with radioactive materials in industry, research institutes and hospitals. It includes lightly contaminated materials, such as plastic, metal or glass objects, tissues and cloth. LILW is usually processed and conditioned using cementitious materials. The inventory volume is estimated in a similar way to that for depleted uranium, based on linear extrapolation of the rate of wastes entering storage over the past years. This extrapolation covers the life expectancy of the nuclear reactors (HFR and NPP Borssele) in the national programme (Burggraaff et al., 2022) leading to an estimated volume of 31.641 m^3. The extrapolation in OPERA went beyond the life expectancies of the reactors, leading to a volume of 45.000 m^3.

4.1.1 Waste packages for LILW

Waste packages for processed LILW are 200 litre steel drums, 1000 litre reinforced concrete containers and Konrad containers with a height of 1.7 m, width of 1.6 m and length of 1.7 m. Konrad containers are also currently envisaged for use as the waste packages for depleted uranium. The 200 litre drums consist of painted galvanized steel drums with an inside layer of cement, containing compacted waste. The 1000 litre concrete containers contain a cemented waste form. The Konrad type II containers will also contain waste conditioned with cementitious materials (see Chapter 6).

4.1.2 Waste packages for HLW

HLW is remotely handled during storage. For disposal, contact-handled waste is foreseen for all types of waste, in order to facilitate radiation protection during the emplacement and the potential retrieval of waste. The relevant design requirements and specifications are defined in RMS Levels 1, 2, 3, and 4.

Spent nuclear fuel from the Borssele NPP is reprocessed. The resulting waste products are vHLW and compacted hulls and ends, both of which are returned from reprocessing in sealed

Waste Category	In storage volume as defined in Burggraaff et al. (2022)[8]		Packaged for disposal		
	Volume [m^3]	Number of canisters / containers	Number of packages	Volume [m^3]	Weight per package [tonne]
Spent research reactor fuel	49[9]	244	244	1840	20
Vitrified HLW (vHLW)	86[10]	478	478	3754	22
Compacted hulls & ends (Non heat generating HLW)	90	502	72	452	20
Dismantling waste (LILW)	3814	-	826	3814	Max 20
TE-NORM (LILW)	49360	-	12600	58070	Max 20
Processed LILW	31461	108400	108400	For three 200 litre drums: max 2.25 tonne For one 1000 litre concrete container: max per 3 tonne	

Table 4-1: Expected inventory of wastes for disposal in 2130 for Waste Scenario 1, showing their mass and volume in storage and their mass and volume when packaged for disposal. The dimensions of the HLW packages are shown in Figure 4-1.

8. Rounding off volumes separately may result in a volume of 227 m3 for HLW, as found by Burggraaff et al. 2022.holds the URF network as a name.
9. The volume at storage is assumed be 0.20 m3 in Burggraaff et al., 2022. The outer dimensions of the container leading to 0.693 m3 were used in the OPERA Safety Case and also in this safety case.
10. The volume at storage is assumed be 0.18 m3 in Burggraaff et al., 2022. The outer dimensions of the container leading to 0.195 m3 were used in the OPERA Safety Case and also in this safety case.

stainless-steel, CSD canisters. Vitrified HLW canisters are passively cooled in double sided wells in order to remove the decay heat during storage. Compacted hulls and ends constitute non-heat generating HLW; this type of waste requires no special cooling measures during storage. In OPERA, the Belgian supercontainer concept was adopted as a reference design for HLW packaging. This concept provides radiation protection during the operational disposal phase, as well as physical containment in the post-closure phase, using a carbon steel overpack around the stainless-steel CSD canisters and a concrete buffer. For passive safety (see section 3.4.8.1), as further explained in section 6.3, integrity of the carbon steel overpack is needed for as long as the waste heats the clay host rock. In OPERA, no distinction was made between the packaging of heat-generating vHLW (CSD-v) and non-heat generating HLW (CSD-c) but future work on another standardised container for CSD-c was envisaged (Verhoef et al., 2017). A carbon steel overpack for a disposal package for CSD-c is not needed, since it is not heat generating and packages will be emplaced in the GDF with sufficient spacing from heat-generating wastes that the surrounding host rock is not significantly heated. Also, the content of ^{137}Cs, whose activity is mainly responsible for external radiation, is at least 100 times lower for CSD-c compared to CSD-v canisters. The maximum activity is 6600 TBq for CSD-v (AREVA, 2007) and 65 TBq for CSD-c (COGEMA, 2001). The low heat generation means that the dimensions of the disposal package can be smaller, or more than one CSD-c can be put in a single disposal package.

In the current conceptual design, the diameter of all disposal packages is 2 m (see Figure 4-1) in order to standardise the handling equipment. Radiation protection calculations in Chapter 6 show that the thicknesses of the carbon steel and concrete buffer in the supercontainers provide sufficient shielding for the heat-generating HLW when each disposal package for heat-generating waste holds one HLW canister, for CSD-v as well as SRRF. For CSD-c, no overpack is needed and seven non-heat generating canisters are envisaged to be contained in each disposal package. The disposal volume for CSD-c in the present study is thereby more than 7 times smaller than assumed in OPERA.

4.2 Layout of the disposal facility

The OPERA Safety Case did not consider in detail how the GDF might be constructed, operated and closed. Demonstration of feasible approaches for each of these activities will progressively reduce uncertainties in project planning. The present safety case begins to look at methods and technologies in more depth, starting with a description of GDF layouts and the currently considered pre-constructional activities.

In the OPERA Safety Case, access to the underground structure of tunnels was proposed to be via an inclined ramp. During construction, ramp access from the surface allows easier transport of materials and equipment into, and excavated host rock out of, the underground facilities, for example using conveyor belts. Ramps may also play a role during the operation and closure of the disposal facility, for example during the emplacement of waste packages or backfill. The ramps considered in France and Switzerland are excavated in limestone, which is a relatively hard and competent rock. In the Netherlands, accessways need to penetrate through several unconsolidated sandy formations overlying the Paleogene clay formations. Closed Mode TBMs such as Slurry Shield TBMs are one means of tunnelling used in such formations, but these have only been demonstrated to depths of less than 100 m. An inclined ramp to great depth in poorly consolidated sediments would be difficult to construct. Consequently, in the current conceptual design, access is by vertical shafts, which can be constructed through all types of formation, using techniques for which there is considerable experience worldwide.

The conceptual design in OPERA was for all types of waste to be emplaced at the same depth. Table 4-2 shows the number of disposal tunnels for each type of waste listed in Table 4-1 for a single level GDF (see Figure 4-4) in which waste is disposed of at 500 m: i.e., the same point of departure as used in OPERA. For a multi-level GDF concept, HLW could be disposed of at 500 m depth, but LILW could be disposed of at shallower depth, for example at 100 m and 200 m depths. This conceptual multi-level model is used in the current COPERA study.

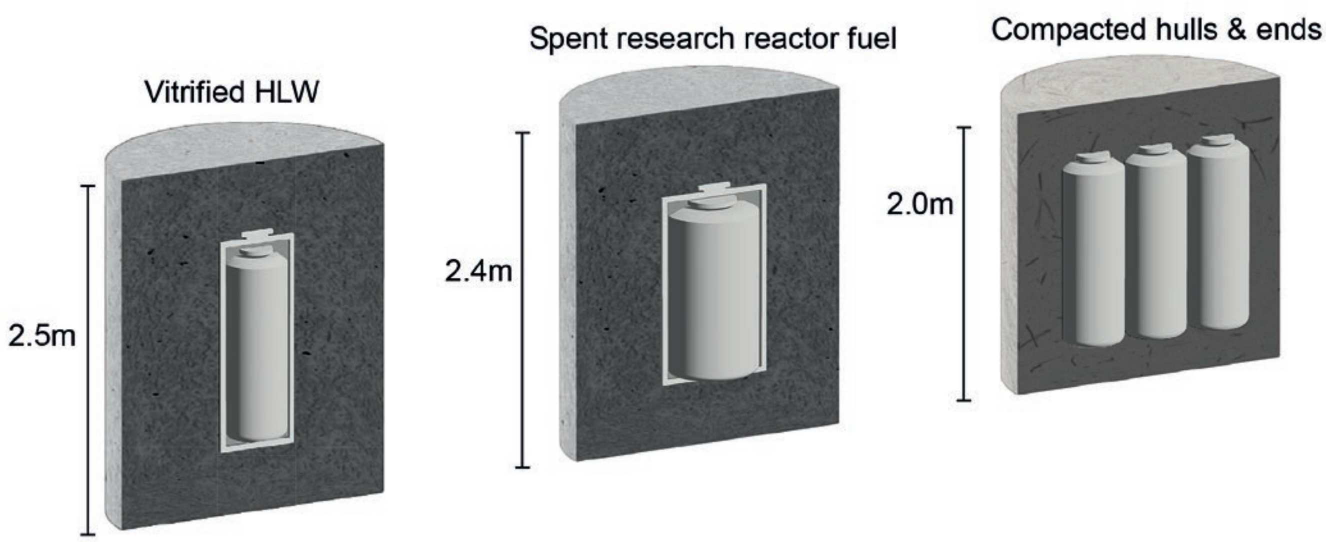


Figure 4-1: Packaged HLW. Diameters for all packages is 2 m to facilitate the handing of HLW in the underground facility.

Waste Category		Packages for disposal			Disposal tunnels		
		Length[L]/Height[H] (m)	Diameter (m) or Width (m) × Length (m)	N packages	N	$\varnothing_{out}/\varnothing_{in}$ (m)	Length (m)
Spent research reactor fuel (HLW)		2.4	2.0	244	6	5.0 / 4.0	100
Vitrified HLW		2.5	2.0	478	12	5.0 / 4.0	100
Compacted hulls & ends (HLW)		2.0	2.0	72	1	5.0 / 4.0	150
Dismantling waste (LILW)		1.7	1.6 x 1.7	826	4	5.0 / 4.0 *(5.0 / 4.6)*	185 *(175)*
TE-NORM (LILW)		1.7	1.6 x 1.7	12600	27	5.0 / 4.0	400
Processed LILW	200 l	0.88	0.59	100000	21 *(20)*	5.0 / 4.0 *(5.0 / 4.6)*	250 *(200)*
	1000 l	1.25	1.00	8400	7 3 *(8)*	5.0 / 4.0 5.0 / 4.0 *(5.0 / ~~4.6~~4.0)*	150 200 *(200)*

[L] LHLW packages and 200 l drums are to be disposed of horizontally along their length, [H]Konrad containers (Dismantling waste and TE-NORM) and 1000l containers along their height, N=Number

Table 4-2: Number and dimensions of disposal tunnels for disposal at 500 m depth and, in italics and brackets, for disposal at 100 and 200 m depth.

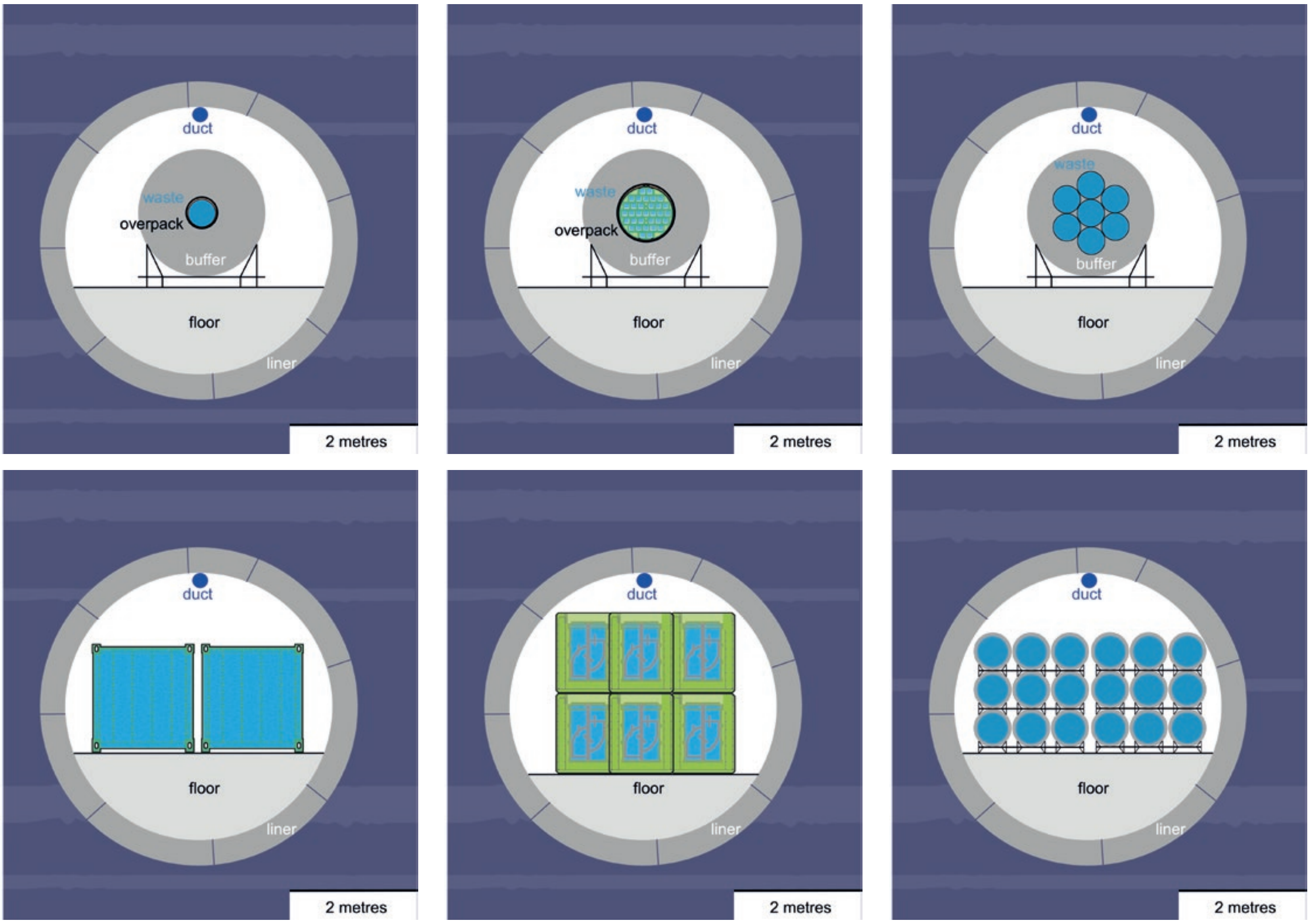


Figure 4-2: Cross sections of disposal tunnels with outer diameters of 5 m for disposal of waste at 500 m depth with 0.5 m liner thickness. At the top concrete containers with HLW: vitrified HLW and SRRF (both heat generating) and non-heat generating HLW (compacted hulls and ends). At the bottom packages with processed LILW: Konrad containers (dismantling waste or TE-NORM), 1000 l concrete containers and 200 l drums.

Figure 4-2 shows cross sections of disposal tunnels at 500 m depth. The disposal tunnel diameter is 5 m and the required thickness of the liner has been calculated using the same methodology as in OPERA (see section 4.4.2). The optimum dimensions of the transport tunnels will be a function of the disposal depth and the formation properties, but in the current conceptual design they are, for simplicity, taken to be independent of the depth, with an outer diameter (Øout) of 10.0 m and an inner diameter (Øin) of 8.4 m (see section 4.4.3).

Two important features of the conceptual design depend on the mechanical loading due to lithostatic pressure of the overburden and, thereby, on the disposal depth:

1. The thickness of the tunnel liners (see section 4.4.2);
2. The distance between the disposal tunnels (see section 4.4.4).

A thicker liner is required for a larger disposal depth, which reduces the available space to dispose of waste. Especially for LILW, this is a disadvantage, since stacking of waste packages is envisaged. A thinner liner (0.2 m) appropriate for a depth of 100 and 200 m shows that 25% more 200 litre drums can be stacked (see Figure 4-4) than for the thicker liner shown in Figure 4-2 and Figure 4-3, for a disposal depth of 500 m. The length of the disposal tunnel can thus be reduced or fewer disposal tunnels could be constructed. Almost 1.5 km less length of disposal tunnels need to be constructed for a multilevel GDF, compared to a GDF at a single disposal depth of 500 m.

At smaller disposal depth, the distance between the disposal tunnels required to maintain mechanical stability can also be reduced, meaning that the length of the transport tunnels can also be reduced. In addition to the need for fewer disposal tunnels, this means that in total, there is a saving of almost 1.2 km in the length of transport tunnel required. Clearly, the eventual design of a multilevel GDF can optimise the depths and dimensions for all the tunnels, with tunnel diameters, liner thicknesses and tunnel spacings being the key variables that can be adjusted to match geotechnical conditions.

4.2.1 Disposal of waste at a single depth

In both CORA and OPERA, a clay formation with a thickness of 100 m was considered to provide sufficient containment in the post-closure phase and this is also assumed in the COPERA safety case. The OPERA (Verhoef et al., 2017) GDF conceptual design had orthogonal X-crossings between transport tunnels. With the current state of technology, it is not feasible to construct tunnels with the same outer diameter that cross each other. A solution could be to remove the need for crossings by using a curved transport tunnel. Some recent developments demonstrate feasible curvatures in an underground tunnel constructed with a TBM. In Beishan (China), a TBM is constructing a ramp with a diameter of 7 m in granitic (crystalline hard) rock, with a radius of curvature of 255 meter. However, excavating the poorly consolidated clays considered in this report requires the use of immediate tunnel support. The Corbulo tunnel in the Netherlands, has curved horizontal sections and is constructed in soft sediments that required a concrete lining support. For this type of construction, the concrete segments can be of varied lengths and curved supported liners can be made by using shorter concrete segments at the inner part of the bend of the traffic tunnels and longer segments in the outer part of the bend (ter Voorde, 2022).

Two phases are currently foreseen for the construction of the GDF where all the wastes are disposed in a clay formation at a single depth. In the first phase, the shafts, half of the length of the transport tunnel and the disposal tunnels for processed LILW (200 litre drums and 1000 litre containers) and dismantling waste (Konrad containers) are constructed. Emplacement of the waste packages begins after construction is completed, in order to separate the construction activities completely from activities involving radioactive materials. The experience gained during construction in the first phase and possibly knowledge of new emerging techniques may lead to optimisation in the second phase.

The waste acceptance facility is connected with the underground facility by a vertical shaft. Figure 4-4 shows a schematic of an operational disposal facility after two constructional phases. Cross sections are shown in Figure 4-2.

4.2.2 Disposal of waste at multiple depths related to the hazard potential of waste

Multiple Paleogene clay formations are present at different depths at locations in the Netherlands (see Figure 5-3 in Chapter 5), which would allow a multi-level GDF concept, with waste isolated as a function of its hazard potential. This also allows operational segregation of the different types of waste and minimizes any interactions between them as they degrade in the post-closure phase. A three-level example is considered in COPERA:

- Level 1, at about 100 m depth, for the 200 litre drums and dismantling waste. This type of waste contains short-lived radionuclides and is often disposed of in surface or near-facilities in other countries. The thickness of clay must be sufficient to prevent inflow of water into the disposal facility.
- Level 2, at about 200 m depth, for the 1000 litre concrete containers and depleted uranium. Figure 4-2 and Figure 4-3 show that the waste is encapsulated in 200 litre drums that are placed in 1000 litre containers to provide sufficient shielding for the short-lived radionuclides. The activity of depleted uranium is predominantly from long-lived radionuclides which motivates this increased disposal depth. A natural analogue provides confidence that a thickness of host rock between the waste and other formations of some tens of metres can provide an adequate barrier: the 5 to 30 m of clay surrounding the rich uranium ore body at Cigar Lake was sufficient to prevent any radiological signature from the ore bein detectable at the surface (Smellie, 2004).
- Level 3, at about 400/500 m depth, for HLW. A thickness of 100 m of clay for HLW i.e. similar to the second (CORA) and third (OPERA) national programmes. The overlying clay layers at Levels 1 and 2 will also contribute to containment of the HLW.

These depths and thicknesses are examples and are nominal points of departure for the design concept. The actual depths for each level of a disposal facility depends on the geological setting of the Paleogene clay formations and would be site-specific. Waste specific assessments could elucidate appropriate required thicknesses of suitable clay formations, appropriate depth ranges and clay contents, taking account of the hazard potential as well as the uncertainties in the alteration processes of each waste form.

A three level GDF can be constructed in three different phases, beginning with the levels nearer to the surface. Each construction phase is followed by emplacement of waste packages in order

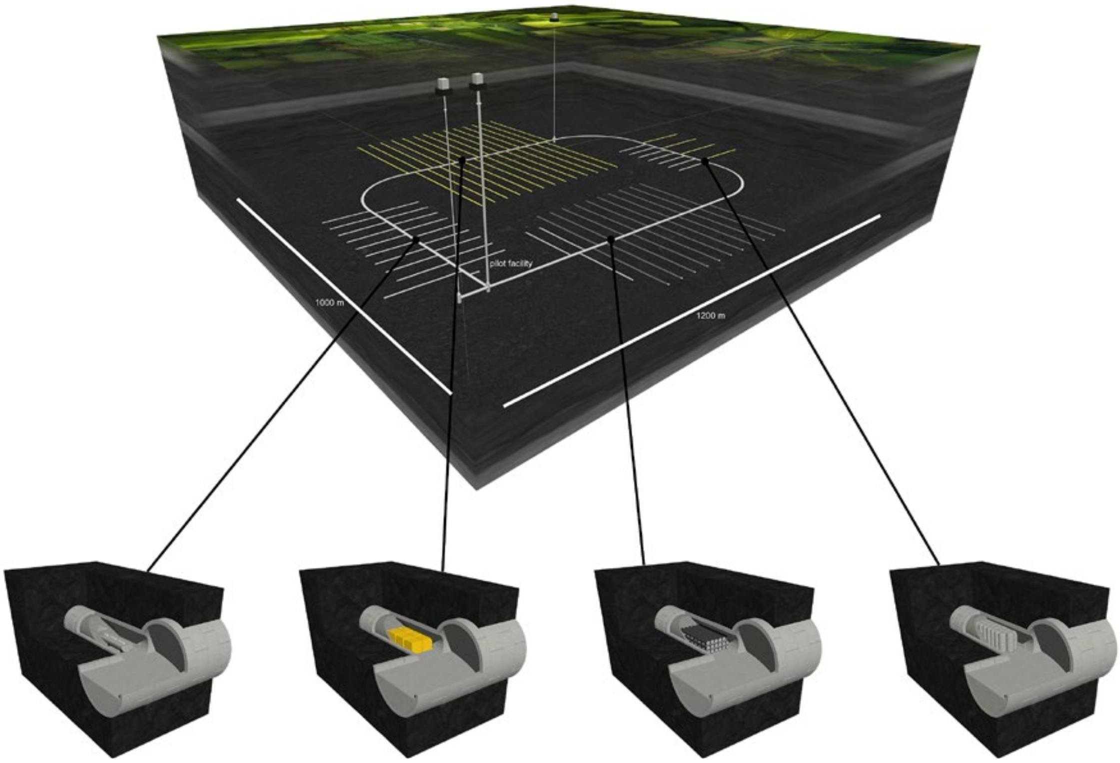


Figure 4-3: Schematic of an operational disposal facility in poorly indurated clay at a single disposal depth with different types of waste: vHLW in supercontainers, Konrad with depleted uranium, 200 l drums and 1000l containers. In a pilot facility (tunnel with smallest length), long-term research is conducted to verify the assumptions made.

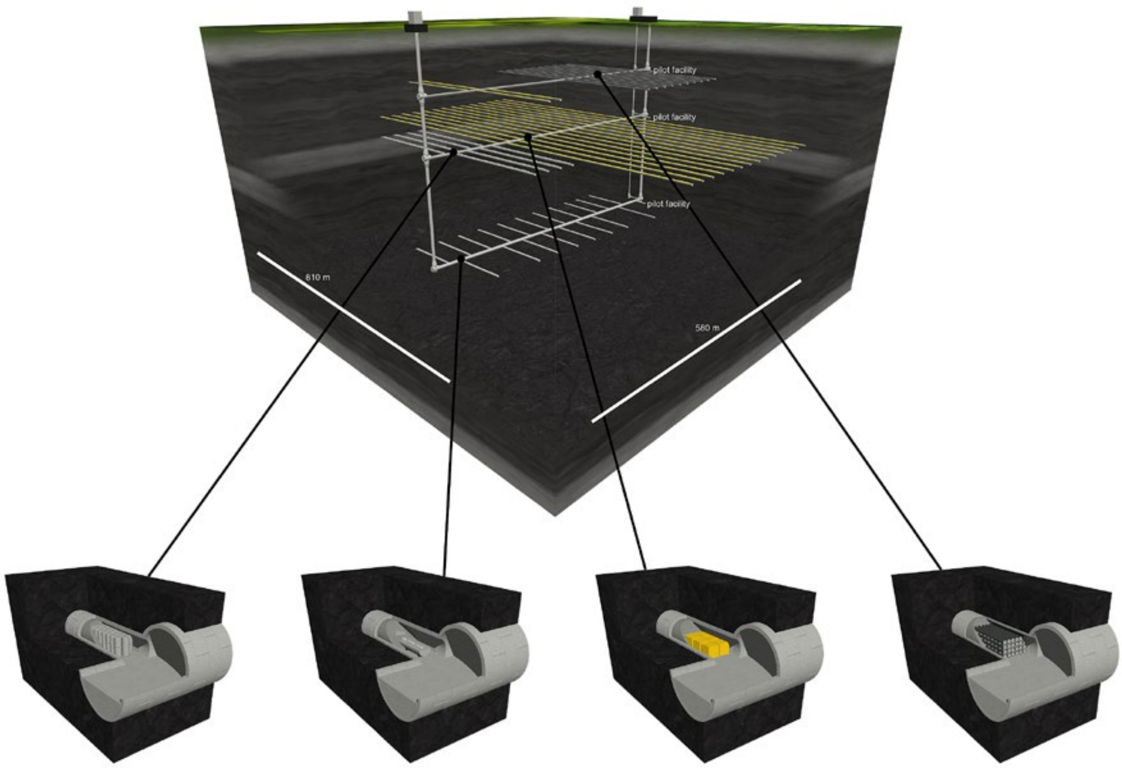


Figure 4-4: Drawing of an operational disposal facility in poorly indurated clay with three disposal depths; the waste acceptance facility is located at the entranc,e by the waste and personnel shaft. Chambers (see Figure 4-7) at crossings between shafts and the transport tunnels, each with a length of 580 m (left). Disposal tunnels for the different types of waste at different levels (right).

to separate construction from operations involving radioactive materials. The experience gained and the possible emergence of new technologies can then be effectively incorporated. Figure 4-5 shows a schematic of an operational three level disposal facility for the packaged waste inventory shown in Table 4-2 and Konrad containers placed with their longest dimension across the tunnel section, as shown in Figure 4-2.

4.3 Pre-constructional activities for the GDF

4.3.1 Site selection

Site selection will require an extensive geological database. The Netherlands is one of the few European countries that has successfully implemented the European INSPIRE Directive (EC, 2007) through the programme BasisRegistratieOndergrond (BRO). All publicly available underground data are compiled and accessible for any citizen and company through the DINOLOKET website, which is controlled by the Dutch Geological Survey (TNO) and hosted by the responsible Dutch Ministry. However, the third national programme (OPERA) showed that there are currently insufficient data and detail available to develop the Dutch GDF concept much further than the conceptual design stage (Vis et al., 2016). The data in DINOLOKET show the depth of the top of Paleogene clay formations and the thickness of each clay formation at many locations in the Netherlands but there is little information on their properties at these depths. COVRA envisages a process in which several volunteering regions are investigated in more detail, and municipalities that agree to host the GDF can use these publicly accessible data to learn more about the underground at their site.

As explained in Chapter 2, the construction of a URF may be considered as part of a site selection process, for example in order to measure in-situ properties of the clay host rock. The URF would be constructed after sufficiently positive results have been obtained by the research on clay material from drill cores and by geophysical and hydrogeological testing in boreholes. The URF may later be converted into a pilot facility for the GDF. A store and a characterisation facility for the clay cores that are extracted need to be included in the surface facilities and the boreholes are envisaged to be used for monitoring. It is intended that extensive research on cored clays is performed before a site is selected, using techniques based on international experience, e.g., Mazurek et al. (2023).

4.3.2 Land purchase & archaeological assessments

Land needs to be purchased for the surface facilities that will be required to construct and operate the underground facility. The length and width (footprint) of the underground facilities of the GDF could be larger than the area needed for the surface facilities. The Netherlands do not yet have a policy on the required land area to be purchased for disposal of waste. It is therefore uncertain whether this area should include an exclusion area to reduce the probability of human intrusion.

No underground constructional work in the Netherlands can be performed without an archaeological assessment. First, it is necessary to be identify whether there is any indication of archaeological value at the chosen site. If there is, then archaeological research precedes any construction activity. This research can also be requested if an archaeological finding is made during the construction.

4.3.3 Construction of surface facilities

Surface facilities are the immediately visible part of an operating a GDF and need to fit into the landscape, with the approval of the local population. For now, these surface facilities have been assumed to be present directly above the GDF, within the footprint of the underground workings. The surface facilities are required for receiving, inspecting and packaging the waste for disposal. Surface facilities also include support infrastructure for construction, operation and closure activities in the GDF. Some surface facilities need to be constructed in advance of the construction of this underground facility.

Around 13 km of disposal tunnels are needed in the underground facility. There are precedents in the Netherlands for tunnels of similar length, such as the Westerschelde tunnel, which has two traffic tunnels, each with a length of 6.6 km. A factory for manufacturing the concrete tunnel liner segments was built nearby in order to minimize transport activities, but also to use the excavated soil in the manufacturing of the segments. A similar approach is assumed for the GDF. Other currently foreseen surface facilities are:

- a core store for the clay cores extracted during the site selection process;
- a sieving and rinsing facility for reuse of the excavated soil for the manufacturing of the concrete segments and concrete containers for the disposal packages for HLW and ILW.
- a facility for the inspection /acceptance of packaged waste with appropriate security facilities;
- a hot cell for the packaging of HLW;
- a visitor centre and office.

The activities within the facility for packaging HLW are:

- encapsulation of heat-generating HLW stainless-steel canisters in a carbon steel overpack;
- emplacement of the encapsulated canisters in concrete containers;
- pouring concrete closures in the top of these concrete containers;
- allowing hardening for 28 days;
- tilting the hardened waste package on a steel rack for transport by a forklift truck.

4.4 Layout of the underground part of the GDF at tunnel scale

4.4.1 Inner diameter of the tunnels

The minimum inner diameter of the tunnels is set by requirements for the transport of material during construction and operation of the underground facility. Compact forklift trucks are considered for emplacement of waste packages, especially those waste packages that can be stacked: i.e., the 200 litre drums and 1000 litre concrete containers. The manoeuvring space for the forklift truck is less than the internal diameter of the tunnel due to the presence of the floor or base plinth beneath the waste packages. A minimum internal diameter of 4 m is used in order to have sufficient space for a forklift truck transporting LILW packages. Such forklift trucks are currently used by COVRA in the storage facility where 200 litre drums and 1000 litre containers are stored.

With a concrete liner thickness of 0.5 m, the outer tunnel diameter would be 5 m. The disposal tunnels cross the transport tunnel.

Investigations carried out in the Belgian programme show that for mechanical stability at crossings, the maximum outer diameter of the tunnel should be half the outer diameter of the transport tunnel, with the currently available techniques (Leonard et al., 2018). This was also proposed towards the end of OPERA (Yuan et al., 2017). The outer diameter of the transport tunnel is therefore set to 10 m in the disposal concept.

4.4.2 Tunnel liner

The load on the concrete liner increases with disposal depth, requiring thicker liners, which are also needed for larger diameters of tunnel (Arnold et al., 2015). The design specification for the mechanical support therefore requires definition of the depth, mechanical properties of the host clay and the strength of the support material. Figure 4-6 shows the hierarchical set of requirements that will lead to the design specification for the liner.

The liner is required to stabilise the dimensions of openings in the underground for decades: i.e., for a sufficiently long period to retrieve the waste up to the expected time of GDF closure, if needed (L4-CD-LINE-CONTR-01). The technical details at the end of this section on existing tunnels show that the techniques proposed to implement the liners in clay are proven.

In the URF in Boom clay in Belgium, it was observed that salt deposits formed between hardened concrete tunnel liner segments (see Figure 4-9). These arise from the inflow of Boom Clay pore water, which evaporates, due to ventilation. In a GDF, a significant hydraulic gradient will be present between the clay host rock and the inside of a tunnel during the operational phase. The pore water pressure in the clay host rock depends on the disposal depth (e.g., 5 MPa at 500 m). The air inside the tunnel is at atmospheric pressure and contains nitrogen, oxygen and traces of water and carbon dioxide. The partial pressure of water is very small, orders of magnitude smaller than 1 atmosphere pressure (0.1 MPa).
The hydraulic gradient between the pore water in the clay and air in the tunnel is thus so large that the clay would become dehydrated if no tunnel liner were present. The lining in the Belgian design is made from a material with a low permeability so that flow of water is only possible between the concrete segments, limiting water influx to the disposal facility. At the same time, the low permeability of the liner prevents the clay host rock from drying by the ventilation air which ensures that any water becoming available in that part of the clay host rock in the vicinity of the EBS can be used for the self-healing of excavated induced cracks (see Chapter 5).

The EURAD-1 WP MAGIC project shows that for a porous medium such as concrete, higher strength is correlated with lower water permeability. Concrete with a cylindrical compressive strength of 80 MPa was used for the construction of the liner for the Belgian URF, and this has been demonstrated to limit inflow of water sufficiently. More details about the permeability of clay and concrete can be found in Chapters 5 and 6, respectively.
Some permeability values of concrete and clay are shown in Table 6-1 in section 6.1.3.

The potential impacts on post-closure safety must be addressed for any type of engineered material that is introduced for operational safety reasons. In the multibarrier system with a clay host rock, the initial post-closure function of the tunnel liner is to prevent the transfer of lithostatic load to the waste packages (see Chapter 3). The self-healing of excavation cracks taking place in the operational phase due to the lack in evaporation of the clay

L1-NPRA-03: Any handling of waste shall be controlled.

L2-COV-03: Simple, robust, reliable and proven techniques shall be used.

L3-D-NPRA-02: Waste shall be retrievable during the operational phase of GDF through until its closure

L4-CD-DEVI-CONTR-01: The waste package shall be emplaced under human control.

L4-CD-LINE-CONTR-01: The liner shall ensure sufficient mechanical support in the operational phase of the facility.

L4-CD-LINE-CONTR-02: The liner shall limit the water inflow

L5-CD-DEVI-CONTR-01: The internal diamater shall be 4 metres in order to have sufficient manouvring space for the forklift truck.

L5-CD-LINE-CONTR-01: Mechanical support by the liner as a function of the radius of the tunnel and thickness of the liner **should be larger than the pressure** as a function of the radius of the tunnel and the properties of clay at the depth of the facility.

L5-CD-LINE-CONTR-02: The liner shall be 'waterproof' i.e. impermeable in engineering terms.

L6-CD-LINE-CONTR-01&2: A thickness of 0.5 metre for concrete with a compressive strength of at least 80 MPa after 28 days hardening at a depth of 500 metres and an internal diameter of 4.0 meter.

Figure 4-5: Hierarchical set of requirements that contribute to handling the emplacement of waste packages with techniques that have been proven and determine the internal diameter (manoeuvring space: L5-CD-DEVI-CONTR-01). The required thickness of the liner and the properties of concrete used depend on the tunnel dimensions, the depth and the mechanical properties of the host rock.

by ventilation, also limits the presence of a zone with a higher permeability towards the shaft that can act as a fast pathway for radionuclides. The hierarchical set of requirements to determine the design specifications for a concrete liner that contributes to the post-closure safety differ from the engineering requirements and is shown in Chapter 6.

Some technical details on tunnels constructed elsewhere are gathered in Table 4-3 in order to support the basis of our assumptions. Concrete segments have been used in all the tunnel examples and, therefore, concrete was chosen in the example design specification in Figure 4-6. These details have been collected mainly from the Netherlands expert centre on infrastructural underground works (www.cob.nl) and from other websites. Boom clay was the only Paleogene clay in which tunnelling construction projects were reported.

The Westerschelde tunnel with reinforced concrete segments, used a lower strength class for concrete (45 MPa) than the cylindrical compressive strength of 80 MPa for the Belgian URF at Mol (Verhoef et al., 2014). Reinforced concrete supported tunnels with an internal diameter of 8.4 m are also envisaged in France for the Cigéo GDF at 500 meters depth. The liner thickness is about 0.8 m (ANDRA, 2016). We use the same dimensions for the transport tunnels in this safety case. The use of reinforced concrete requires a different definition of the design specification than used in Figure 4-6, and would include definition of requirements for concrete and the density of steel rebars of a specified diameter. In the current conceptual design, the lining in the transport tunnels is made with reinforced concrete to facilitate the intersections between transport and disposal tunnels (see section 4.5.3). The lining in the disposal tunnels is made with unreinforced concrete. Unreinforced concrete was proposed for all tunnels in the OPERA programme.

Name	Purpose	Main tunnel			Tunnel perpendicular to main tunnel		
	Building period	External Diameter (m)	Liner Thickness (m)	Length (m) / Depth (m)	Intersection every (m)	Dimensions	Thickness (m)
Westerschelde	Public road / 1997 - 2003	11.3	0.45 (reinforced)	6600×2 / 40	250	2.10 m ×1.50 m	0.40
Sluiskil	Public road / 2011 - 2015	11	0.45 (reinforced)	1150×2 / max. 33m	250		
Mol	URF	4.7 Connecting gallery (2002)	0.40	90	-	2.5 m diameter PRACLAY (2007)	0.30

Table 4-3: Details of tunnelling examples in Boom clay

L1-NPRA-03: Any handling of waste shall be controlled.

L1-NPRA-04: Waste shall be enclosed by a series of engineered barriers

L2-COV-04: Materials for which broad experience and knowledge exists, shall be used.

L3-D-NPRA-02: Waste shall be retrievable during the operational phase of the GDF through until its closure.

L3-D-IAEA-02: The radionuclides in the waste shall be contained by the engineered barriers and natural barriers until radioactive decay had significantly reduced the hazard potentiel posed by the waste.

L4-CD-FLOO-CONTR-01: The floor shall ensure sufficient mechanical support for the weight of the package and transport device.

L4-CD-FLOO-CONTA-03: The low permeability of the floor shall minimize the access to water to the waste package in order to limit its alteration and minimze further spreading of radionuclides into the clay host rock.

L5-CD-FLOO-CONTR-01: The floor shall ensure sufficient mechanical support for the axle load of a loaded transport device.

L5-CD-FLOO-CONTA-03: The material used to construct the floor shall have a smaller diffusion value for water than the clay host rock.

L6-CD-FLOO-CONTR-01&CONTA-03: A material with a compressive strength of at least 4 MPa and saturated diffusion value for water of 5×10^{-14} m^2/s

Figure 4-6: Hierarchical requirements set to determine the design specification for the tunnel floor, allowing the emplacement of waste packages to facilitate handling in the operational phase and preventing a fast pathway of released radionuclides in the post-closure phase to contribute to containment in the post closure phase.

4.4.3 Floor

There will be many transports by small and large forklift trucks and air cushion vehicles during the construction and operation of the GDF. These require a flat floor with sufficient strength, during the operational phase of the GDF. Figure 4-7 shows the hierarchical set of requirements to derive the design specifications. The contact area between the floor and the vehicle is smallest for the tyres of the trucks, giving the highest point loads, which thereby constrain the design requirement of the floor.

In the manufacturing of traffic tunnels, concrete debris is used to provide sufficient floor strength and this is finished with an asphalt layer to obtain a flat and stable floor. Asphalt has a compressive strength of around 4 MPa, which is therefore included as the necessary strength of the floor in the design specification in Figure 4-6. If an engineered material used for operational safety is not removed, the choice of materials for the floor will also be controlled by the post-closure safety. In our conceptual design, the floor is not removed, so its properties should also be suitable for the post-closure phase.

In the post-closure phase, the hydraulic gradient between the pore water in the clay host rock and the pore water in the porous engineering materials of the tunnel floor, liner and EBS will eventually disappear. At this stage, diffusion may be assumed as the dominant transport mechanism for water and solutes, including dissolved radionuclides from the wastes. In order to prevent the floor becoming a preferential pathway for radionuclide migration in the post-closure phase, it is therefore required that the material used to manufacture the floor has a smaller value for diffusion of water than the clay host rock.

A floor material that satisfies the required compressive strength in the operational phase and also the required slow transport properties of water in the post-closure phase is the same cementitious material that will also be used as a backfill; this is foamed concrete, as described in sections 4.7 and 6.1.3.

4.4.4 Distance between disposal tunnels

The construction of disposal tunnels causes mechanical stresses within the Paleogene clay. The thermal load of HLW may also increase the pore water pressure in the clay. Higher pore water pressures are associated with higher plasticity of the clay.
For heat generating HLW, the thermal load will be less restrictive on design than the mechanical stresses caused by the construction of a disposal tunnel. This is a result of the long period of pre-disposal cooling in above-ground storage that is envisaged in the current Dutch programme and the large volume of the disposal packages foreseen for HLW. Consequently, only mechanical aspects are considered in the calculation of a safe distance between disposal tunnels.

Poorly indurated clay formations such as the Paleogene clays have relatively low mechanical strength. For example, Boom Clay has an unconfined compressive strength (UCS) of 2 MPa (e.g., Delage (2013) . The lay-outs shown in Figure 4-4 and Figure 4-5 would result in stresses between the disposal tunnels that would exceed the compressive strength of poorly indurated clay if the stress were assumed to be uniform between disposal tunnels. In fact, stress calculatations (e.g. Arnold et al. (2015), show that the regions of clay in the vicinity of the tunnel are in a plastic regime, while an elastic regime is maintained further away. The volume that remains in the elastic regime should be large enough to provide sufficient mechanical stability.

In CORA, the extent of the plastic zone was calculated to be 7 m from the tunnel centre for a tunnel outer diameter of 4.6 m at a depth of 500 m, based on the available geotechnical properties. For a centre-to-centre distance of 50 m between the tunnels, the elastic zone is 36 m wide. The size of this elastic domain is considered enough to support the overburden and to create stable mechanical conditions between adjacent tunnels (Van de Steen and Vervoort, 1998). In OPERA, this was translated into a design requirement limiting the plastic radius to one third of the centre-to-centre distance between disposal tunnels (Arnold et al., 2015) but this design requirement does not include the impact of disposal depth. The Westerschelde tunnel is also constructed in Boom Clay, but at a smaller depth, and the tunnel diameter is larger. The two traffic tunnels are separated from one another by only 12 m at a depth of 40 m (Kooijman, 1996). It is therefore expected that the stress induced by the overburden plays a role in the determination of a safe distance between the tunnels. For now, the centre-to-centre distance between the disposal tunnels at the largest disposal depth in Figure 4-3 is the same distance of 50 m that is used in Figure 4-2. For a multi-level GDF, the distances between disposal tunnels at smaller depths have been chosen to be between that of the Westerschelde tunnel and the 50 m proposed for our largest disposal depth: i.e., a centre-to-centre distance of 20 m at 100 m depth and 40 m at 200 m depth.

4.5 Construction

4.5.1 Shafts

Construction activities are separated from the operational activities as described in section 4.2. Transportation of excavated clay to the surface never occurs at the same time as the emplacement of waste packages. Two shafts with an internal diameter of 6.5 m are to be constructed for transport of material. This is larger than the internal diameter of 5 m used in OPERA but smaller than the 8 m proposed in the Belgian programme, since the disposal packages foreseen in the Dutch case are smaller. A smaller shaft with an internal diameter of 2 m is a rescue shaft and reserved for personnel. This three shaft disposal concept is similar to the earlier published Belgian design from ONDRAF/NIRAS (e.g., Leonard et al. (2018). The Belgian design is still in a development stage and the necessity for more shafts to ensure operational safety is still being studied.

The material shafts are connected to the transport tunnels, which have an outer diameter of 10 m. The internal diameter of the transport tunnel can be larger than the internal diameter of the shaft since the tunnelling shield of the TBM is transported down the shaft in pieces and assembled underground. A chamber is envisaged to be built at each intersection of a shaft and a transport tunnel (see Figure 4-2 and Figure 4-3). The shafts will pass through sandy formations that overlie the clay formations. The construction of these shafts requires freezing of the water in the sandy formations. An impermeable prefabricated lining will be placed, using concrete segments, finished with shotcrete and asphalt.
Finishing with asphalt prevents water inflow once the freezing is stopped. Belgian experience at the URF in Mol for the construction of the second shaft (1997-1999) down to a depth of 225 m

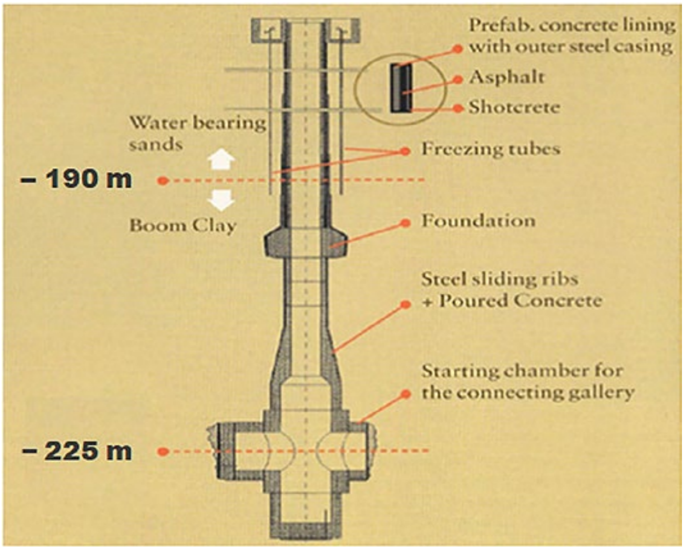


Figure 4-7: Layout and design of the second shaft of the Belgian URF[11] (Li et al., 2023).

demonstrated that freezing is only needed in the sections where the shafts intersect water bearing sands. Boom Clay could be excavated without freezing (Li et al., 2023). After emplacement of steel sliding ribs, concrete was poured. Figure 4-8 shows the shaft at the URF in Mol.

4.5.2 Tunnelling techniques

Open Face TBMs are used to construct tunnels in soft rocks with low water inflow, such as the Paleogene clays. The face is excavated using either a cutter head or a road header within a shield.
The shield is jacked forward, and cutters on the front of the shield excavate the rock to the same circular profile. The transport of excavated material can take place by a conveyor belt or by small vehicles (Bernier et al., 2003).

11. The preference in Belgium for the underground laboratory in Mol has also become URL instead of URF. But as explained in section 2.3.1 for the French case, the IAEA still holds the URF network as a name.

Hardened concrete segments are emplaced immediately after a certain length of excavation has been achieved. The wedge block lining technique finishes the tunnel ring: i.e., a conical small segment is pushed between two ordinary segments for the construction of each ring. The use of TBMs with immediate emplacement of hardened concrete segments has been demonstrated in Paleogene clays (see section 4.4.2). The construction rates of the tunnels in Paleogene clay listed in Table 4-3 are:

- the Westerschelde tunnel at a depth of about 40 m had a construction rate using a Slurry Shield TBM of 12 m per day and used reinforced concrete segments;
- the Sluiskil tunnel at a maximum in depth of 33 m had a construction rate using a Slurry Shield TBM of 13.5 m per day and used reinforced concrete segments;
- the URF in Mol at a depth of 225 m had a construction speed of at least 2 m per day using an Open Face TBM (Li et al., 2023) for the connecting gallery and an average of 3 m per day for the PRACLAY gallery; both tunnels used high strength unreinforced concrete segments.

Slurry Shield TBMs have not yet been demonstrated at large depth and are therefore not assumed in the current conceptual design. The rates of removal of excavated clay and of supply of liner segments in in a GDF would be similar to the traffic tunnel examples if an inclined ramp could be used. However, as stated in section 4.2, it is currently assumed in COPERA that a ramp cannot yet be constructed in Paleogene clay at large depth.

The experience of construction of the URF at Mol provided valuable information for the assumptions to be made for the construction rate of a tunnels for a GDF. The excavation rate for the connecting gallery was limited by the capacity of the shaft hoisting system (Li et al., 2023). This limitation is also expected for the Dutch GDF, despite the larger envisaged internal diameter of the material and waste shafts, since the internal diameter of the transport tunnel is also assumed to be larger: i.e., 8.4 m. In addition, inspections for nuclear safeguards purposes may increase the time for construction, but this is currently not considered in our time estimates.

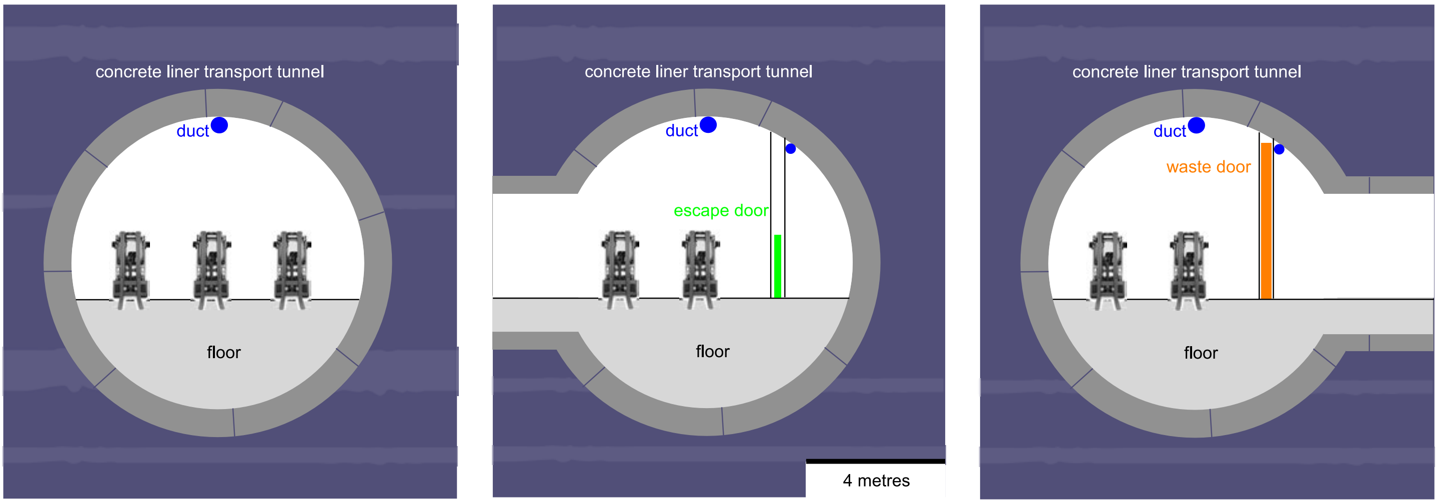


Figure 4-8: Transport and safety arrangements in the main tunnel during construction (left) and operation, with a fire wall and escape door (middle) and waste door (right). Inner diameter is 8.4 m.

4.5.3 Intersections

Constructing intersections will be an expensive part of tunnelling and these are therefore minimized, taking into account, however, that operational safety requires sufficient escape routes. Intersections are made in road tunnels to allow people to leave the tunnel in case of emergencies; they were foreseen in OPERA (Verhoef et al., 2017) and for the Belgian GDF (Leonard et al., 2018) and are also foreseen in the ramp in Bure (ANDRA, 2016). The Belgian design is still in a development stage and rescue chambers in the dead-end disposal tunnels are currently considered. Intersections between transport tunnels and disposal tunnels are also needed.

Intersections are frequently used in public road tunnels and those in Table 4-3 have two tunnels in order to make one-way traffic possible, with escape pathways between the tunnels. Exclusion of intersections would minimize the costs. Exclusion of intersections has been achieved in the design of the Groene Hart Tunnel. The reference design had two one-way tunnels for the subway and intersections (Leendertse and Burger, 1999). In the contractor's design, there was a single tunnel with a larger diameter for the subway and sufficient escape pathways between the two rails. This design was cheaper than the two smaller tunnels with intersections for escape routes (COB, 2022). Providing sufficient escape routes without two (transport) tunnels is also used for the design of the GDF, as shown in Figure 4-8. In the constructional phase, several small forklift trucks can fit in the transport tunnel and these trucks can, for example, carry a one cubic metre container filled with excavated clay material towards the shaft. In the operational phase, these trucks can lift and stack the 200 litre drums and 1000 litre concrete containers, carrying them from the shaft to a disposal tunnel and having sufficient exit pathways during operation of the GDF via a walled-off passageway with escape doors.

Orthogonal X-crossings were assumed in OPERA. Only T-junctions are considered in the Belgian GDF (Leonard et al., 2018) and we make the same assumption here, in order to minimize weak spots in the concrete liner at intersections. Intersections with a diameter of about 5 m have recently been realised for the Corbulo tunnel (COB, 2022) - the same outer diameter as the disposal tunnel.

Reinforced concrete segments have been used for the construction of the Westerschelde tunnel and unreinforced concrete segments for the URF in Mol, in order to minimize the use of metal. Steel reinforced rings were assembled and emplaced in advance for the construction of an intersection (PRACLAY) at the URF in Mol (EURIDICE, 2007; Van Marcke et al., 2013). In COPERA, reinforced concrete is proposed for the transport tunnel(s) since this would facilitate the manufacturing of crossings with disposal tunnels. The intersections that need to be constructed may require engineering designs that redistribute the stresses around the opening of the concrete lining. Pre-support can be applied by steel rings, as done in Mol, but this pre-support could already be present if reinforced concrete is used for the transport tunnel (Yuan et al., 2017). In OPERA, reinforced concrete was excluded due to the potential gas generation by anaerobic corrosion of the steel rebars, which might disturb the clay host rock in the post-closure phase. To assess whether the potential hydrogen flux gas generation rate from the reinforced concrete liner can be sufficiently dissipated by diffusion of hydrogen in clay pore water, it is necessary to calculate whether hydrogen solubility is exceeded. The hydrogen generation rate can be determined by the carbon steel bar corrosion rate and ratio of the surface of steel to the surface of concrete. The maximum carbon steel corrosion rate is 0.1 µm per year in alkaline media (Smart et al., 2017). The ratio with the surface of steel was 0.61 in a reinforced concrete segment (Leegwater et al., 2009). Hydrogen solubility in clay pore water increases with pore water pressure at increasing disposal depth. In a scoping study (Neeft and Grigaliuniene, 2017), the hydrogen concentration was calculated to be 26 mol H_2/m^3 assuming a ratio of surface steel of 1. This hydrogen concentration exceeds the solubility at a depth of 225 m of 18 mol H_2/m^3. The lower ratio of 0.61 would lead to a hydrogen concentration of 16 mol H_2/m^3 which would not exceed the hydrogen solubility. More detailed studies, in which the low water permeability of concrete is taken into account, are needed. For now, only the transport tunnel is assumed to be lined with reinforced concrete.

Figure 4-9: Salt deposits at joints of concrete segments in the Belgian URF (left) and Dutch traffic tunnels - Westerschelde (middle) with whitish deposits and Sluiskil (right) with joints between segments in the top, no salt deposits yet . All tunnels have been constructed in Boom (Rupel) Clay.

4.6 Operation

4.6.1 Ventilation

In the operational phase, the ventilation required to provide a safe environment for personnel is also expected to be sufficient to keep the GDF dry. Salts will be deposited on the intrados of the concrete liner, especially at joints between concrete segments, as can be seen clearly in the URF at Mol in Figure 4-9.

Only electric vehicles are envisaged for the transportation of waste packages in the underground facility in order to prevent exhaust emissions. Monitoring of the air at the shafts inside the underground part of the disposal facility will be performed by COVRA but also by an independent authorised organisation, for example the National Institute for Public Health and the Environment. This monitoring will also be required in order to detect any additional radiological exposure from emplaced waste packages.

The GDF contains a large amount of concrete in the packaged waste and engineered components, such as the tunnel liner and floor. Concrete contains traces of uranium and thorium from which gaseous radon can be emitted. The rate of radon emission increases with decreasing degree of saturation of concrete. Ventilation of the underground facility must be sufficient to prevent build-up of radon. Clay also contains traces of uranium and thorium, but radon emission is expected to be less than from concrete, as the clay remains saturated due to the low permeability of the concrete liner which prevents its dehydration.

4.6.2 Techniques for packaging waste

Currently at COVRA's premises, LILW is conditioned with concrete, while HLW is returned from reprocessing in welded stainless-steel canisters. Future LILW packaging and conditioning activities are considered to continue (see section 4.3.3) and to take place at COVRA's premises, since the equipment and expertise is available. The canister is welded in a carbon steel overpack, similarly as COVRA currently welds SRRF into stainless steel canisters. After this welding, the overpack is lifted by the mushroom into a concrete buffer. The void volume is filled with the same type of concrete as the buffer, only the size in aggregates is different in order to reach as much void space as possible. A visualisation of encapsulating HLW canisters into supercontainers is shown in Figure 4-10.

Figure 4-10: Encapsulation of vHLW in supercontainers.

The current storage facility for HLW needs to be extended to receive this device to pour cementitious fluid than can harder into concrete and reeive the concrete bufferes. The CSD-c canisters will be put into a concrete container into which concrete filler is poured, with additional concrete added above for shielding the canisters.

4.6.3 Techniques for emplacement of waste packages

The arrival of waste packages at storage and disposal facilities can be by truck, train or ship. As explained earlier, only electric trucks are considered within the disposal facility. Several techniques for emplacing waste packages have been demonstrated, especially in storage facilities. For example, forklift trucks and cranes are used to lift, transport and position waste packages at COVRA's storage facilities.

Cranes are used for emplacement of waste, e.g., for disposal vaults in Sweden (Vahlund and Andersson, 2015), where the width of the vaults is 14 to 20 m, which allows positioning of several waste packages next to one another. Current information, however, significantly restricts the width of disposal tunnels constructed in poorly indurated clay, so that emplacement of waste packages by crane is not considered.

Small forklift trucks such as those COVRA currently uses to emplace 200 litre drums and 1000 litre drums in the storage facility (see Figure 4-11) can also be used in the disposal facility. For other waste packages, other means need to be considered.

The heaviest waste packages are Konrad containers, at 20 tonnes, and the supercontainers for vitrified HLW, at 22 tonnes. These can be lifted with currently available electric forklift trucks, which can carry a load of 33 tonnes (e.g., Kalmar type). The large dimensions of these heavy-duty forklift trucks mean that they would not fit in a disposal tunnel with an inner diameter of 4.6 m or 4 m and could only function in the transport tunnels and in the surface facility.

Air cushion vehicles are foreseen as an appropriate alternative, and these are assumed in deriving COVRA's cost estimates. This technique has also been employed by COVRA (see Figure 4-11) to lift Konrad and other containers. Several models are available, with maximum capacities ranging from 20 to 400 tonnes (AeroGo, 2022). Consequently, the smallest model, with the following standard dimensions would be sufficient: width 1.8 m, length 4.3 m and height 0.432 m. The disadvantage of the use of these cushion vehicles is that stacking of containers is not possible.

The air cushion vehicles require a smoother, airtight and flat floor. Consequently, finishing of the floor in the disposal tunnels must use either epoxy or floated concrete. The use of floated concrete introduces fewer new materials that might affect the chemical evolution of the EBS and has therefore been chosen to reduce uncertainty in the post-closure safety assessment.

4.7 Closure

The closure activities of the GDF will involve the removal of ducts, lights, cables etc. and backfilling of the void volumes. In the disposal tunnels, backfilling with foamed (cellular) concrete is foreseen. The duct in each disposal tunnel can be used to position the pipes in which cementitious backfill can be injected. Formwork will be emplaced at intervals in order to backfill the disposal in stages.

Sections of the ducts are retrieved and removed from the disposal tunnels as backfilling proceeds towards the transport tunnel. Closure of the GDF is performed by emplacement of engineered materials with known characteristics.

The water permeability of closure materials in an important characteristic. European standard EN 12390-8 is usually used to investigate whether manufactured concrete is impermeable in engineering terms. COVRA's waste package concrete is frequently compared with this standard. The cemented plugs foreseen by Posiva to close the Finnish GDF must also meet this standard to assure the quality of the manufactured concrete (Vehmas et al., 2017).

The permeability of concrete is determined by the content and type of aggregates, water to cement ratio and other additives. Siliceous aggregates usually have such a small porosity that their contribution to the permeability of concrete can be neglected. The permeability is therefore constrained by the cement paste and the reaction layers between the cement paste and aggregates. COVRA's waste package concrete has been investigated in EURAD-1 WP MAGIC and in EURAD-1 WP ACED. As the aggregate content in the concrete buffer and concrete segments in the liner is similar to COVRA's waste package concrete, it is tentatively assumed that these concrete materials have the same low permeability. Foamed concrete with the same content and type of cement, but with a lower content of aggregates, was also investigated. Due to its lower aggregate content, foamed concrete has a higher permeability than COVRA's waste package concrete.

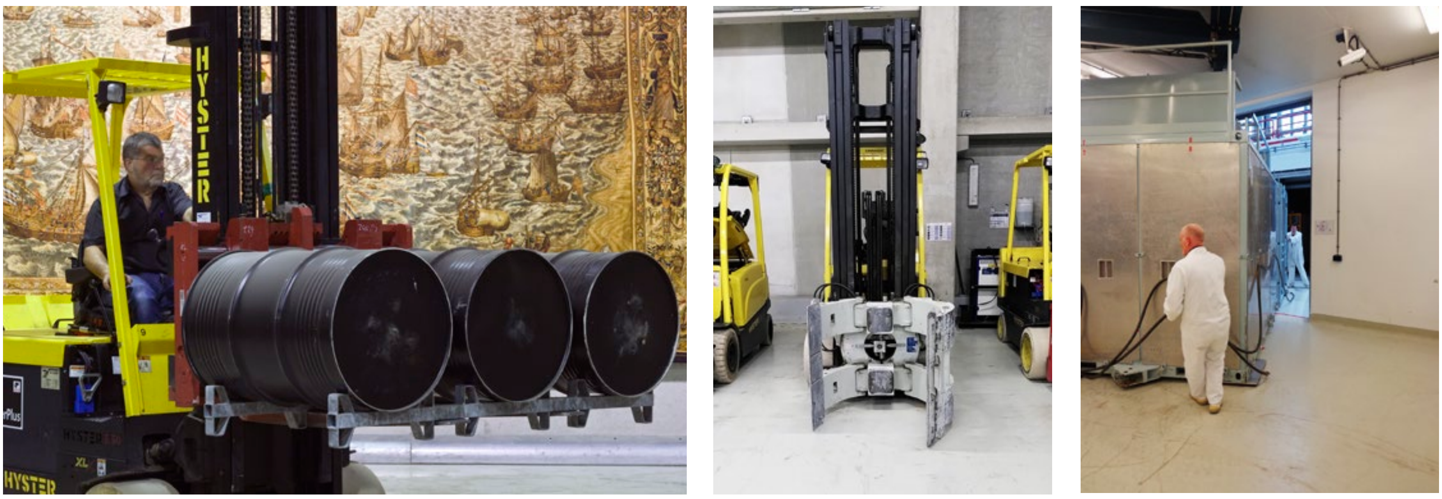
In the post-closure phase, hydraulic gradients between the clay host rock and the porous engineered materials will eventually disappear and diffusion will be the dominant mechanism for water and solute movement. More detailed information about diffusion of water within concrete has become available since OPERA. Table 6-1 shows that the diffusion values of water within cementitious materials are smaller than within clay.

The saturated permeability and diffusion values are the maximum values. The permeability value of 9×10^{-20} m^2 that was determined previously for COVRA's waste package concrete (Verhoef et al., 2014) is quite close to the recently determined value of 7.3×10^{-20} m^2 given in Table 6-1, which was obtained by a comparison between the predicted weight of water in the sample and the measured weight during drying. The diffusion values have a similar order of magnitude: i.e., 1.4×10^{-11} m^2/s (Verhoef et al., 2014) and 0.8×10^{-11} m^2/s in Table 6-1.

Foamed concrete is a tailor-made material and the permeability property is expressed 5 kg of water that penetrated 1 square meter in 10 years (CUR, 1995). The calculated permeability obtained from the measured hydraulic conductivity of 1.6×10^{-11} m/s[12] (Verhoef et al., 2014), then becomes 1.6×10^{-18} m^2 (which is close to the 2.2×10^{-18} m^2 given in Table 6-1. A diffusion value was not derived in OPERA. Foamed concrete is different from waste package concrete in its pore characteristics. In waste package concrete, only gel pores and capillary pores are present. Foamed concrete also contains other pores with a negligible capillary force, which has a high influence on the characteristic radius of the pores. However, the water diffusion value through capillary pores has been found to be only twice that of waste package concrete (Blanc et al., 2024). It is therefore tentatively assumed that the backfilling of disposal tunnels using with foamed concrete is adequate for the post-closure phase.

For the transport tunnels, backfilling with excavated clay is foreseen. Backfilling and compacting are demonstrated techniques in civil engineering. Isostatic compaction of clay (bentonite) in blocks has been investigated in detail for GDFs. The Canadian programme demonstrated the manufacturing of large compacted blocks with

12. 0.5 kg water per m^2 per year provides with a density of 1000 kg per m^3 0.5×10^{-3} m per year which is 1.6×10^{-11} m per second.



Forklift trucks for emplacement of three 200 l drums of one 1000 l concrete container

Air cushion vehicle for heavier objects

Figure 4-11: The different techniques considered to emplace waste packages (left and middle) forklift trucks, air cushion vehicle (right).

a weight of about 4 tonnes (Crowe et al., 2016). Compacted blocks have a mixture of clay and sand, to control the swelling pressure once the clay absorbs water. Emplacement of compacted blocks is proposed in the Swedish and Finnish programmes for backfilling transport tunnels constructed in crystalline rock. A similar technique is proposed for the clay GDF investigated in the current COPERA safety case, except that the blocks are fabricated using excavated, poorly indurated clay which is a natural mixture of clay with sand.

The excavated clay can be dried, and some grinding may be necessary in order to obtain a fine-grained powder, which can be isostatically compressed in order to manufacture compacted blocks. The so-called reconstituted clay may need to be mixed with some cement in order to reduce the pH decrease caused by oxidation of pyrite, which is present at about 2 wt% or less in the Paleogene clay formations (Neeft et al., 2019). The acid neutralizing capacity of cement is high, so adding 2 wt% cement to the excavated clay is therefore envisaged to be sufficient. The diffusion values for reconstituted clay are smaller than the natural, in-situ clay, due to loss of the sedimentary structure. The preferred migration route for radionuclides, if released from the waste form and cementitious containment, will therefore be into the natural clay host rock, at the end of the disposal tunnel or between the joints of concrete segments (see Figure 4-12).

Box 4-1 Uncertainties under investigation and to be investigated

Significant uncertainty in the geomechanical properties of Boom Clay was noted in the third national programme OPERA (Arnold et al., 2015). This uncertainty was attributed to both the scarcity of high-quality data, as well as the variability of the geological, geochemical and geomechanical host rock properties. COVRA's recent research programme has therefore been looking for relevant existing boreholes (Verhoef et al., 2020) and has also supported the DAPWELL borehole project near Delft University of Technology (TU-Delft), since this was expected to provide sufficient Paleogene clay core material for analysis (Abels and Vardon, 2020; Munsterman, 2020). In 2022, cores were extracted from about 400 m depth (Vardon et al., 2022). Geotechnical analysis on these cores started in 2023, through the research project SECUUR, which is mainly funded by the Netherlands Organization for Scientific Research and is executed by TU-Delft.

Knowledge of geotechnical properties of clay material obtained from representative depths would aid in defining design requirements and would significantly contribute to confidence in the optimisation of the layout of the underground facility at various potential disposal depths.

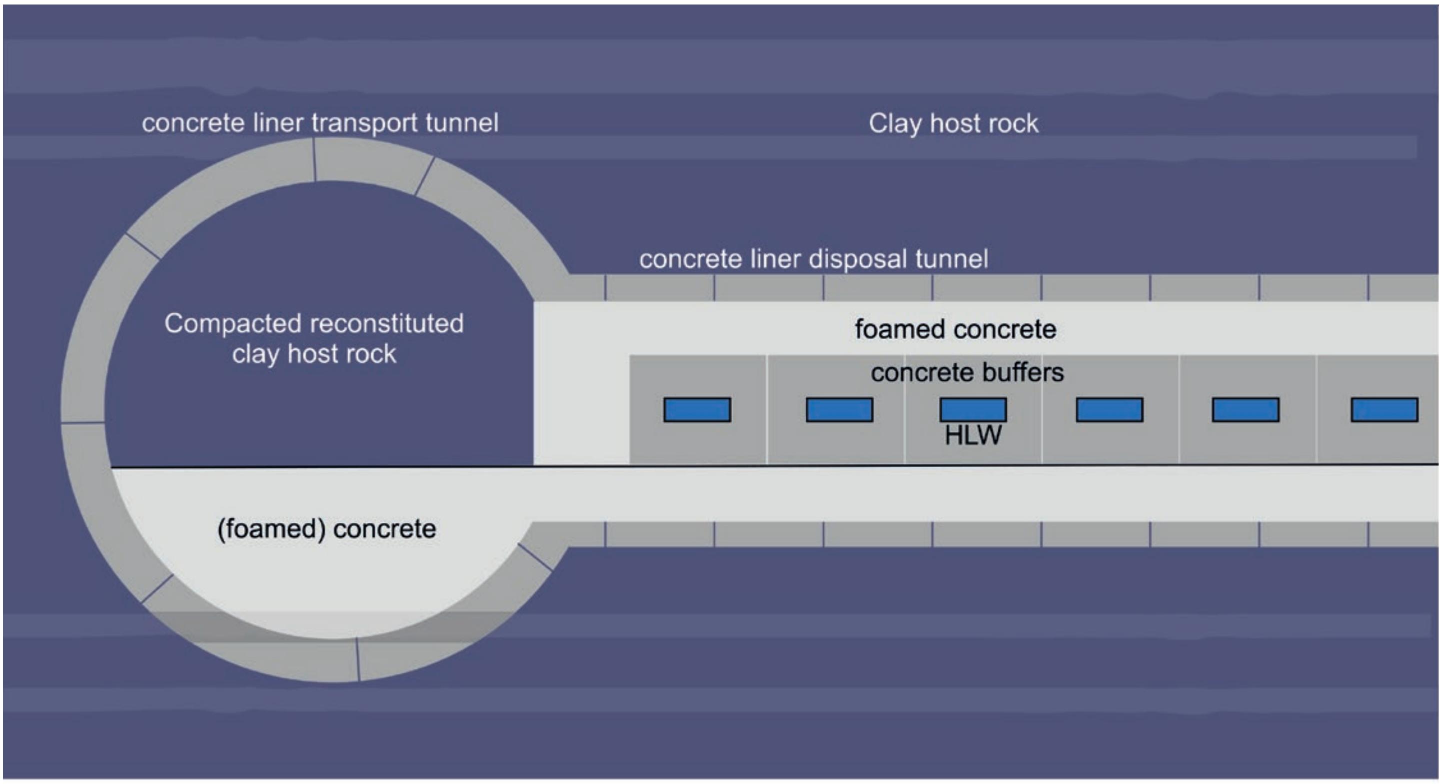


Figure 4-12: Impression of a cross section of a disposal tunnel and transport tunnel with the upper part of the (outer diameter 10 m) backfilled with compacted excavated clay

Box 4-2 How much will the GDF cost?

In accepting wastes, COVRA charges a contractual tariff to cover all the phases of radioactive waste management, with the objective of accumulating sufficient financial resources for the construction of a GDF and its subsequent operation, which starts in 2130. In COPERA, probabilistic cost estimates have been made. The best estimate of the overnight costs for the construction, operation and closure of a GDF are estimated to be 3.0 billion euros Euros, at a price level of 2022, if the disposal facility is constructed at a single depth, as assumed in OPERA. Overnight means that activities are assumed to have taken place in a single year for the cost estimate.

For the same waste inventory (Waste Scenario 1), these average costs would be 2.9 billion euros with a price level of 2022[13] using a constant yearly inflation rate of 2%. There is therefore little impact on total estimated costs using the updated COPERA estimates, despite a significant reduction in the disposal volume for non-heat generating HLW (CSD-c). There are several reasons for this, of which the most important is the updated and more realistic engineering concept, which has involved:

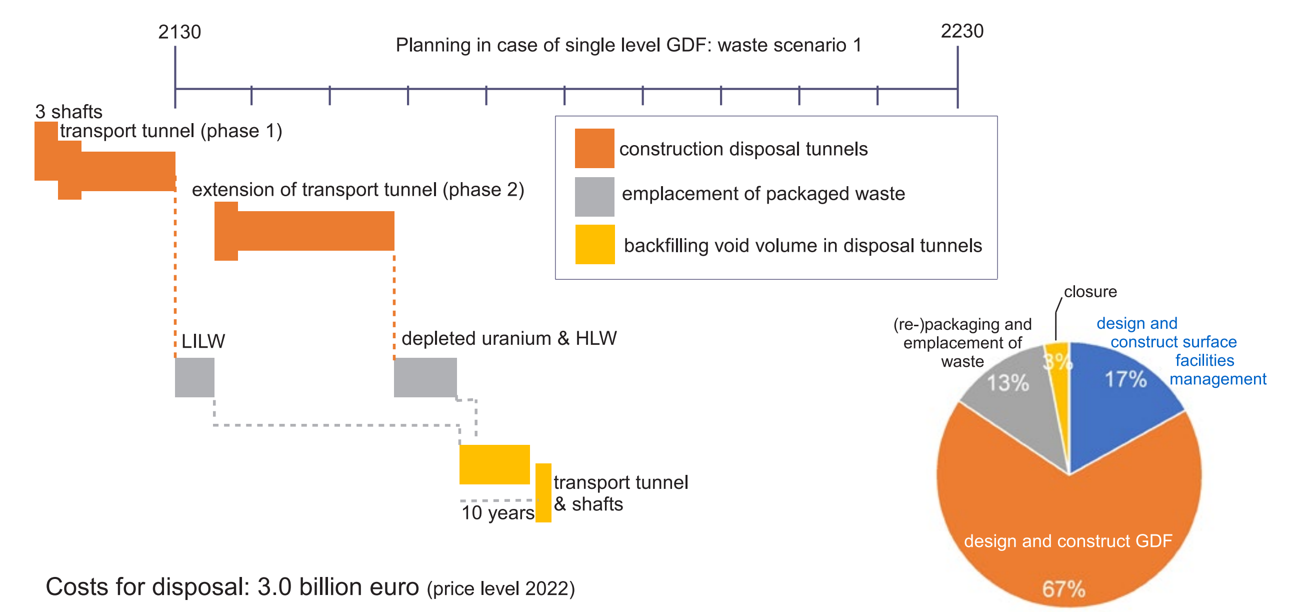
- removal of the excavated clay to the surface via shaft, rather than through an inclined ramp (see section 4.2);
- increasing the dimensions of the transport tunnel for constructional feasibility and to accommodate the envisaged outer diameters of the disposal tunnel (see section 4.5.3);
- increasing the manoeuvring space to emplace waste packages with a vehicle, which is now constrained by the dimensions of the fork lift truck envisaged to emplace a waste package (see section 4.4.1 and 4.6.3).

The SSK calculation methodology has been used to make the cost estimate. SSK is the Dutch acronym Standaard Systematiek Kostenramingen (SSK) for a calculational model in which the cost estimate is standardized through an agreed system. In this system, the costs are subdivided into subjects related to the activities. Each subject has an overview of the following four categories: construction costs, engineering costs, real estate costs and other costs. Each of these categories has both investment costs and maintenance costs. The SSK methodology is frequently used by the government and businesses in the Netherlands, and COVRA was advised to use it after an audit of the cost estimate in 2014 (Bruinsma and Tempels, 2017). The cost categories used in the current estimate are:

- Surface facilities, or above ground activities related to construction and maintenance of surface facilities;
- Underground activities related to construction and maintenance of the underground facility;
- Operational activities related to the handling of waste;
- Closure activities that ultimately lead to a so-called green-field end state.

The total overnight costs for a multi-level GDF with disposal at three different levels are estimated to be 2.7 billion Euros, which is less than the costs of the single level GDF discussed above. The main reason for these lower costs is that a larger disposal volume per unit length of disposal tunnel is possible due to greater tunnel diameters when a smaller thickness of concrete liner is required at smaller depths.

13. the cost estimate in OPERA was estimated to be 2.1 billion Euros, at a price level of 2017 (Verhoef et al., 2017) using a constant yearly discount rate of 2.3%. Without discount, the average costs of the GDF was estimated to be 2.61 billion euros with a price level of 2017.



Costs for disposal: 3.0 billion euro (price level 2022)

Also, the length of transport tunnels can be reduced, due to a smaller assumed distance between disposal tunnels at a shallower disposal depth.

The cost breakdown in the figure on the previous pageshows the planning with a focus on the construction, operation and closure of the GDF. The construction of the surface facilities and underground facility accounts for almost 70% of the total cost. This percentage is similar to OPERA, in which design and construction costs accounted for 70% of the costs.

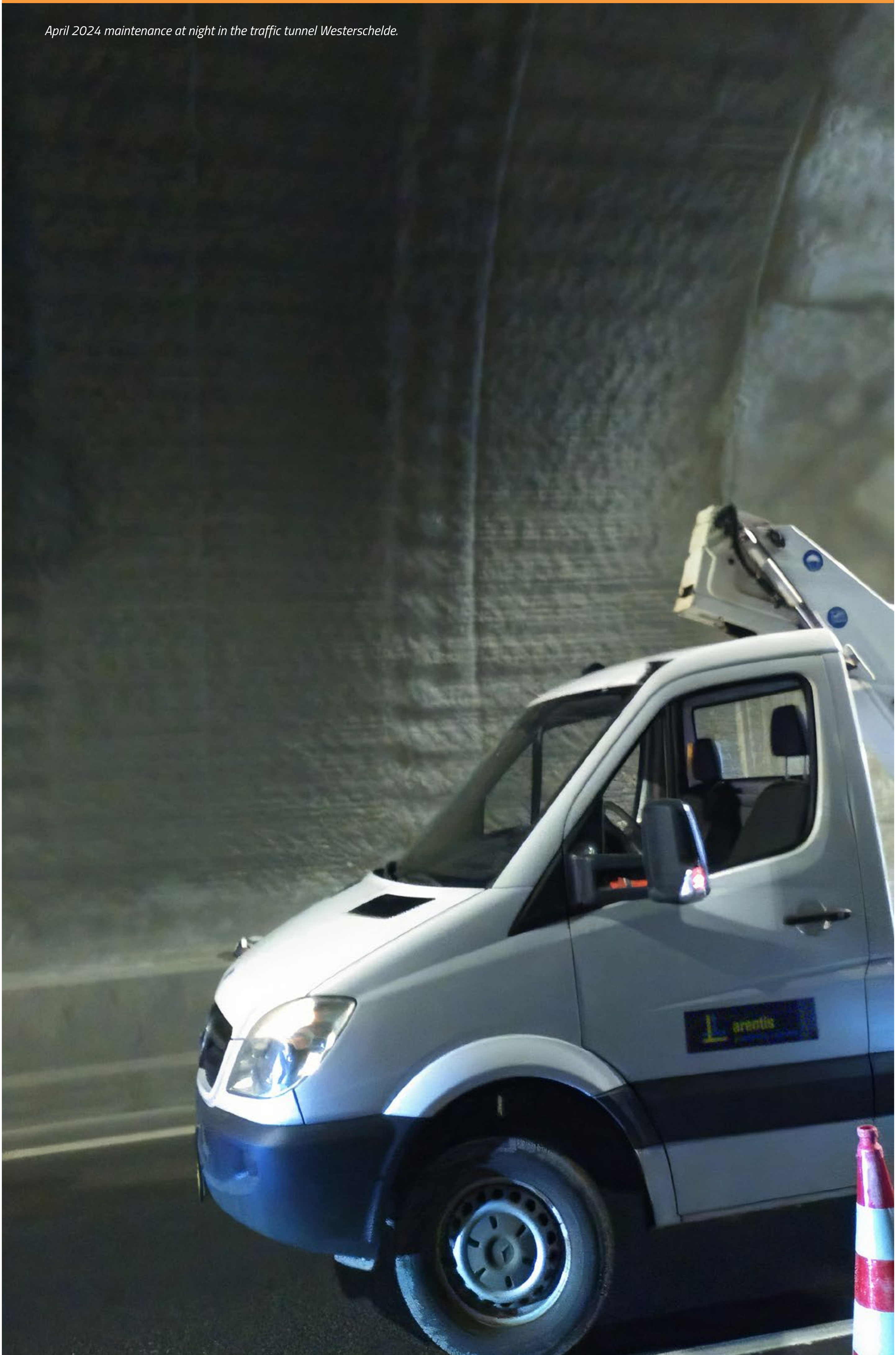
The previous cost estimate was based on Belgian prices for components and COVRA's salaries for personnel, indexed for 2017. The auditors of this cost estimate also advised us to make local estimates of important cost items and to use unit base data from Dutch sources (Bruinsma and Tempels, 2017). The sources from which the costs of manufacturing of components have been extracted are preferably costs determined by the Dutch Association of Cost Engineers (DACE). Almost every year, DACE publishes a price booklet with an upper and lower level in the costs of each item. 2022 was an extreme year in terms of inflation, but there is no price booklet with the publication year 2022. DACE (2023) was used for the current cost estimate, after an audit of a cost estimate in which DACE (2021) was used (Tempels et al., 2023).

Costs provided by DACE do not allow for price discounts. Discounts are common, especially if large quantities are used. COVRA's experience is that the costs indicated in the price booklet are beyond the maximum of price bids. Other key figures for cost items were obtained from the Dutch website for underground infrastructure Centrum Ondergronds Bouwen (COB). If a component could not be found at COB either, other websites were consulted and recorded for traceability. If no open accessible source was available, the following sources were used: information as supplied by the Belgian WMO ONDRAF/NIRAS in 2014 indexed for 2022, costs available at COVRA from purchases in the past and costs from companies that had previously supplied COVRA and which were contacted in 2022 in order to obtain costs. COVRA's salaries for personnel, indexed for 2022, were used for the handling and transportation of waste packages under radiation protection control, engineering, research, maintenance, management, communication, security and administration.

Cost optimisation of the GDF by using low level waste as construction materials has not been included, as that may reduce the retrievability of waste.

Costs of the disposal facility in clay host rock - Price level 2022 - to be published at the same time as this safety case

April 2024 maintenance at night in the traffic tunnel Westerschelde.



March 2022 deep drilling for extraction of Paleogene Clay cores o.v.v. Courtesy TU Delft

How nature can isolate the waste from people and the environment for at least a million years

5. The Natural Barrier System

This Chapter looks in more detail at the geological environment in which the GDF is to be constructed. The host rock for the GDF and the surrounding geological formations together comprise the natural barrier system within the multibarrier system that was introduced in Chapter 2.

Argillaceous (clay-rich) formations are being considered as potential host rocks for disposal of radioactive waste in numerous countries including France, Switzerland, Belgium, Canada, Germany and the UK (Boisson, 2005). The clays studied in our work in the Netherlands and in Belgium are described as poorly indurated, meaning that they have not been highly compacted and hardened by deep burial, and consequently have lower mechanical strength than the claystones and mudstones being considered in most other countries. However, all clay formations have the following favourable attributes that contribute to the containment of radionuclides and to minimising the transfer of radionuclides to the surrounding rock formations:

- significant lateral continuity;
- low permeability, meaning that transport of radionuclides in clay formations is by slow diffusion in pore waters rather than by more rapid advection in flowing groundwater;
- self-sealing (in indurated clays) or self-healing (in poorly indurated clays) of fractures;
- capacity to chemically contain or retard radionuclide migration.

Chemical containment or retardation of radionuclides results from the limited solubility of redox sensitive radionuclides (such as uranium, selenium and plutonium) under the chemically reducing conditions found in deep clay formations, from ion exchange of radionuclides dissolved as cations (e.g., caesium), and from complexation of radionuclides with dissolved organic matter. Such complexes are too large to pass through the connected porosity of clays, as the pore 'throats' are narrow. So called 'self-analogue' studies that study how dissolved neutral species and species dissolved as anions have moved through clay formations over geological time show that transport processes are extremely slow: e.g., Mazurek et al. (2009), Mazurek et al. (2011), Rufer et al. (2024). The first steps in evaluating a self-analogue for dissolved cations and dissolved cationic complexes in Dutch Paleogene clays have been taken in this safety case.

Section 5.1 describes in detail the clay formations in the Netherlands that are considered as potential host rocks for the GDF. The geological formations that surround these clay formations are described in section 5.2. Although the waste is isolated from major changes in Earth's surface environment, including those resulting from climate change, understanding the impact of these changes on the properties and behaviour of natural barriers is important. Section 5.2 introduces these dynamic processes, discusses their potential impacts and examines the measures that can be taken in GDF design and eventual siting to minimize them. As with the OPERA programme, there is still little direct information from drill-cores on the characteristics of the Dutch clay formations at suitable disposal depth, so that significant uncertainties remain, and these are also discussed in this Chapter.

5.1 Paleogene clays as the preferred host rocks

Neogene and Paleogene clay formations were considered as clay host rocks in the second national programme, CORA (e.g., Simmelink et al., 1996). One of the Paleogene clay formations was the basis for the third national programme, OPERA (Verhoef et al., 2017). This is the Boom Member of the Rupel formation that is informally known as Boom clay (Vis et al., 2016). It was selected for OPERA in order to use information and learn from experience in the mature Belgian programme, which has identified the Boom clay as a preferred host rock for its national GDF project (Verhoef and Schröder, 2011). There are, however, additional options to Boom clay among the Paleogene clays of the Netherlands.

Clay formations are considered aquitards in groundwater management. Fresh groundwater can be extracted from sandy formations that are overlain and underlain by clay formations. In many parts of the Netherlands the hydrological base for groundwater extraction is the Maassluis formation, which is of early Pleistocene, c.2.5 Ma age, (Dufour, 2000). A clay host rock confined by sandy formations containing non-potable saline waters is preferred for disposal of waste, in order to minimize interactions with groundwater management activities. The limited data available on the much older, deep Oligocene (i.e., late Paleogene, including the Boom clay at about 28 - 34 Ma) aquifers shows them to be saline to brackish (Griffioen, 2015), so of no interest as sources of potable water supply.
All the Paleogene clay formations are therefore being considered in COVRA's current COPERA research programme (Verhoef et al., 2020).

5.1.1 Marine sedimentation of Paleogene clays

The Paleogene clays were sedimented as marine deposits on the seafloor over the period from c.23 million to c.66 million years ago. The Paleogene period comprises three epochs (Knox et al., 2010):

- Oligocene (23.03 to 33.9 Ma, divided into the Rupelian and Chattian ages), including the Rupel, Watervliet and Veldhoven clays;
- Eocene (33.9 to 56 Ma, divided into the Priaborian/Bartonian, Lutetian and Ypresian ages), including the Asse and Ypresian clays;
- Paleocene (56 to 66 Ma, divided into the Thanetian, Selandian and Danian ages), including the Landen and Gelinden clays; the Danian stage comprises a Chalk Group with marine chalk or limestone facies.

The sea level in the Paleogene period was 150 to 200 m higher than today (van der Meer et al., 2022) and the area of the present-day Netherlands was submerged, with marine sand and mud being sedimented. Figure 5-1 shows the paleogeography in the Rupelian age (28.1-22.9 Ma) and the Ypresian age (47.8 - 56 Ma), during which these marine clays were deposited (Knox et al., 2010; Vis and Verweij, 2014). In fact, the deposition of marine clays took place throughout the entire Paleogene period in the current area of the Netherland. During COPERA, Ypresian cores have been extracted at about 400 m depth.

The present-day depth of each Paleogene clay formation has been controlled by the progressive subsidence of the adjacent North Sea Basin over many millions of years, with the more deeply located formations occurring in the northwest of the Netherlands and shallower formations occurring at the edges of this basin, i.e., in the central eastern and the southwestern parts of the Netherlands.

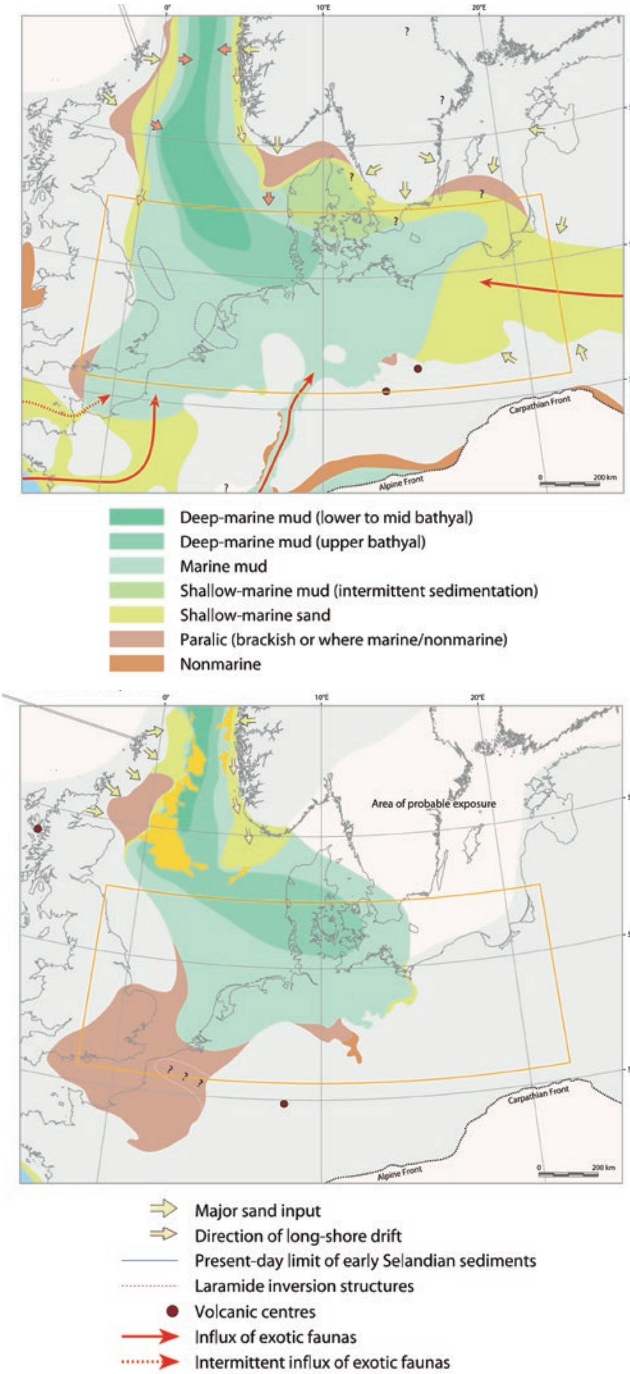


Figure 5-1: Paleogeography during the early Oligocene (left, Rupel: 31 million years ago) and the earliest Eocene (right, Ypresian: 56.5 million years ago) (Knox et al., 2010)

Paleogene clays are absent in the southern part of the Province of Limburg (Duin et al., 2006; Simmelink et al., 1996).

5.1.2 The present day form of the Paleogene clays

The past evolution of Paleogene clays in the Netherlands has been characterised by progressive burial during continued sedimentation. Figure 5-2 shows the regional variation in burial history of the Paleogene sediments, with a focus on the Rupel clay member (denoted as NMRFC) in the northern and central part of the Netherlands, as computed by basin modelling (Verweij et al., 2016) in which Paleogene sediments belong to the stratigraphic Middle North Sea Group (NM) and Lower North Sea Group (NL).

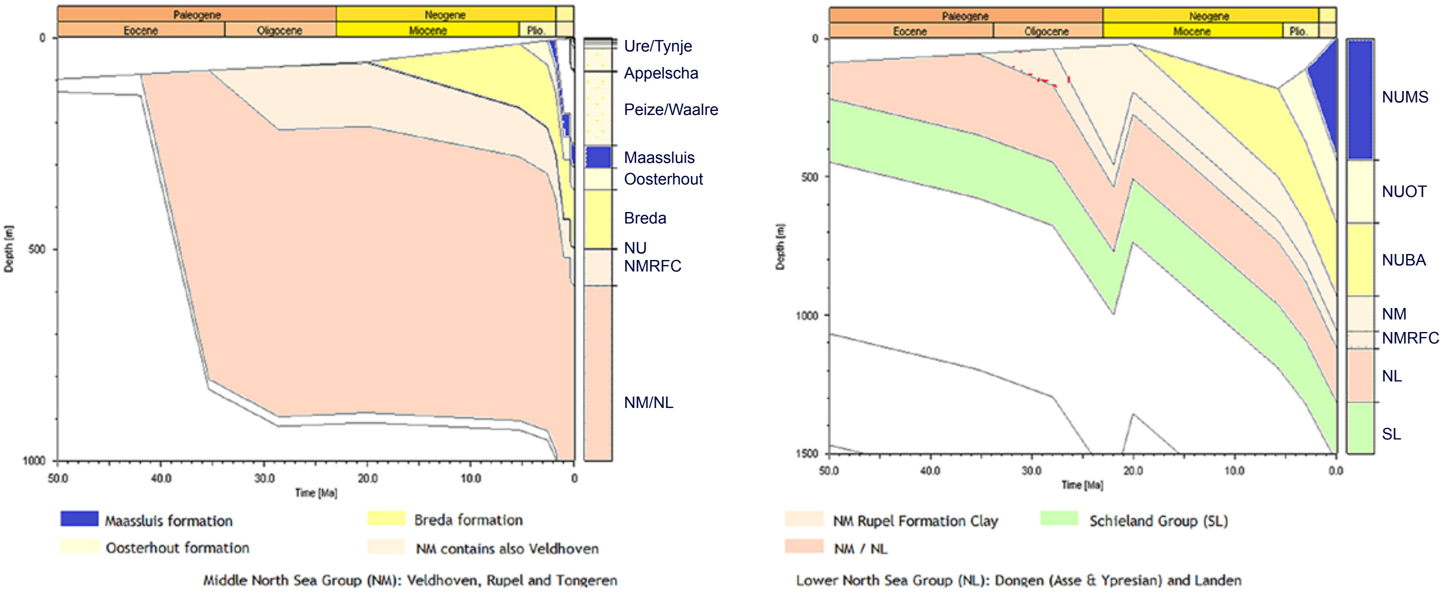


Figure 5-2: Example of regional variation in burial history of the Rupel Clay Member (NMRFC): 1D extraction from 3D basin models in the northern (left) and central (right) part of the Netherlands, adapted from Verweij et al. (2016).

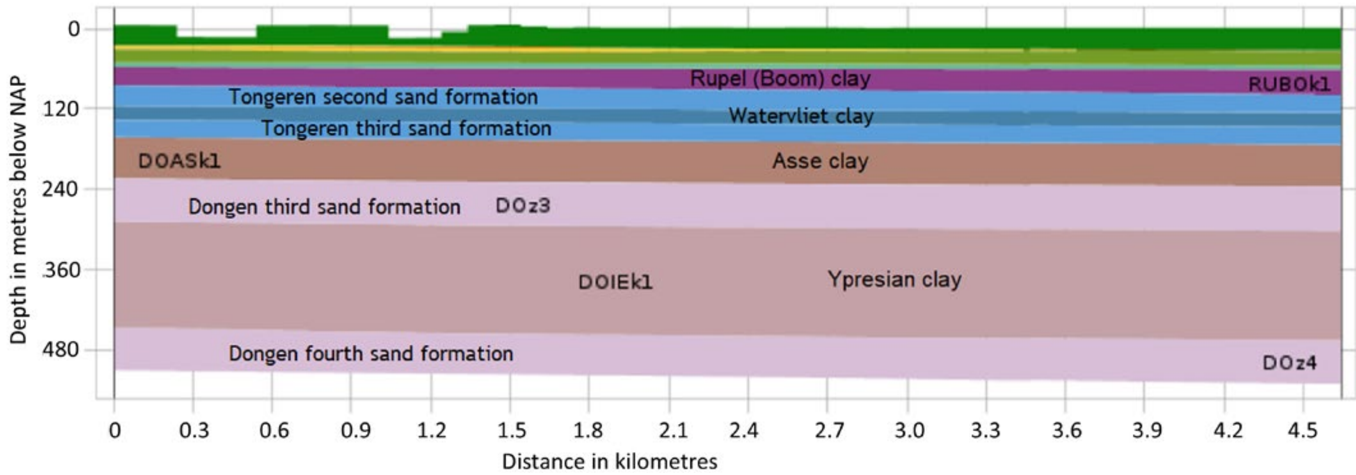


Figure 5-3: Example of a vertical section at a location in the Netherlands with several Paleogene clay formations present. From DINOloket, obtained in 2021, model REGIS with numbered sand formations; NAP=Normaal Amsterdams Peil is the Dutch reference level for heights and depths, and is about equal to the average sea level of the North Sea; kl is the Dutch abbreviation for clay, z is the Dutch abbreviation for sand.

Figure 5-2 shows an increase in burial rate during the Quaternary, to about 50 m per million years. The figure also shows the separation between the hydrological base of the groundwater management (Maassluis) formation and the Paleogene formations; the formations in the Upper North Sea Group (Breda and Oosterhout) lie between these formations.

5.1.3 Thickness and depth

The containment potential of a clay formation as a GDF host rock is determined by its thickness and the extent of favourable properties (see section 5.1.7). The depth and lateral extent (see section 5.2.1) determine the isolation potential. There are usually several Paleogene clay formations present beneath locations in the Netherlands: Figure 5-3 shows an example.

The Paleogene clay formations have not yet been documented at the level of detail in Dinoloket for all places in the Netherlands: only the North Sea Groups listed in Figure 5-2 are available. Figure 5-4 shows an example of a vertical section across the Netherlands. The Paleogene sediments are denoted as NL_NM.

The data in DINOLoket have frequently been obtained from boreholes drilled for exploration for oil and gas, many of which have penetrated Paleogene clays. Because the exploration targets have been much deeper formations, high quality logs of Paleogene clays such as the Boom clay are usually not obtained by the oil and gas industry (Vis et al., 2016). The thickness of any Paleogene clay formation in the database should therefore be treated as indicative. Figure 5-5 shows the top and thickness of the Paleogene sediments in the Netherlands geographical area. The top of the Paleogene

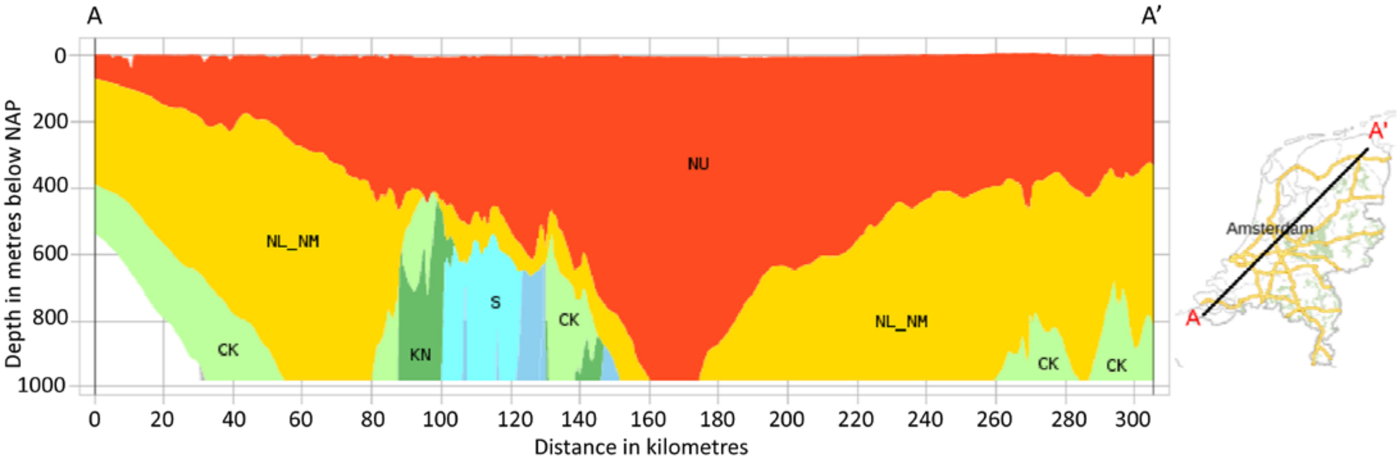


Figure 5-4: Example of a vertical line of section across the Netherlands from DINOloket, obtained in 2024, model DGM with groups of formations: NU: Upper North Sea Group, to which Neogene sediments (clay and sand) belong, NL_NM: Middle and Lower North Sea Group, to which the Paleogene sediments (clay and sand) belong, CK: Late-Cretaceous Limestones, KN: Early-Cretaceous group, S: Late-Jurassic - Early Cretaceous Group.

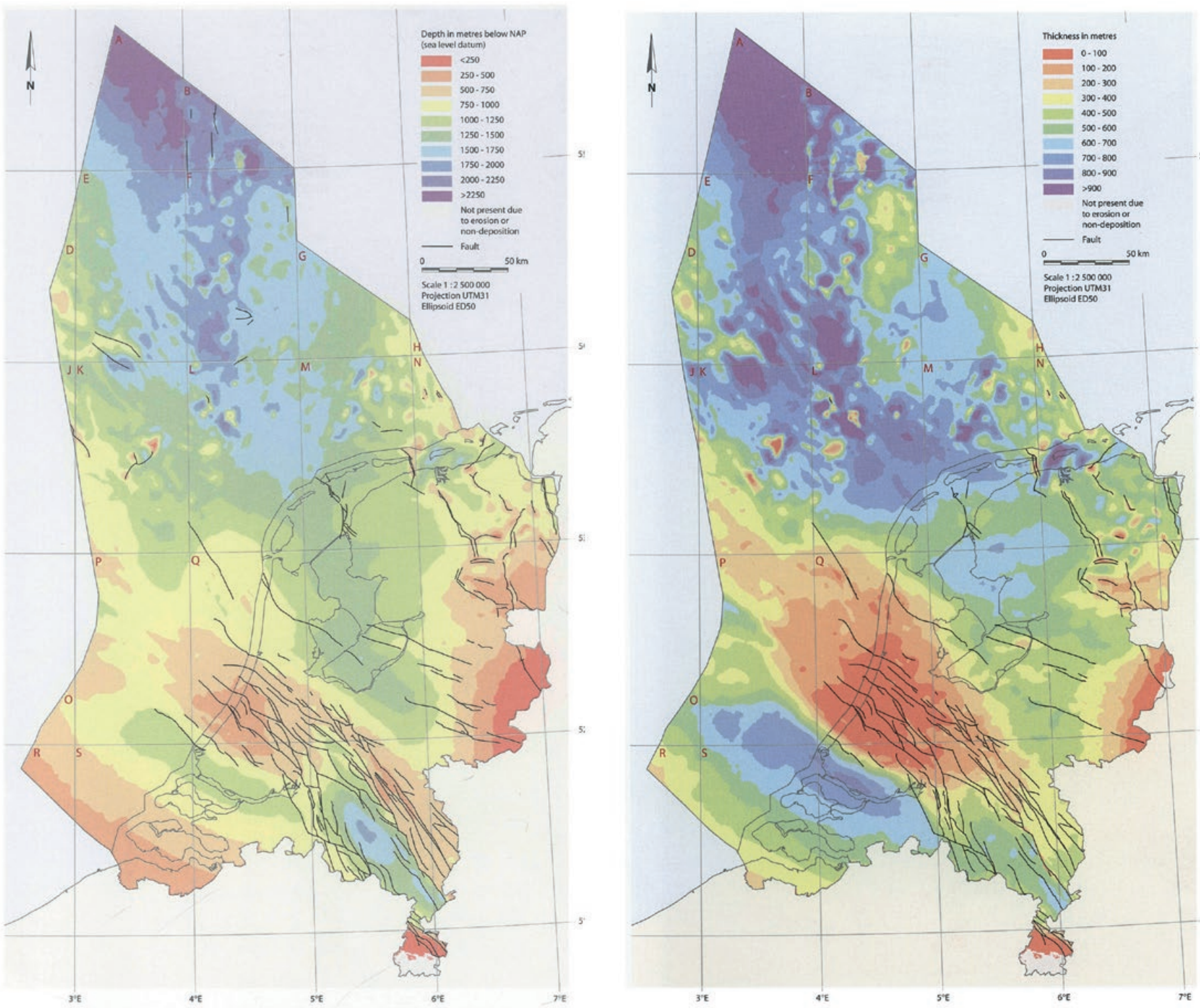


Figure 5-5: Depth to the top (left) and thickness (right) of the Paleogene sediments (Duin et al., 2006), mainly determined using data provided by the oil and gas industry.

sediments is usually deeper than 250 m, with a thickness of more than 200 m in most areas of the Netherlands, implying a wide-spread potential for useable clay host rock.

A tectonic structure in the southwest of the Netherlands (the Roer Valley Graben) displaces the Paleogene clay formations to greater depths than in the northwest of the country. In addition, tectonic uplift during the Oligocene period exposed some Paleogene sediments to wind erosion at the surface, resulting in areas in the Netherlands where one or more Paleogene clay formations are absent.

5.1.4 Clay cores for research

Until a URF in a clay host rock in the Netherlands is available, borehole cores are the only direct source of information on local Paleogene clay characteristics, but only limited geotechnical research has been performed on these to date:

- In 1998, Boom clay was extracted in Blija, from between 450 and 480 m depth (Barnichon et al., 2000). Only reconstituted clay could be investigated as the clay cores were too mechanically disturbed by the drilling technique used.
- In 2011, Boom clay was extracted in Borsele, from 70 to 75 m depth, Watervliet clay from 90 to 100 m depth and Asse clay from 140 to 200 m depth (PCR, 2013). A push-coring technique was used to obtain the clay cores. This drilling technique generated a sufficient quality of core to perform proper investigations on undisturbed clay material, including clay pore water characterisation and determination of geotechnical properties. All clay cores are also encapsulated in PVC liners with paraffin wax on the top and bottom of each core. The Boom clay cores have been studied in OPERA (e.g., Behrends et al. (2015).
- In 2022, Paleogene clay cores were extracted from the DAPWELL borehole in Delft at about 400 m depth, using the same push-coring technique as used in Borsele (Vardon et al., 2022). This location is within the area indicated in red in Figure 5-5, with a clay thickness of less than 100 m. COVRA participated in the DAPWELL drilling in order to evaluate techniques for finding sufficient clay material suitable for research into geological disposal of waste. TNO maintains a core store with cores and cuttings from drilling, most of which was performed in the previous century. Palynological analysis of cuttings obtained in Delft that had been identified as Paleogene was performed as part of the COPERA project. However, palynological analysis on deep samples taken during the 2022 drilling work indicates that about half of the cored samples are actually from the Neogene period (Munsterman, 2023). All clay cores are also encapsulated in PVC liners with paraffin wax on the top and bottom of each core. Additional sealing was made with resin on top and bottom of each core and wrapping the whole core in aluminium foil. The cores are stored at 4 °C.

COVRA welcomes any cooperation with organisations planning push-coring boreholes that will intersect Paleogene sediments, and is willing to support some of the drilling work if the cored clays can be used to contribute to our knowledge base. However, COVRA's participation in drilling projects needs to be acceptable to the organisation carrying out the drilling, and needs to be openly and transparently acknowledged. This is considered essential for societal acceptance of COVRA's research. Such arrangements are expected to be an important aspect of our future work, as it is not expected that there will be any drilling at potential GDF sites, as part of the future siting programme, until after 2050.

5.1.5 Pore water composition

The evolution of the EBS is determined by the ingress of dissolved species from the clay pore water, or egress of dissolved species from engineered porous media, currently planned to be cementitious materials. Concrete pore water composition is controlled by the cement minerals present and any leaching of dissolved calcium from concrete will be controlled by the composition of clay pore water. The potential for leaching is important, because it increases the size of pores, increasing the permeability and reducing the mechanical strength of the concrete in the EBS.

Permeability, self-healing of fractures and other properties of poorly indurated clay may also depend on the pore water composition.

So far, there are no measurements available of Dutch Paleogene clay pore water compositions at suitable disposal depths for a GDF. The limited data available on Paleogene aquifers surrounding the clay formations of interest show them to be more saline than brackish (Griffioen et al., 2016). The only data on Paleogene (Rupel) clay pore waters are from much shallower depths and these have a salinity similar to seawater (Behrends et al., 2016; Behrends et al., 2015).

Measured pore water compositions for clay host rocks are influenced by experimental artefacts (e.g., De Craen et al. (2004) and Gaucher et al. (2009):

- the partial pressure of CO_2 is larger at disposal depth than at the surface and degassing of CO_2 takes place when core samples are taken, which has an impact on the measured bicarbonate (HCO_3^-) content and the pH of the pore water;
- redox potentials cannot be measured with sufficient accuracy, due to the introduction of oxygen during the installation of the measurement device; these potentials are therefore usually calculated based on thermodynamic equilibria;
- the clay host rock in the deep underground has usually been depleted in oxygen for millions of years. Samples of host rock can be sensitive to the oxidizing conditions when brought to the surface, e.g., pyrite (FeS_2) can be oxidized, acidifying the sample and thus leading to unrealistically high measured concentrations of sulphate (SO_4^{2-}) and cation concentrations, as a result of dissolution of carbonates. The carbonates buffer the decreasing pH.

Data obtained from clay core samples, for example by mechanical squeezing or aqueous leaching, can be used as input for thermodynamic modelling, which was used to estimate the pore water compositions in Table 5-1 for brackish, seawater salinity and very saline (>seawater) compositions, using the mineralogy as measured in Boom clay.

Parameter	Unit	Brackish	Sea	Very saline
Temperature	$-\log(H^+)$	26	26	26
pH	$-\log(e^-)$	7.0	6.9	6.5
pe	log(bar)	-2.9	-2.8	-2.4
pCO_2	mmol/kg	-1.5	-1.5	-1.5
Na^+	mmol/kg	114.4	460.9	1897.0
K^+	mmol/kg	2.4	9.8	40.4
Ca^{2+}	mmol/kg	5.7	13.2	44.0
Mg^{2+}	mmol/kg	13.9	56.1	230.8
Fe^{2+}	mmol/kg	0.0000027	0.0000031	0.0000035
Al^{3+}	mmol/kg	0.000041	0.000033	0.000021
SiO_2 (aq)	mmol/kg	0.3	0.3	0.2
Cl^-	mmol/kg	134.3	541.0	2227.0
SO_4^{2-}	mmol/kg	7.0	28.4	116.8
HCO_3^-		7.4	7.2	5.6

Table 5-1: Estimated compositions of Paleogene (Rupel) clay formation waters (Griffioen et al., 2017)

Trace elements such as caesium, strontium and uranium are also present in clay host rock pore water and their speciation and concentration can be indicative of their solubility controls.
The trace elements have, however, generally not been measured. An exception is the measurement of trace elements from Boom clay pore water from Belgium (De Craen et al., 2004). The concentration of a trace element in the solid or immobile phases of the clay formation (including clay minerals and dissolved organic carbon) divided by the concentration of that trace element dissolved in the clay pore water provides representative in-situ distribution factors (K_d, see 5.1.6.4.1) to support the modelling of radionuclide solubilities and transport in the clay host rock. For example, the average uranium content has been measured to be 3 mg/kg in Boom Clay (Koenen and Griffioen, 2014) while the uranium content in clay pore water has been measured to be between 0.2 µg per litre to 3.5 µg per litre (De Craen et al., 2004). The 1000 times or more larger concentration in the solid or immobile part of the clay compared to the clay pore water solution leads to a K_d value of more than 1000 L/kg and indicates how well uranium would be contained by a Paleogene clay formations, as only the dissolved radionuclides can leave the clay host rock. More information will be gathered using pore waters from clay samples obtained from the DAPWELL drilling project. This work will be carried out by Utrecht University and funded by COVRA. The major dissolved components of pore water will also be characterised in COPERA(2020-2025) and in the research project SECUUR, in order to understand the impact of the salinity of clay pore water on the mechanical properties of the clay rock (see Text Box 4-1).

5.1.6 Favourable containment properties of clay formations

The favourable properties of clay formations were outlined at the start of this Chapter: significant lateral continuity, low permeability, self-healing of fractures and capacity to chemically retard radionuclide migration. This section describes the available quantitative knowledge on these favourable attributes for the Paleogene clay formations.

5.1.6.1 Significant lateral continuity

Due to the laminar nature of clay minerals, which are generally sedimented parallel to the bedding plane, and the considerable lateral continuity of clay formations, any advective movement of water or solutes tends to take long pathways along the bedding planes of the sediment, rather than vertically into the surrounding formations. Migration behaviour is thus related to the anisotropic texture and fabric of the clay formations, about which little is known for the Dutch Paleogene clays. This knowledge will need to be improved, but currently the potentially positive impacts of anisotropy are conservatively not taken into account in our post-closure safety assessments. Section 5.1.6.2 provides some quantitative values of anisotropy in the Boom Clay from Belgium. Anisotropy is also an important property if gas, generated by corrosion of metals in the EBS exceeds the solubility limit, since gaseous migration is also preferentially horizontal, following the sedimentary layering.

5.1.6.2 Low permeability leading to diffusion dominated transport

As explained in Text Box 5-1, the low permeability of the clay limits the rate at which groundwater can enter disposal tunnels. This restricts the rate at which engineered barriers can alter and degrade during the post-closure phase. In addition, since the low permeability implies that the pore water within the clays is virtually stagnant, movement of any radionuclides away from a GDF will only be by diffusion.

The intrinsic permeability and the hydraulic conductivity of the clay are related to each other (see Text Box 5-1). Quantitative values for the hydraulic conductivity of Dutch Paleogene clays measured at suitable disposal depth are not available at present. Boom clay in Belgium at 225 meters depth has been found to have a vertical hydraulic conductivity of 1.7×10^{-12} m/s and a horizontal hydraulic conductivity of 4.4×10^{-12} m/s (Levasseur et al., 2021) or 3.2×10^{-12} m/s (Aertsens et al., 2023). The permeability of poorly indurated clay such as Boom clay is sensitive to changes in effective stress, which can influence the pore structure. For example, the permeability can be a factor of two lower at 500 m depth, compared to 225 m depth (Harrington et al., 2017). Further Boom clay water permeability and diffusion measurements as a function of compressive load are being made by the British Geological Survey in the EURAD-1 WP GAS project. The distribution in size of pores also depends on the salinity. Larger pores become dominant with increasing salinity, leading to higher permeability and reduced swelling capacity and compressibility of the clay (Nguyen et al., 2013).

The diffusion values of water and charged species such as radionuclides can be extracted from modelling of the experimentally measured migration of tritiated water (HTO). The in-situ diffusion values for HTO at the Belgian URF facility at a depth of 225 meters in Boom clay are 1.4×10^{-10} m²/s to 1.9×10^{-10} m²/s horizontally and 0.7×10^{-10} m²/s to 0.8×10^{-10} m²/s vertically (Aertsens et al., 2023): i.e., they show an anisotropy of 2:1. The diffusion values of dissolved species in water are multiplied by a porosity-relationship such as expressed in Archie's law (Van Loon et al., 2003) to obtain diffusion values in porous media such as clay and concrete. Millington and Quirk (1961) developed an analytical expression to account for the effects of the size of pores, which is used in the EURAD-1 WP GAS project (Levasseur et al., 2021). Their expression was used to determine the diffusion value of water for COVRA's waste package mortar (Blanc et al., 2024). The expression by Millington and Quirk was also used to determine the diffusion values for charged dissolved species in cementitious materials (Samson and Marchand, 2007).

The diffusion values for dissolved neutral species, anions, cations and cation-dissolved organic complexes were determined in the Belgian programme. A different expression to Millington and Quirk or Archie's law was used to account for the distribution in the size of pores: a so-called pore diffusion coefficient or apparent diffusion coefficient that is multiplied by the porosity. The pore diffusion coefficient is divided by a retardation factor to account for sorption. This retardation factor is determined by the porosity, distribution coefficient (K_d value) and bulk density of the clay. For anions, this retardation factor is 1 (i.e., there is no retardation) and, for cations, is larger than 1 (Weetjens et al., 2012). In the conceptual model developed in Belgium, cations can never have a diffusion value larger than neutral species and anions.

In the conceptual model developed in OPERA, some diffusion values for dissolved cations did exceed the diffusion values for dissolved anions and neutral species (see in Table 5-2).
The maximum pore diffusion values for cations such as Cs (85×10^{-10} m²/s) are larger than the pore diffusion values of neutral species such as HTO (max. 2.6×10^{-10} m²/s) and anions such as Se (max. 1.3×10^{-10} m²/s) (Meeussen et al., 2017). The minimum in the range of determined distribution coefficient (K_d value) (Schröder et al., 2017a) is so small that retardation becomes negligible.

Permeability

A key contributor to the safety case for geological disposal is the emplacement of the radioactive wastes in an environment with extremely low groundwater flow rates. The movement of water through a saturated rock formation is determined by its hydraulic conductivity K (m/s) which is the ratio of the volumetric flux (m³/s) to the hydraulic head (m), as expressed in Darcy's Law. Hydraulic conductivity depends on the properties of the fluid (saturation, viscosity, temperature, and density), whereas permeability k (m²) is an intrinsic property of a porous material, and it depends only on properties such as pore size, tortuosity, and surface area.

Type of dissolved species	Accessible porosity	Pore / apparent diffusion coefficient in $\times10^{-10}$ m²/s	Source
HTO - parallel to bedding	0.35-0.37	4-5	(Aertsens et al., 2023)
HTO - perpendicular to bedding		2.1	
Neutral species e.g. HTO	0.14-0.40 0.27	2.0-2.6 2.3	Meeussen et al. (2017) for the range accessible porosity and pore diffusion coefficient as determined for an effective stress from 2.4 MPa (Belgian URF) until 6.9 MPa & the fresh water conditions in Belgium until as saline as seawater ionic strength of 0.6 M Schröder et al. (2017b) for the single (default) value for the accessible porosity and pore diffusion coefficient
Anions Cl and I	0.05-0.40 0.23	1.0-1.6 1.3	
Anion Se (I)	0.05-0.40 0.23	0.84-1.3 1.0	
Anion Nb	0.05-0.40 0.23	0.67-1.1 0.86	
Cations (alkali) e.g. Cs	0.14-0.40 0.27	1.4-85 11	
Cations (alkaline earth) e.g. Sr	0.14-0.40 0.27	1.9-3.3 2.5	
Dissolved Organic Matter (cationic dissolved complexes: (post-)transition metals (e.g. Pb, Pd, Cd), lanthanides and actinides)	0.07-0.17 0.12	0.057-0.57 0.18	
Cation dissolved complex e.g. U	0.07-0.17 0.12	0.057-0.57 0.18	

Table 5-2: Pore diffusion coefficients and accessible porosity for dissolved species used in OPERA and at in-situ scale measured under in-situ stress conditions at depth in the Belgian URF.

Permeability is a measure of how well fluids in general flow through a material. Porosity is a measure of the amount of void space in a material. A material can have a high porosity, but if the voids in the material are not connected, its permeability will be lower.

Hydraulic conductivity or permeability can be measured on cores in the laboratory by flowing water through a core sample and measuring the pressure drop across the core, or by setting the pressure difference, and measuring the flow rate produced. The relationship between hydraulic conductivity K and permeability k is given by $K = k\rho g/\mu$, where ρ is the density, g the acceleration due to gravity and μ the viscosity. For example, for water at 20C with a viscosity of 0.001 Pa.s and a density of 998 kg/m^3, an intrinsic permeability of 1.7×10^{-19} m^2 corresponds to a (saturated) hydraulic conductivity of $1.7x10^{-12}$ m/s.

In the multibarrier system with clay host rock, the clay has a very low permeability and is saturated with groundwater. Any water that enters the disposal tunnels to fill initial void space or to react chemically with the engineered barriers must be replenished from the groundwater in the host rock. The low permeability of the clay sets an upper limit on the potential inflow rate to the EBS. The calculations illustrated in Chapter 7 for disposal tunnels imply that this limit is less than 0.1 kg/m of tunnel length per year. In Chapter 6, on the engineered barrier system, it is shown that the same a rguments apply to the low permeability concretes employed.

Diffusivity

Radionuclides from the waste that are dissolved in flowing groundwater will be transported through the clay host-rock formation at a rate determined by the flow rate and also by the chemical interactions that can take place with the clay. If the permeability is so low that flow is negligible, then radionuclides can still migrate by diffusion, driven by the concentration gradients in the system (Fick's Law). The ratio of the diffusive flux to the concentration gradient is the diffusion constant, D.

Because diffusion measurements are very time consuming, it is not feasible to measure diffusion coefficients for all radionuclides of interest. It is therefore important to develop procedures for making reliable estimates of diffusion coefficients.

5.1.6.3 Self-healing of fractures

Fractures will form in the clay host rock in the vicinity of a tunnel during its excavation and will locally increase the host rock permeability. The characterisation of this Excavation Disturbed Zone (EDZ) in clay host rocks has been investigated in the EC's SELFRAC project (Bernier et al., 2007). EDZ fractures can close up again, given sufficient compressive load or confining pressure and access to water. The closure of fractures can be measured via the increase in pore water pressure. The conceptual understanding of this process has been developed in the Swiss programme (Alcolea et al., 2014): water suction occurs towards the fractures, which are at atmospheric pressure immediately after excavation. Equilibrium is achieved when the water pressure in the fractures reaches the formation pressure. There is evidence that the impact of construction in poorly indurated clays is much less than in the indurated clays considered in Switzerland. The self-healing process for poorly indurated clay can be so fast that, in the safety assessment, the same permeability can be assumed to pertain at the start of the post-closure phase in the safety assessment. For example, the measured hydraulic properties near the interface between concrete and clay of the PRACLAY tunnel in the Belgian URF were similar to those measured further away from this interface (Dizier et al., 2017): i.e., the EDZ was too small to cause a measurable impact on the hydraulic conductivity of clay. In contrast with indurated clay such as Opalinus Clay in Switzerland with a virgin hydraulic conductivity of 4.4×10^{-14} m/s increased initially till 10^{-7} m/s in the EDZ (Alcolea et al., 2024) i.e. 6 to 7 orders in magnitude higher.

The poorly indurated Paleogene clay formations have a high self-healing capacity due to their content of swelling clay minerals such as smectite (e.g., montmorillonite). This swelling potential depends on the salinity of the clay pore water (e.g., Nguyen et al. (2013). Table 5-3 shows the mineralogical compositions of:

- poorly indurated clay: Boom Clay in Belgium (Honty and De Craen, 2012) and Rupel Clay in the Netherlands (Griffioen et al., 2017);
- indurated clay: Callovo-Oxfordian clay at Bure (Wenk et al., 2008), the reference host rock in France, and Opalinus Clay (Traber and Blaser, 2013), the reference host rock in Switzerland.

The mineralogy of the Paleogene clays has changed little since they were laid down. There has been some development of microbially formed pyrite, calcite and other calcareous minerals during the shallow burial of the sediments (De Craen et al., 1999). Glauconite is a microbially formed mineral alteration of clay minerals and is found in samples from the DAPWELL drilling in the Paleogene sand and silt formations (Vardon et al., 2022). The limited cementation caused by the formation of carbonates ensures that Paleogene clays can deform plastically (NIROND, 2013).

As the Paleogene clays were being deposited, the continued sedimentation and thickening induced compaction. The connecting pore throats in host rocks such as Boom Clay are less than 10 to 50 nm. The range in diameters of microbes is between 0.2 µm and 2 µm so that they are immobile. In addition, microbial activity stopped, since the connecting pore throat sizes became too small to allow transfer of the proteins and nucleic acids essential to maintain life (Wouters et al., 2016). The microbes present in clays at disposal depths are in a dormant phase, but their activity can be triggered when fractures are generated, increasing porosity and allowing water movement (Swanson et al., 2018).

Mineral	Chemical formula	Belgian	Dutch	French	Swiss
Muscovite	$KAl_2(AlSi_3O_{10})(F,OH)_2$				
Illite/Muscovite		18.8			
Illite	$(Ca_{0.05}Na_{0.03}K_{0.61})(Al_{1.53}Fe^{3+}{}_{0.22}Fe^{2+}{}_{0.03}Mg_{0.28})$ $(Si_{3.4}Al_{0.6})O_{10}(OH)_2$		10.9	33.9	24
Ill/Sm mixed layer		21.8		1.9	9
Smectite^{montmorrilonite}	$(Na,Ca)_{0.33}(Al,Mg)_2(Si_4O_{10})(OH)_2\bullet nH_2O$		25.4		
Kaolinite	$Al_2Si_5O_5(OH)_4$	7.4	4.1	3.3	18
Chlorite	$(Mg,Fe)_3(Si,Al)_4O_{10}(OH)_2\bullet(Mg,Fe)_3(OH)_6$	2.2	1.1	3.8	9
Chl/Sm mixed layer					0
Clinoptilolite/ Heulandite	$((Na,K,Ca)_{4\text{-}6}Al_6Si_{30}O_{72}\bullet 24H_2O/$ $((Na,Ca)_{4\text{-}6}Al_6Si_{30}O_{72}\bullet 24H2O$		0.6		
Quartz	SiO_2	38.9	42.0	24.0	20
Calcite	$CaCO_3$	0.6	5.3	24.3	13
K-feldspar	$KAlSi_3O_8$	4.8	6.7		2
Albite	$NaAlSi_3O_8$	2.0	2.4		
Plagioclase	from $NaAlSi_3O_8$ till $CaAl_2Si_2O_8$			3.9	0.9
Anorthite	$CaAl_2Si_2O_8$				
Dolomite/ankerite	$CaMg(CO_3)_2$	0.3		3.5	0.4
Siderite	$FeCO_3$	0.1			4
Pyrite	FeS_2	2.1	1.4	1.4	1
Apatite	$Ca_5(PO_4)_3(F,Cl,OH)$	0.2			

Table 5-3: Average or reference mineralogical compositions (wt%) of clays being considered as GDF host rocks in various national programmes, as collected in EURAD-1 WP ACED (Neeft et al., 2022; Neeft et al., 2019)

A useful natural analogue illustrating negligible microbial activity in clays is the Dunarobba fossil forest in Italy, in which 2 million year old trees have been preserved in Quaternary clay. These trees were protected against microbial degradation by the clay and therefore have cellulose contents similar to present-day wood (De Putter et al., 1997; Lombardi and Valentini, 1996). Figure 5-7 shows that these trees retained many of their natural properties, and the wood can still be sawn and chopped.

Figure 5-6: 2 million year old tree remains preserved, as wood that can be sawed, in a clay formation at Dunarobba, Italy Image taken by Neil Chapman during the preparation of the movie Traces of the Future (1994) currently available at the website natural-analogues.com.

5.1.6.4 Chemical retardation capacity

In a low permeability clay formation in which the movement of both water and dissolved chemical species is controlled by diffusion (see 5.1.6.2), the distribution of an element naturally present in the formation between the clay material itself and its pore waters will be influenced by:

- its chemical form and location within the clay mineralogy (whether bound within a mineral structure or present as an exchangeable ion in a clay mineral layer);
- its solubility with respect to the minerals in which it is located and its solubility limit in the pore waters;
- the composition of the pore waters.

The potential diffusive movement of an element in pore waters will then be controlled by its pore water concentrations and concentration gradients across the formation, and out into surrounding formations, along with the ability of clay mineral phases to adsorb and retain the element. The way that an element is currently distributed, and how it has been redistributed in response to the evolving properties of the formation and its surrounding environment since the clay formation was deposited, can give useful analogue insights into how radionuclides might migrate out of the EBS over long periods of time through the same environment.

At the simplest level, the distribution of trace elements between the minerals in the clay formation and its pore waters can provide insights into the retention capacity of clays. When the clay formation was laid down as a marine deposit it would have comprised a muddy mixture of clay and other minerals in a matrix of sea water. Cation exchange and other sorption processes (see below) allowed the clay minerals to progressively scavenge the small amounts of trace elements present in the large volumes of sea water with which they were in contact, and to concentrate them. As compaction and burial of the sediments took place, remnant sea water was expelled, with the remainder incorporated into the developing porosity of the clays. Present-day concentrations of trace elements such as U, Cs and Ni are expected to be orders of magnitude higher in the solid clay materials than in their pore waters. Currently, however, uranium is of one of the very few trace elements for which a measurement in clay pore waters is available (see Figure 5-8). The concentration of trace elements in sand pore water overlying or underlying the clay formation could be an alternative indicator of concentrations of trace elements dissolved in adjacent clay pore water. Data on nickel are available (Griffioen, 2015) and Figure 5-8 shows that the nickel content in sand pore water is about 10,000 times smaller than in the clay rock. The overlying sandy formation is much more permeable than the clay host rock. If these data are not available, data from seawater (Nozaki, 1997) can also be used as an indicator. The trace elements are then assumed to have been in the clay minerals at start from sedimentation (see Figure 5-1). Figure 5-7 shows that the concentrations of dissolved caesium and niobium in seawater are at least 20,000 times smaller than in the clay rock.

All these examples highlight the potential chemical retardation capacity of the clay rock. These data show the ability of clay minerals to collect and retain trace metals from water with which they are in contact. Any radionuclides entering solution into clay pore waters would be subject to such processes. As a start, we consider elements that have been selected because some of their radioactive isotopes are present in the waste and have been shown to contribute to exposures from releases in the OPERA safety assessment. Figure 5-7 shows that long-lived radionuclides such as ^{135}Cs and ^{94}Nb can be present in smaller concentrations in a waste form than the non-radioactive caesium and niobium in the clay rock. Even in cases in which the concentrations of radionuclides in the waste are larger than the concentration of non-radioactive counterparts in clay rock, the volume of the wastes is many times smaller than the clay rock surrounding the EBS. Consequently, the in-situ distribution of non-radioactive trace elements may be indicative of the behaviour of radionuclides from the disposed of waste in the clay host rock.

The capacity of a clay host rock to chemically retard radionuclide movement is enhanced by reducing conditions, which limit the solubility of redox sensitive elements such as Se, U and Pu, and by ion exchange with clay minerals and immobile dissolved organic matter, which limits the amount of dissolved cations and cationic complexes. Ion exchangers are insoluble solid materials or immobile dissolved materials that carry exchangeable, positively charged cations and negatively charged anions. Ion exchange in these minerals is a reversible chemical reaction that takes place between ions held near a mineral surface by unbalanced electrical charges

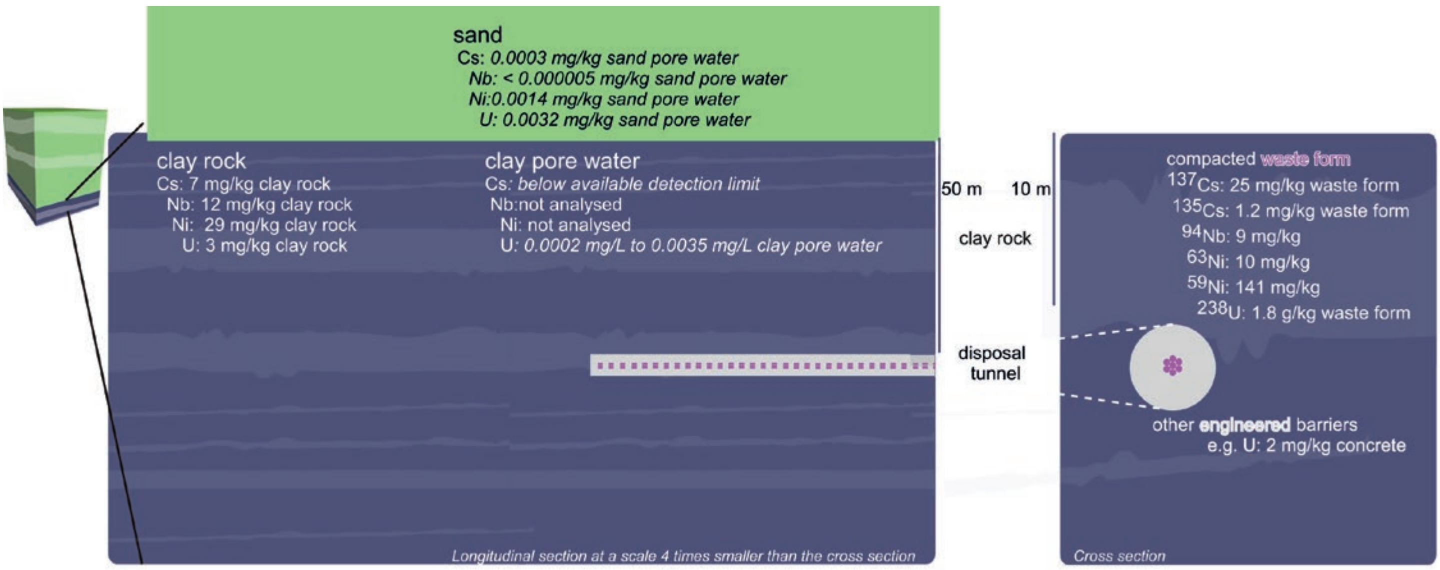


Figure 5-7: Schematic depiction of a part of the clay host rock with a disposal tunnel and encapsulated compacted metallic waste forms. Examples of concentrations of elements as found in Oligocene pore water almost as saline as seawater (Griffioen, 2015) or seawater (Nozaki, 1997), Boom Clay in the Netherlands (Koenen and Griffioen, 2014), Boom Clay pore water collected in Belgium (Mol) (De Craen et al., 2004). Radionuclide concentrations in the waste form are determined by the activity per canister and weight (Verhoef et al., 2016) and their half-lives.

within the mineral framework and ions in pore waters in contact with the mineral. Generally, the excess charge on the clay minerals is negative, and they attract cations from the pore waters to neutralize this charge. The chemical reactions in ion exchange are restricted by the number of exchange sites on the mineral and by the strength of the bonding of the exchangeable cations to the mineral surface. Over millions of years, the clay minerals in Paleogene formations have exchanged cations with pore waters. For example, Ca in the clay has been shown experimentally to be exchanged with dissolved lanthanides such as Eu (Baeyens et al., 1982). The effectiveness of cation exchange in controlling the concentrations of radionuclides that might become mobilised from the waste into pore waters in the clay host rock is defined by the Cation Exchange Capacity of the minerals involved (see Text Box 5-2).

Gathering information on the locations and concentrations of trace elements in the Paleogene clays is thus an important aspect of developing the safety case, as it enables modelling of the behaviour of radionuclides released from the waste, as well as providing support for the results by allowing comparison with the behaviour of analogous elements in the same natural system over long periods of time. In this respect, the current analysis aims to address the question (Verhoef et al., 2020): What is the speciation of the naturally occurring radionuclides and their chemical analogues within the Paleogene clays at suitable disposal depth? To do this, we use data on clay mineralogy and composition used in OPERA (Griffioen et al., 2016; Koenen and Griffioen, 2014) as provided by the Dutch Geological Survey, which maintains a core and cuttings store, with samples from past borehole projects.

Figure 5-8 shows the locations of boreholes from which cores identified as Boom clay have been taken. Despite the differences in depositional environment (marine mud or deep marine mud in Figure 5-1) and thickness in Figure 5-5, the clay content and organic carbon content seems to be uniform. The content of trace elements, sulphur, total carbon and organic carbon have been determined in samples taken at all the 17 locations labelled by Roman numerals in Figure 5-8.

Box 5-2 Cation exchange capacity

The affinity for ion exchange by cations decreases as follows (Helfferich, 1962; Stumm, 1992)

$Ba^{2+} > Pb^{2+} > Sr^{2+} > Ca^{2+} > Ni^{2+} > Cd^{2+} > Cu^{2+} > Co^{2+} > Zn^{2+} > Mg^{2+} > UO_2^{2+} > Tl^{+} > Ag^{+} > Cs^{+} > K^{+} > Na^{+} > Li^{+}$

Thus, a clay mineral containing exchangeable Na could exchange some of that for Cs in pore water. The total amount of ion exchanged cations in the clay rock is determined by the Cation Exchange Capacity (CEC). For the average mineralogical composition of the clays in Table 5-3, the theoretical CEC has been estimated to be 29.5 meq/100 g clay (Griffioen et al., 2017) for an average total clay content of 41.5 wt%. For a sample with a similar total clay content (44.9 wt%), the measured CEC is 18.5 ± 4.5 meq/ 100 g clay (Behrends et al., 2016; Behrends et al., 2015). This value is similar to the CEC values used in the Belgian programme: 18.5 meq/100 g clay (Salah and Wang, 2014) or 19 meq/100 g clay (Honty and De Craen, 2012). The measured CEC can be smaller than the theoretically determined value if other cations, with a higher affinity, have already used up a part of the exchange capacity. Trace elements (Cs, transition metals, lanthanides, Th etc) use less than 1 or 2 meq/100 gram of the CEC of clay: i.e., assuming them to be ion exchanged and not incorporated within minerals. Consequently, there is more than sufficient ion exchange
capacity in the clay host rock if radionuclides are released from the waste form and packaging material.

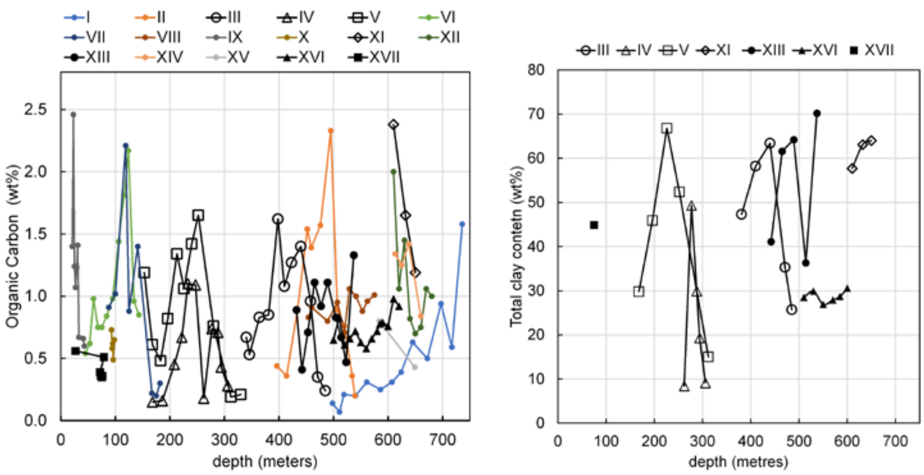


Figure 5-8: Boom clay sample location used in OPERA (left), their clay content as determined with XRD and their organic carbon content (Koenen and Griffioen, 2014).

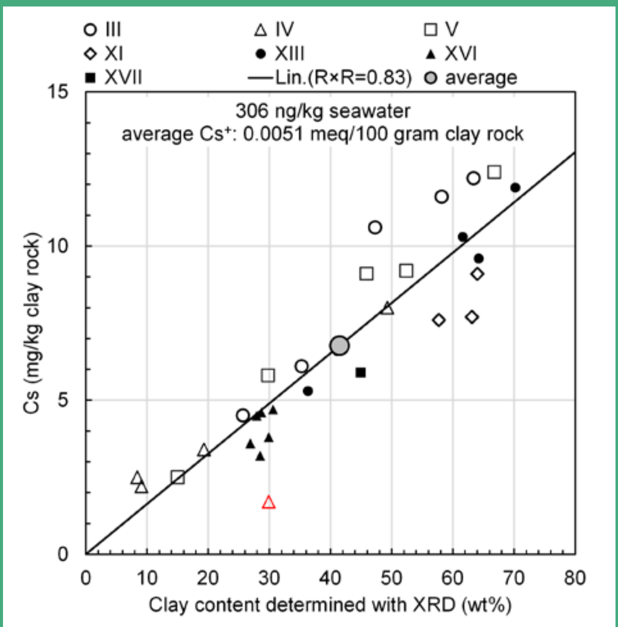
Box 5-3 Evidence of caesium containment by a Paleogene clay

Traces of Cs caesium present in Paleogene clays are highly correlated with the clay mineral content, as shown in the figure below. If the caesium is assumed to be almost totally present in the clay due to ion exchange, then a negligible fraction - 0.0051 meq/100 g - would occupy the total CEC of the clay of 18.5 meq/100 gram clay.

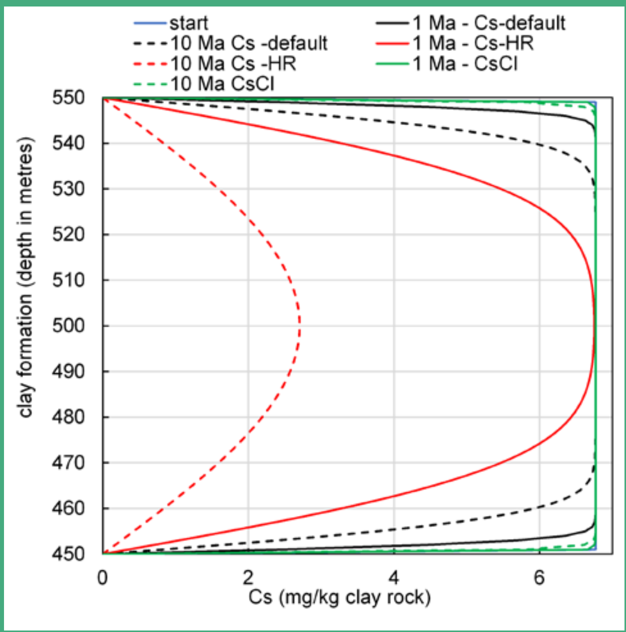
The assumption that the chemical enrichment of caesium in clay rock is caused by ion exchange is supported by additional experimental results. The caesium content in clay pore water samples taken in the Belgian URF is below the detection limit of 0.5 µg/l. Mechanical squeezing of the clay at six times the formation pressure extracted water with a caesium concentration of 3.8 µg/l (De Craen et al., 2004). The hypothesis is that surface-sorbed water, or water that is part of the diffuse layer surrounding clay mineral surfaces, may be expelled during squeezing by the collapse of the pore space, so that more soluble Cs (and Rb, REE, Th and U) could be extracted from the squeezed sample than occurs in natural pore waters taken at formation pressure at the URF depth.

The presence and chemical behaviour of caesium in the clay host rock at formation scale can be used to provide constraints for estimating diffusion values that are affected by ion exchange, which minimizes the dissolved amount available for diffusion. To do this, we modelled the outwards diffusion of Cs from a hypothetical 100 m thick clay formation (with the average Cs content shown in the figure above), into overlying and underlying sandy formations containing mobile water with a concentration of dissolved Cs equal to seawater. The calculations use the diffusion values in OPERA (see Table 5-2) and retardation factors as shown in Appendix 5.
The figure below shows the modelled concentration profiles (red curves) after 1 Ma and 10 Ma and predicts that the majority of Cs would have migrated out of the clay host rock by the present day, for the case with the lowest retardation and maximum porosity and pore diffusion values.
When retardation (blue curves) is included, Cs is predicted to have migrated out of the clay only in the regions adjacent to the sand formations. This limited transport would also be expected for radioactive caesium in the post-closure safety assessment (see Chapter 8). If it is assumed that cations cannot diffuse faster than anions and neutral species, as assumed in the Belgian model, and retardation (determined from 7 mg Cs clay host rock divided by 306 ng Cs/kg seawater) is included (green curves), the prediction of how much Cs has migrated out of the clay, reduces even further. Further work, using a higher sample resolution than was available in OPERA, will provide the evidence of whether such calculated profiles are observable in Paleogene clay, when more detailed borehole sampling becomes available for COVRA's programme.

Modelled Cs content for a hypothetical clay formation with a thickness of 100 m and an initial average content of 7 mg Cs per kg clay rock. Red and blue curves with OPERA parameters; green curves with retardation assumed with experimental data; diffusion value for CsCl is a calculated fraction of the diffusion value for water in clay.



Caesium content versus clay content in a Paleogene clay (Boom Clay). The Roman numerals in the key refer to the boreholes shown in Figure 5-8. The red triangle has a high glauconite content.

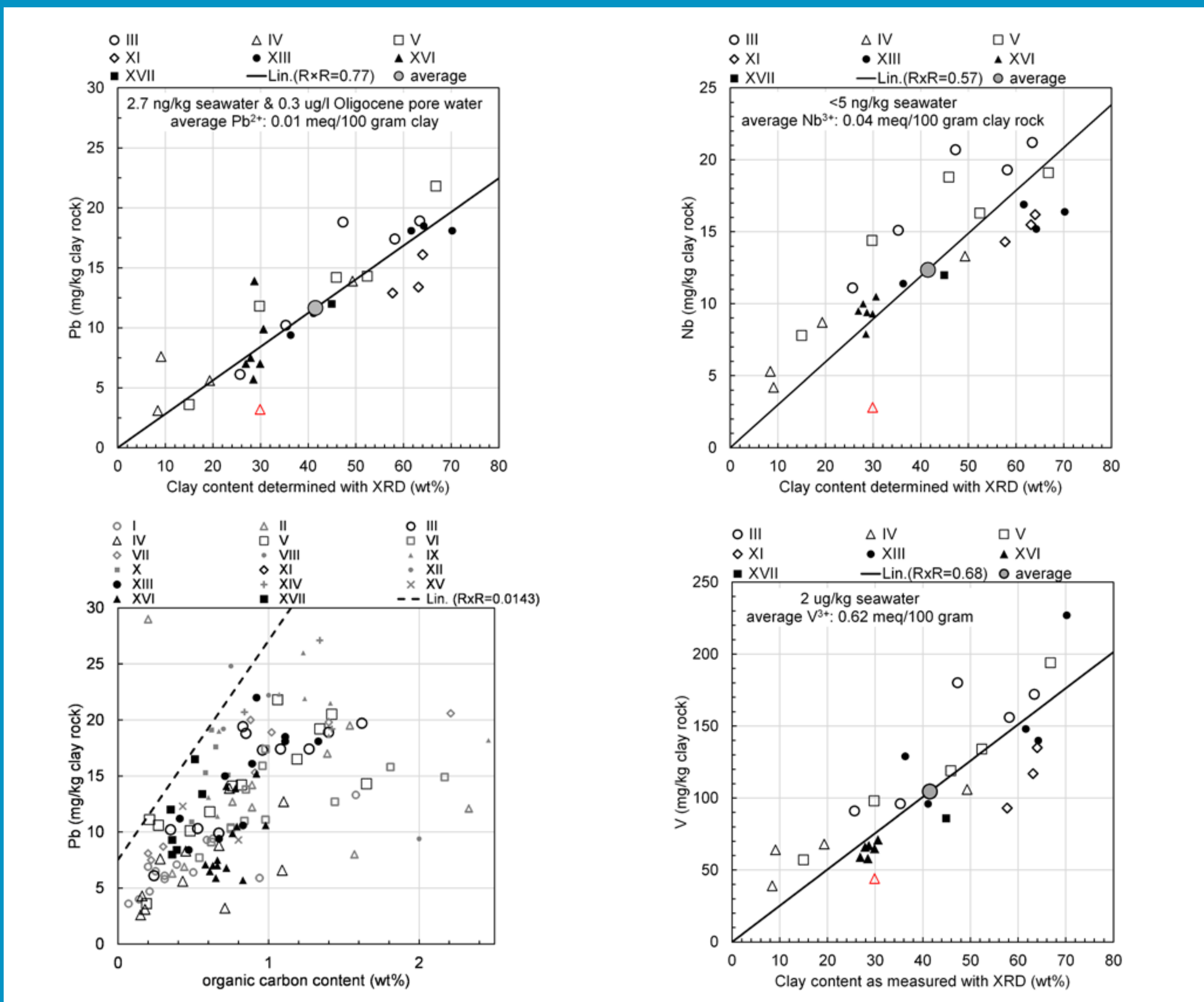


Modelled Cs content for a hypothetical clay formation with a thickness of 100 m and an initial average content of 7 mg Cs per kg clay rock. Red and blue curves with OPERA parameters; green curves with retardation assumed with experimental data; diffusion value for CsCl is a calculated fraction of the diffusion value for water in clay.

Box 5-4 Evidence of containment of (post-)transition metals in a Paleogene clay

A number of other elements show the same strong correlation with the clay content as does caesium (see Text Box 5-3). The figure below shows some examples of transition elements that are present in Paleogene clays, are also present in wastes, and have safety relevance: Pb (which is a chemo-toxic element), Nb (an isotope of which was calculated in OPERA to enter the biosphere) and V, which has similar expected behaviour to Nb and is present in greater amounts.

The similar correlations with the clay content may imply that these elements have similar chemical behaviour in the clay host rock. In OPERA, all transition metals, except Nb, were assumed to exchange with immobile dissolved organic matter, instead of clay minerals (see Table 5-2). The figure below shows that Pb is not well correlated to the organic carbon content but is much better correlated with the clay content. In the same manner as Cs (see Text Box 5-3), the distribution values for (post-)transition metals between the solid phases of the clay rock and its pore water can be used to determine their retardation factors. Also in these cases, data for clay pore water are lacking, but Pb has been measured in Oligocene sand formations (Griffioen et al., 2016), which are a better approximation for clay pore water than seawater. The Pb concentration measured in these groundwaters with a salinity close to seawater is an order of magnitude smaller (0.3 μg/l) than in the clay host rock (12 mg /kg clay rock). All transition metals were assumed to be dissolved as cations, except Nb, which was assumed to act as an anion in the clay host rock in OPERA (see Table 5-2) and anions were assumed not to be retarded in the clay rock (infinite solubility). However, the figure below clearly shows that Nb, like the other transitions metals, may behave as a cation, whose migration is retarded by ion exchange with clay minerals. The assumption of whether radioactive Nb is retarded in the clay host rock has implications for the post-closure safety assessment (see Chapter 8).



Lead, niobium and vanadium concentrations in a Paleogene clay (Boom Clay) as a function of the clay content. The Roman numerals in the keys refer to the boreholes shown in Figure 5-8. The red triangle has a high glauconite content. The bottom left-hand figure shows the lead concentration as a function of organic carbon content. Locations II & XII have, respectively one and two measurements with more than 100 mg Pb/kg (Koenen and Griffioen, 2014) i.e., these measurements are out of the scale.

5.2 Rock formations that surround Paleogene clay formations

The Paleogene clays are underlain and overlain by sands of Paleogene and/or Neogene age (see for example Figure 5-3). Above this, most of the superficial formations in the Netherlands are Quaternary sediments. All Paleogene, Neogene and Quaternary sediments are relatively soft, unconsolidated formations, and are susceptible to erosion by wind and water. The Paleogene clay formations therefore need to be overlain by a sufficient thickness of other rock formations to ensure continued isolation of the GDF, which means that processes that cause deep or more rapid erosion need to be understood. Here, the impacts of major climate changes need to be considered.

5.2.1 The potential impact of climate change on isolation

The latitudinal position of the Netherlands makes it susceptible to ice cover when glaciations occur. The north of the Netherlands has been affected by ice cover during the glaciations that occurred during the past half million years. The ice sheets that formed during the Elsterian (0.475 to 0.410 million years ago), Saalian (0.370 to 0.130 million years ago) and Weichselian (0.115 to 0.010 million years ago) never reached the southern provinces of the country (see Figure 5-9).

The physical load of the ice cover above the location of a GDF would impact the containment properties of the clay host rock. The physical load on an ice sheet could force pore water out of the clay formations. This would depend on the assumed thickness of the ice sheet. In OPERA, the ice-sheet thickness was estimated to be around 200 m in the northern Netherlands during the most severe, Saalian glaciation (ten Veen, 2015), which is significantly smaller than the 1000 m assumed earlier in CORA (Wildenborg et al., 2000). During the retreat of the Elsterian ice sheet, subglacial erosion scoured sediments, producing channels that are up to 600 m deep in the northern Netherlands (ten Veen, 2015), as shown in Figure 5-10. Glacial basins caused by erosion during the Saalian glaciation are however rarely deeper than 150 m (van Dijke and Veldkamp, 1996).

Present models that take account of global warming indicate that the next glaciation is, in any case, unlikely to occur for more than 100,000 years (Ganopolski et al., 2016; Lord et al., 2020), so that glacial loading and erosion become a negligible hazard for some types of waste, including vitrified HLW for which the radiotoxicity after 20,000 years is smaller than that of uranium ore (Gruppelaar et al., 1998).

High flow of channelled water can also occur in situations other than the retreat of an ice-sheet. The Netherlands has a long coast-line and incision of unconsolidated formations by tidal channels in which considerable flow energy is concentrated in a narrow zone, is the most powerful type of non-glacial vertical erosion.
Modern examples include the Holocene tidal channels of the 'Pas van Terneuzen' and 'Everingen' (Westerhoff et al., 2003b) in the

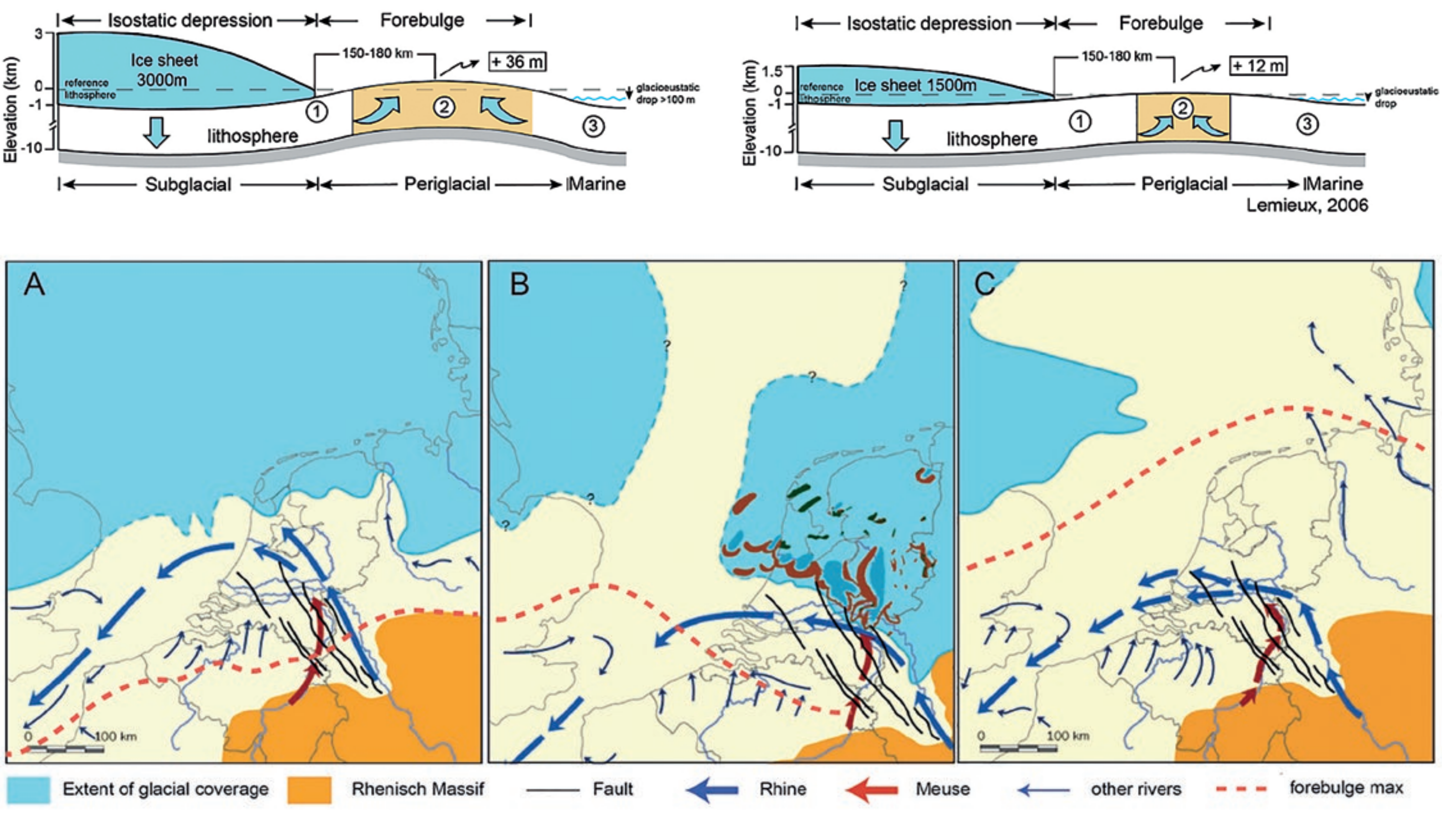


Figure 5-9: Location of ice cover (blue) and the forebulge, where the lithosphere is squeezed upwards (orange), for three different ice advance scenarios analogous to the Elsterian (A), Saalian (B) and Weichselian (C) glaciations that occurred in the Quaternary (ten Veen, 2015; Westerhoff et al., 2003a). Saalian glacial basins are shown in deeper blue and Saalian push moraines in dark brown (van Dijke and Veldkamp, 1996). Figure also presented in the OPERA Safety case.

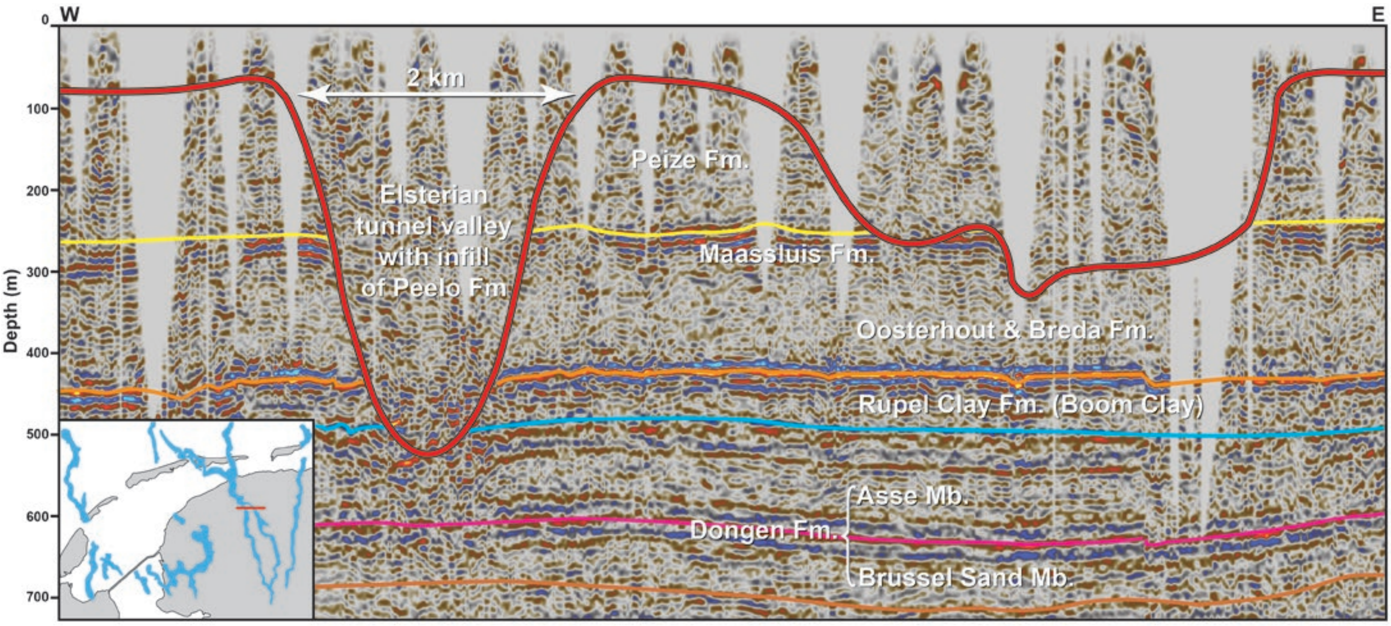


Figure 5-10: Erosion by tunnel valleys from the Elsterian ice age in the north of the Netherlands (ten Veen, 2015). Figure adapted as presented in the OPERA Safety case.

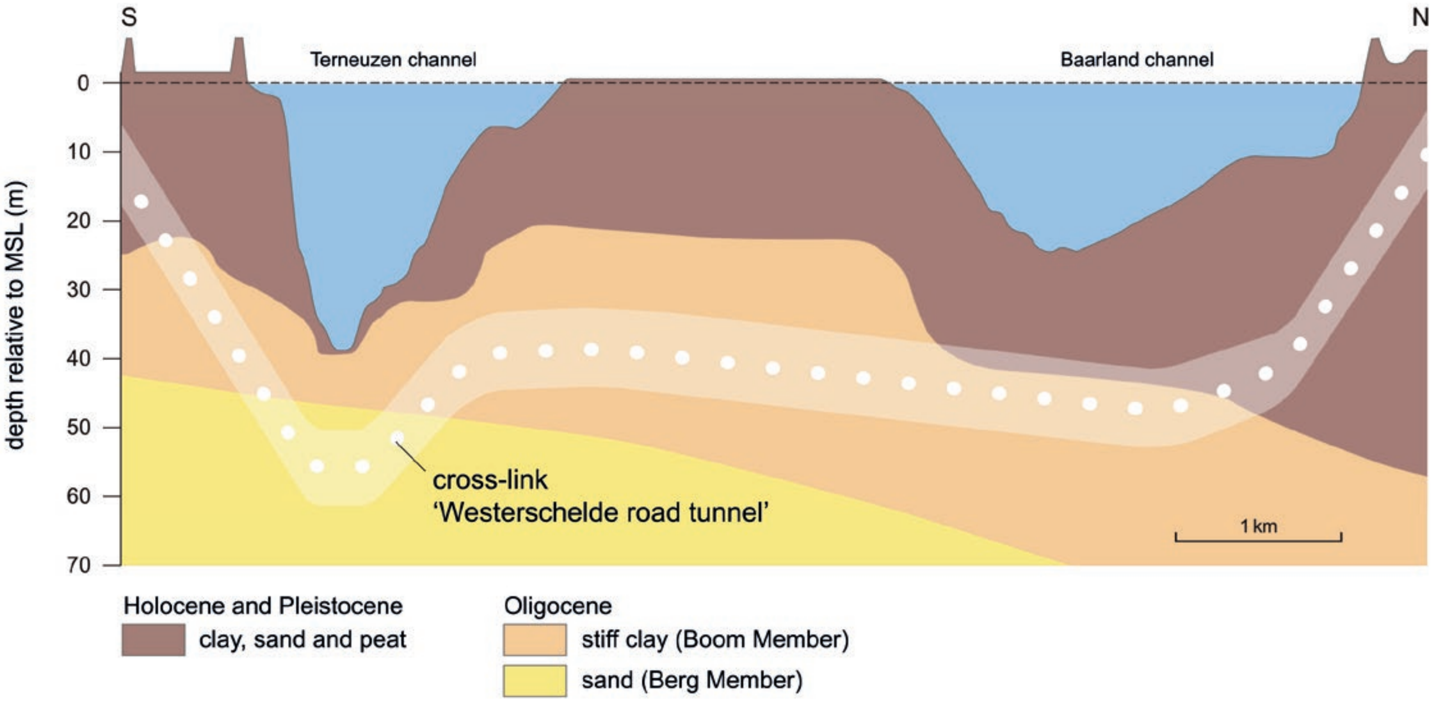


Figure 5-11: Erosion by tidal channels in the southwest of the Netherlands (de Mulder and Ritsema, 2003). Note that the Westerschelde road tunnel is constructed in the clay of the Boom Member (the tunnel is indicated by the pale marking, with intersecting escape tunnels, indicated as white spots).

southwest of the Netherlands (Figure 5-11) which are tens of meters deep. Elsewhere, these channels can be up to 100 m deep and have eroded down into the Boom Clay. The difference between subglacial erosion in the north (up to 600 m) and tidal channel erosion in the southwest (up to 100 m), illustrates that controls on suitable disposal depths for isolation of HLW may differ geographically in the Netherlands. A minimum disposal depth of 200 m is envisaged to be adequate for the geological isolation of HLW, as it is considered that deeper glacial erosion within the safety relevant timescale is not likely to occur anywhere in the Netherlands.

As explained in Chapter 2, isolation also requires that the likelihood for drilling or excavation in the search for natural resources in the future is very low. The sandy formations that surround the clay host rock have such a high permeability that these formations are aquifers, which can be used to extract water (if potable) or extract heat (geothermal energy) or store heat. Activities for the extraction or storage of heat that would result in potential radiological exposure are limited to drilling, which could affect drilling crews, but not cause wider exposures of the public (see Chapter 7). Paleogene clay host rocks are preferred due to their expected salinity in order to reduce potential conflicts with groundwater activities (see at the start of section 5.1).

5.2.2 Seismicity and deformation in rock formations beneath the clay host rock

All national programmes avoid siting a GDF near an active fault structure. Major active fault systems, such as the Roer Valley Graben, are scarce in the Netherlands (see dark lines in Figure 5-12) and would be avoided in siting the GDF. Seismic events have occurred in the past 60 years in the northern part of the Netherlands due to gas exploration activities, but these activities ceased in 2024. Neotectonic features and natural and induced seismicity will be factors to be taken into account during siting of the GDF, but they are considered unlikely to have significant bearing on the safety case. Natural tectonic seismic events (shown in grey in Figure 5-12) are concentrated in the south-east of the Netherlands, while events induced by gas extraction, underground gas storage, geothermal heat extraction, salt solution mining and post-mining water ingress are shown in blue in Figure 5-12. The Royal Dutch Meteorological Institute (KNMI) interpreted events as induced based on (a combination of) the following three criteria:

1. Location, north or south of the Netherlands;
2. Preliminary hypocentre depth estimate: shallow (< 5 km) versus deep (> 5km);
3. Proximity to current subsurface operations.

Salt domes occur beneath the Paleogene clay formations, especially in the northern part of Netherlands. Depending on the rate of upward movement (diapirism) and dissolution at the top and flanks of these domes by deep groundwaters, the overlying clay formations could be compacted, leading to local thinning of the Paleogene clay formations. Compaction rates of Paleogene clay formations induced by the rise of salt domes are expected to be smaller than the potential compaction rates by glacial loading, since the rates of upward movement of the salt domes are small. However, the associated tectonics may alter the integrity of the Paleogene clay formation. In OPERA, therefore, locating a disposal facility in a Paleogene clay formation above a salt dome was not recommended (Vis and Verweij, 2014).

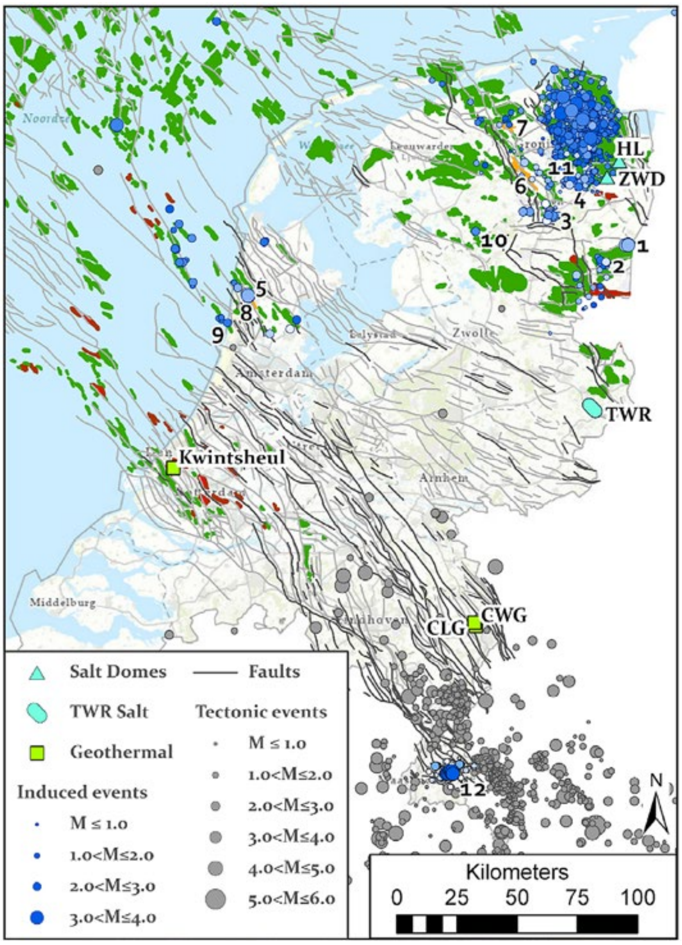


Figure 5-12: Overview of seismic events recorded for the period 1986-2021 in the Netherlands. The dark lines indicate potentially tectonically active faults, the light grey lines indicate faults in Permian formations (Muntendam-Bos et al., 2021). Oil (denoted as red), gas (denoted as green) fields and gas storage (denoted as orange).

Box 5-5 Uncertainties being investigated

The permeability of poorly indurated Paleogene clays is sensitive to stress, with lower permeabilities being measured at higher stresses (Harrington et al., 2017). The reason for this behaviour is that the connecting pore throats are expected to be smaller at higher stress (higher compaction load at higher depth) for clay rock with the same mineralogy and salinity. The connecting pore throat is determined by the size distribution of pores. For a post-closure safety assessment, diffusion values are needed, and these values also require knowledge of the tortuosity. The tortuosity is also assumed to depend on the size distribution of pores. Therefore, smaller diffusion values for water are expected for clay formations at greater depth. There are, however, no measurements to support this hypothesis. The British Geological Survey is performing measurements on saturated synthetic clays in which gas is used as a probe to obtain the diffusion values of dissolved gases as a function of the compaction pressure. These results can be used later to obtain diffusion values for water as a function of stress.

The concentration of trace elements present in clay host rock frequently exceeds the concentration of the radioactive isotopes in the waste forms. Investigating the distribution and behaviour of trace elements in clay host rock may therefore be used to test and verify assumptions made in the post-closure safety assessment, such as assumptions about the solubility of elements in the clay pore water and their distribution coefficients between the solid or immobile phases in the clay rock and the clay pore water.
Utrecht University is investigating the concentration of the main components and trace elements of pore water obtained from clay cores extracted in Delft (Vardon et al., 2022).
The clay cores have been obtained with push-coring techniques that minimize the disturbance of clay and the cores are conditioned and stored under appropriate conditions to minimize drying and oxidation. TNO is measuring trace elements in the clay rock obtained in Delft, as well as in Watervliet clay obtained in Borsele (PCR, 2013), together with the main components and clay mineralogy.
The concentration of trace elements in waters collected from the sandy formations that overlie and underlie the Watervliet clay is also being measured, in order to elucidate potential concentration gradients at formation scale.

Unreinforced concrete dome of the Pantheon in Rome was built in about 120 AD and retains its structural load-bearing integrity 1900 years later. Source: Neil Chapman.

How the waste are contained when radioactivity levels are the highest

6. The Engineered Barrier System

This section describes the materials, safety functions, behaviour and evolution of the components of the engineered barrier system (EBS) in the disposal tunnels.

The EBS is dominated volumetrically by cementitious materials in the form of various types of concrete used in conditioning some wastes, in waste packages and the buffer in the supercontainer, as tunnel floors and backfills, and in tunnel support liners. Consequently, the properties and behaviour of concrete, in particular its interaction with steel and the clay host rock, are central to understanding how the EBS evolves, and these are dealt with first in section 6.1. The following section, 6.2, examines the waste forms, their packaging and evolution, and how these are dealt with in the safety assessment. Section 6.3 looks in detail at the supercontainer design for HLW.

6.1 Properties of concrete

Concrete is employed throughout the GDF and must fulfil its functions in contact with steel, clay, vitrified HLW and other waste forms. To predict its short and long-term behaviour in all relevant situations, it is important to understand the physical and chemical processes that determine its evolution. A key factor in this respect is the nature and evolution of cement pore waters.

6.1.1 Concrete pore water composition

Concrete is a solid material made by mixing an aggregate of sand or stone with a cement paste that is composed of high-temperature heat-treated and powdered clays and limestone, and water.
The pH of aged concrete pore water depends on which minerals are present in the cement (see Figure 6-1, adapted from Atkinson et al. (1985)). Initially, concretes made with many commercially available cements contain dissolved alkalis (KOH, NaOH) which makes the initial concrete pore water highly alkaline with a pH 13 or higher (Atkins et al., 1991; Kempl and Copuroglu, 2015). The pore water in concrete exposed to or immersed in an external source of water (e.g., in clay or granite) will exchange dissolved species, initially losing alkalis to the external water (stage I). The pH of the pore water is then controlled by the presence of portlandite, $Ca(OH)_2$, and remains 12.6 until portlandite is depleted or until the formation of calcite reaction rims around portlandite grains stops further depletion (stage II). The quantity of portlandite originally present depends on the type of cement used. For example, the portlandite content is about 20 wt% for concrete made with CEM I (cement with almost 100% Ordinary Portland Cement (OPC)) and about 5 wt% for concrete made with CEM III/B (cement in which about one-third of OPC is blended with two-third of Blast Furnace Slag) (Kempl and Copuroglu, 2015). After depletion of portlandite or when it is no longer accessible to water, calcium-silicate cement minerals control the pH of pore water. Figure 6-1 shows the stages of pH evolution in pore water in concrete, along with the dissolved calcium and silicon concentrations (Jacques et al., 2024). The reduction in pH can be caused by leaching, but reactions with ingress of dissolved

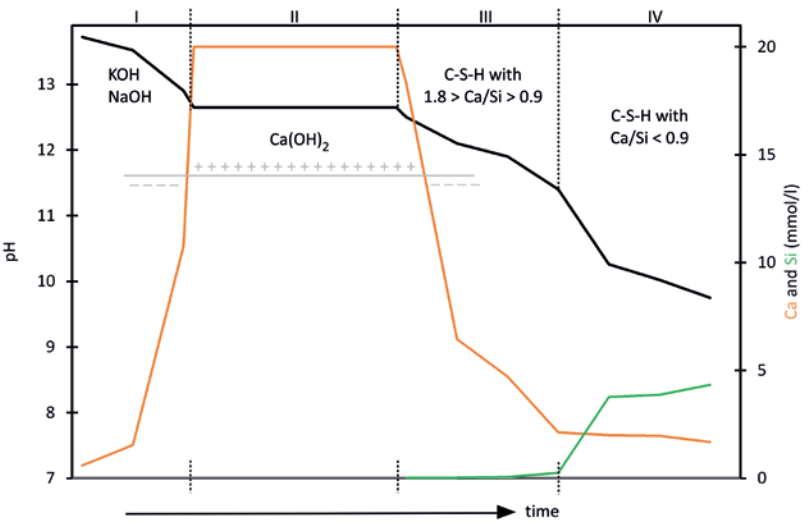


Figure 6-1: The evolution with time of concrete pore water pH (black line), dissolved calcium (orange line) and silicon (green line) at room temperature. Negative or positive of surface potential of C-S-H mineral is indicated (see section 6.1.5.1). Stages in pH by Atkinson et al. (1985) with added calcium and silicon by Berner (1992) at a pH < 12.65, dissolved calcium at a pH > 12.65 from van Eijk and Brouwers (2000) and the surface potential of C-S-H mineral from (Pointeau et al., 2006).

carbon dioxide, bicarbonate and dissolved magnesium from clay pore water are more likely processes, especially when the dissolved calcium concentration in concrete pore water has become larger than that present in the clay pore water. As pH reduces from a value of 12.6 (stage III & IV):

- the ratio between calcium and silicon in C-S-H minerals decreases (decalcification);
- the dissolved calcium concentration in equilibrium with the C-S-H mineral decreases;
- the dissolved silicon concentration in equilibrium with the C-S-H mineral increases.

Dissolved calcium leaches out from concrete if its concentration in concrete pore water is larger than in the water to which the concrete is exposed, for example, the pore water in adjacent clay. The dissolved calcium concentration in clay pore water with the salinity of seawater has been estimated to be 13.2 mmol/kg (see Table 5-1). This is similar to the dissolved calcium concentration in equilibrium with C-S-H minerals with a calcium to silicon ratio between 0.9 to 1.8. The majority of C-S-H minerals in concrete made with CEM III/B has a calcium to silicon ratio of 1.1 (Neeft et al., 2022; Neeft et al., 2019). Leaching is therefore not expected to be the main process in the chemical evolution of this concrete in contact with clay in the GDF post-closure phase. New minerals are formed in the concrete by ingress of dissolved species (magnesium, bicarbonate) from the clay pore water; for example calcium carbonate ($CaCO_3$) and brucite ($Mg(OH)_2$) replace portlandite and C-S-H minerals are replaced by M-S-H phases and $CaCO_3$.

Chemical interactions between concrete pore water and steel are understood, but the impact of calcium in the concrete pore water on steel corrosion has not yet been included in any predictive modelling (Neeft et al., 2022). Dissolved calcium allows faster generation of a protective film on the steel surface that minimizes further corrosion (Kreis, 1991) (Swanton et al., 2015). Pitting of carbon steel does not take place under anaerobic conditions with less than 100 mg chlorine per litre, even in the presence of high radiation dose rates, when oxygen is generated by the radiolysis of concrete pore water (Smart et al., 2017). In deaerated solutions, the protective film is broken at concentrations of chlorine higher than 1000 mmol/l at a pH of 13.5 (Qiu et al., 2022). Clay pore water as saline as seawater (see Table 5-1) has a chlorine concentration of 540 mmol/l which makes pitting of the overpack in contact with evolved concrete pore water unlikely as long as the pH of concrete pore water remains higher than 10 since the protecting film is then stable. The corrosion process also generates hydroxyl ions, which can also increase the pH in the vicinity of steel if they are insufficiently dissipated.

6.1.2 Mechanical strength of concrete

The required thickness of the liner depends on the diameter of the tunnel and the strength of concrete, which is highly dependent on the quality of its manufacture (see section 4.4.2). For high mechanical strength, aggregates are responsible for the strength and the cement paste holds the aggregates together. The content of air incorporated during manufacturing of the concrete should be minimized in order to avoid void volumes that act as weak points. Commercial types of cement are available with the compressive strengths required for the production of GDF tunnel liners and other EBS components. COVRA manufactures certified concrete for the containment of its compacted waste in the 200 l drums, i.e., the waste package concrete (mortar). This type of concrete has a similar content of quartz aggregates (1700 kg/m^3) to those proposed in our conceptual design for tunnel liners and the concrete buffer for HLW[14].

More detailed mechanical analysis of COVRA's waste package concrete shows that its mechanical strength increases when exposed to a decreasing relative humidity (see Figure 6-2), because more water has evaporated. This behaviour is understood to be due to the rise in capillary force by the desaturation of pores within the concrete (Neeft et al., 2021). The strength of the tunnel liner may thus increase in the operational phase of the GDF, due to evaporation of water by ventilation. In the post-closure phase, this capillary

14. Limestone aggregates have been proposed in the OPERA programme for the concrete buffer made with CEM I, but the density of limestone and quartz are the same so their shielding capacity is the same.

force diminishes as the pores become filled with water from the clay host rock. Still, the strength remains high, at around 80 MPa, so that the liner remains undeformed i.e. in the same circular shape as installed and the mechanical disturbance of the clay host rock is negligible in the long term (L4-CD-LINE-CONTA-01 in Figure 3-7). The low permeability of concrete prevents the clay host rock from forming desiccation cracks because of the ventilation, especially at the roughly 50% relative humidity that is favourable for the operational working environment. Excavation of the clay rock may cause some drying of the clay but in the vicinity of the concrete liner it becomes saturated soon after construction of the low permeability liner, which prevents the clay from drying while the hydraulic gradient is oriented towards the void space of the tunnel in the operational phase. The high compressive strength of concrete also ensures that its permeability is insensitive to stress, in contrast to poorly indurated clay host rock as explained in section 5.1.6.2.

Cement minerals in concrete can react with ingress of dissolved species from the clay pore water and the resulting minerals may degrade its strength. The type of cement determines whether these reactions take place. For example, ingress of sulphate from clay pore water might not lead to degradation of the concrete strength if sulphate resistant cement is used. This type of cement has a low tricalcium aluminate content, is commercially available and is used for the manufacturing of COVRA's waste package concrete. The cement content and the water to cement ratio for the manufacturing of concrete, how well the ingredients are mixed, how the concrete is hardened and the accessibility of dissolved species together determine how fast the ingress of dissolved species occurs.

Manufacturing of concrete is well developed and tailored according to an environmental class with its associated degradation processes. COVRA's waste package concrete is designed to sustain the most severe chemical degradation (XA3). For example the strength of concrete should not degrade when concrete is exposed to a solution with sulphate concentrations higher than 3000 mg/l and lower than 6000 mg/l, and higher than 3000 mg dissolved magnesium up to saturation of magnesium (Betonpocket, 2019). All these dissolved concentrations are higher than the concentrations found in clay pore water with the salinity of seawater (see Table 5-1). Moreover, concrete is not exposed directly to a saline solution but to a clay interface. Dissolved sulphate concentrations in clay pore water higher than 12000 mg/l are allowed for clays with hydraulic conductivities less than 10^{-5} m/s (i.e., a permeability less than 10^{-12} m^2). The permeability of a Paleogene clay (parallel to the bedding plane, which is the highest possible permeability) has been estimated to be 4.4×10^{-19} m^2 (see Table 6-1), which is more than six orders of magnitude lower than a clay with a permeability of 10^{-12} m^2. A loss in the strength of COVRA's waste package concrete would therefore not be expected, even if this concrete were to be exposed to a Paleogene clay with the very saline clay pore water in Table 5-1. This expectation of lack in a reduction of strength is confirmed by mechanical tests performed on small concrete specimens (cubes with an edge of 5 cm) that did not show any reduction in strength after 6 years of exposure to a solution as saline as seawater. This was true for concrete that is, in engineering terms, impermeable (COVRA's waste package concrete) as well as for foamed concrete that is quite porous (Vidal et al., 2024). Such expectations are very important for the post-closure safety assessment since high strength concrete usually has a very low permeability.

6.1.3 Low permeability of concrete, leading to diffusion dominated transport

The impacts of low permeabilities on the transport of chemical species through clay were discussed at Text Box 5-1 in Chapter 5. These properties are equally important to understanding the behaviour of the containment functions of the concrete materials used in the EBS. Concrete can have water permeability and diffusion values that are lower than those of clay (see Table 6-1), which allows the tunnel liner to minimize water leaving the clay by ventilation in the operational phase and minimize ingress of water from the clay rock into the GDF in the post-closure phase.

The distribution in size of pores to determine the capillary force has been used to determine the connecting pore throat for the permeability in Table 6-1 and this permeability is smaller than the permeability measured in Boom clay in the vicinity of the URF in Mol.

The low permeability of concrete is important in the post-closure phase since the mass of concrete in the liner, backfill and buffer components of the EBS limits the inflow of the water required for the anaerobic corrosion of metals, and for the formation of zeolites

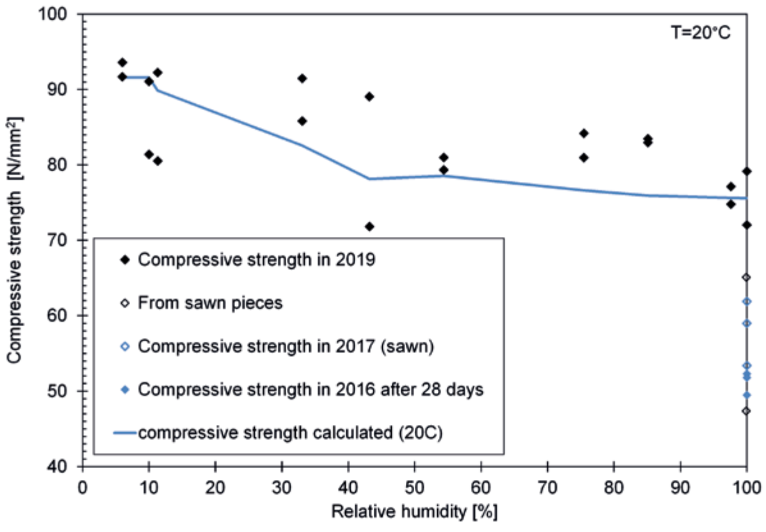


Figure 6-2: Mechanical strength as a function of the relative humidity to which hardened cubical concrete samples with an edge of 5 cm have been exposed, at 20 °C for about 1000 days: content of aggregates 1690 kg/m³ and cement (CEM III/B) 400 kg/m³, water/cement=0.35 (Neeft et al., 2021). Distribution in size of pores see Text-Box 6-1.

Feature	Porosity	Permeability K_{sat} [10^{-20}×m²]	Deff [10^{-11}×m²/s]	Technique	Sample	References
COVRA's waste package concrete	13%	7.3*	0.8**	Drying at 20°C	Cubical edge 5 cm	(Neeft et al., 2021) (Mladenovic et al., 2024) (Blanc et al., 2024)
Foamed concrete (1600 kg m⁻³)	21%^{capillary} 24-25%^{total}	220*	1.6**			
Boom Clay - perpendicular to the bedding plane	35-37	17	7-8	In-situ, HTO diffusion	Belgian URF at 225 m depth	(Levasseur et al., 2021) (Aertsens et al., 2023)
Boom Clay - parallel to the bedding plane	35-37	44	14-19			

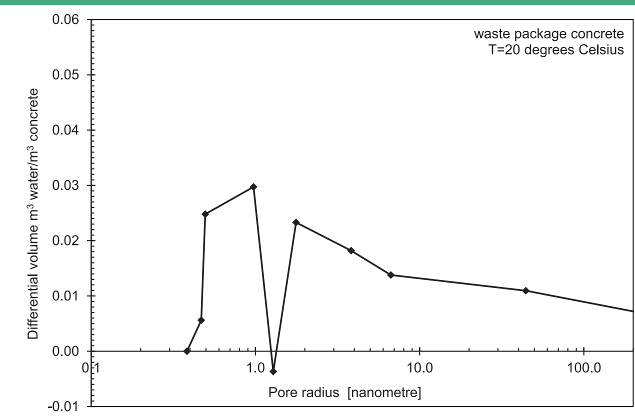
*Permeability values determined by characteristic radius at 20°C (Mladenovic et al., 2024) **Diffusion values at 20°C (Blanc et al., 2024).

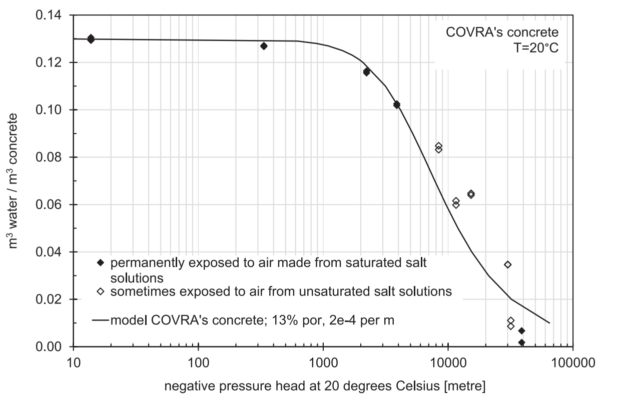
Table 6-1: Some permeability and diffusion values of water for two different types of concrete, compared with the clay host rock: all values are for saturated materials.

Box 6-1 Permeability and diffusivity for concretes in the EBS

As discussed in Box 5-1, permeability is a measure of the ability of a substance to allow gases or liquids to go through it under a pressure gradient. Voids through which moisture can move must be interconnected and of a certain size. Discontinuous pores and pores with narrow entrances retard the flow of moisture. In a mature, well-cured concrete, permeability can be low, even if high porosity exists, but concrete that is more porous tends to be more permeable. The permeability of concrete is determined by the content and type of aggregates, its water to cement ratio and the presence of other additives. Siliceous aggregates usually have such a small porosity that their contribution to the permeability can be neglected. The permeability is therefore determined by the cement paste and the reaction layers between the cement paste and aggregates.

COVRA's waste package concrete is, in engineering terms, impermeable but over the long timescales involved in the evolution of the EBS, even negligible permeabilities become relevant for performance, so more detail is required.
The permeabilities of concrete in Table 6-1 were determined from data on the porosity and characteristic pore radius (Millington and Quirk, 1961). The following two figures show the distribution in size of pores of COVRA's waste package concrete and determination of the characteristic pore radius from the Genuchten parameters (Mladenovic et al., 2024). From these Genuchten parameters, the gas entry pressure is determined that is later used in section 7.1.3.





Genuchten parameters estimated using a best fit (Mladenovic et al, 2024)

Type of concrete	ws vol%	wr vol%	temperature-independent		Technique Sample		References	
			n	m	m⁻¹	Pa⁻¹	m⁻¹	Pa⁻¹
Waste package concrete	13	0	2.0	0.50	2.0×10⁻⁴	2.0×10⁻⁸	1.5×10⁻⁴	1.5×10⁻⁴
Foamed concrete	21	0	1.7	0.41	8.2×10⁻⁴	8.2×10⁻⁸	2.0×10⁻⁴ * 3.0×10⁻⁴	2.0×10⁻⁸

*measurements obtained for foamed concrete at 5°C show a larger variation by which a larger value for α also fits the data. The reciprocal of α is the gas entry pressure (a parameter that defines the threshold capillary pressure necessary to displace the wetting phase) e.g. 2.0×10⁻⁸ Pa⁻¹ for waste package concrete becomes 49 MPa.

and clay minerals during dissolution of the vitrified HLW waste form, once its canister and overpack are breached. It also limits the rate at which the dissolved constituents can be transported away from the interface with the waste form so that the rate at which the concentration of dissolved constituents is smaller than the solubility of the corrosion or alteration product is very low. The porosity and degree of saturation of water in the pores also determine the diffusion rates of dissolved species (Samson and Marchand, 2007). The corrosion rate of steel in concrete has been measured to decrease by two orders of magnitude when the concrete is exposed to a relative humidity of 50% to rather than 90%. This reduction in corrosion rate has been attributed to the reduced amount of water in more and more pores with decreasing saturation degree (Stefanoni et al., 2018).

6.1.4 Self-sealing of fractures

Even if concrete is well engineered, it is possible that fractures will form in the backfill and buffer in the post-closure phase. These fractures reduce the strength and may increase the permeability of concrete. However, it is envisaged that self-sealing will occur on contact between water and cement that has not yet reacted, due to precipitation of material produced in water-cement reactions. This self-sealing process has recently been understood to contribute to the acknowledged durability of Roman concrete (Seymour et al., 2023) and is also observed in COVRA's waste package concrete exposed to a solution with a salinity equivalent to seawater (Vidal et al., 2024).

6.1.5 Chemical containment properties

Alkaline conditions prevailing in concrete pore waters have a pronounced limiting effect on the solubility of radionuclides present in the wastes. An alkaline near-field environment in a cement dominated EBS was recognised early as potentially beneficial to containment in many design concepts for geological disposal of wastes in other national programmes. For example, the estimated low solubility of thorium and uranium under alkaline reducing conditions of 3×10^{-9} mol/l led to chemical containment becoming a central part of early ILW disposal concepts in the UK (Chapman and Flowers, 1986). Alkaline reducing conditions are immediately present when concrete is manufactured with CEM III/B (see section 6.1.5.2).

Information on how naturally occurring trace elements such as U and Th are bound in concrete can support our understanding of its chemical containment properties. As explained in Chapter 3, the natural radioactive content of building materials makes a significant contribution to the public radiological exposure and is therefore monitored in the Netherlands. This exposure is determined by the concentration of radionuclides in concrete and assumed exposure pathways. Based on measured values, the average amounts of radioactivity in concrete are 24 Bq/kg for Ra-226 (daughter of uranium-238: 2 mg U/kg concrete); 20 Bq/kg for Th-232, i.e., 5 mg ^{232}Th/kg concrete); 227 Bq/kg concrete for K-40, i.e., 1 mg ^{40}K/kg concrete (Smetsers and Bekhuis, 2021). The origin of these radionuclides is primarily the cement used to manufacture the concrete. The low solubilities of U and Th are indicated by their contents being many times greater in the solid phases of concrete than in pore waters. More detailed searches and studies are needed on how trace elements such as U and Th are bound in the solid phases of the concrete. Chemical containment of radionuclides by concrete is therefore not yet included in the COPERA post-closure safety assessment. The following two sections highlight some studies in order to show how solid phases in concrete can contribute to containment, as well as how concrete might be expected to evolve in the post-closure phase.

6.1.5.1 Ion exchange

Like clay, concrete can also be an ion exchanger, but its exchange capacity is pH dependent. Ion exchange capacity can be an advantage for retardation of radionuclides but can also enhance metallic corrosion, especially if the charge of the C-S-H mineral has become negative (see Figure 6-1). The nature and type of ion exchange for cement minerals depends on the calcium concentration in concrete pore water (Pointeau et al., 2006), which depends on the pH (Berner, 1992; van Eijk and Brouwers, 2000; Vehmas et al., 2019). Figure 6-1 shows the pH stages at which there is a positive and negative surface potential on the CSH hydration phase in cement paste, which is positively charged between pH 11.7 and 13.0, with a maximum around 12.6 (Pointeau et al., 2006). The uptake of dissolved anions has been measured to increase with increasing charge of the anion or anionic complex and with atomic number (Pointeau et al., 2008). Uptake of radionuclides by the cementitious materials is not yet considered in the post-closure safety assessment, as this will require more information on the evolution of the pH of concrete.

Below a pH of 11.7, the uptake of dissolved cations is expected as the charge of the C-S-H mineral becomes negative. Positively charged iron is released in the corrosion process of carbon steel and this enhanced uptake of dissolved iron by concrete is expected to result in higher corrosion rates. In the post-closure safety assessment, this increase is implicitly included in the post-closure phase by making a conservative assumption of the corrosion rate and thus the period after which steel is completely corroded.

6.1.5.2 Reducing conditions

Evolution of the HLW package and the concrete tunnel liner are partly controlled by the interfacial reactions between concrete and steel. Aerobic corrosion rates of steel are higher than anaerobic corrosion rates (Crossland, 2005) since iron solubility is higher at oxidizing conditions. Concretes made from pure Ordinary Portland Cement (OPC) and OPC blended with fly ash are slightly oxidizing, since they lack redox sensitive species. Concrete with blast furnace slag (BFS), which is a by-product from steel production, contains pyrite (iron sulphide) and is therefore reducing. Experiments to quantify the reducing capacity have recently been carried out for COVRA's waste package mortar, which is made with OPC blended with BFS. Concrete cubical specimens with an edge of 5 cm were exposed to atmospheric air at different relative humidities. The oxygen penetration front shown in Figure 6-3 is indicated by that part of the concrete that has become colourless; the part that is dark blue still contains iron sulphide. These oxygen penetration fronts were predicted with a model incorporating water and oxygen diffusion, the degree of saturation and reaction rates determined by the size of the iron sulphide particles (Blanc et al., 2024).

The presence of iron sulphide has no impact on the corrosion of steel if the steel has no etching or abrasion, as explained in Neeft et al. (2022). The rest potential of carbon steel, the potential at which passive (slow, uniform and predictable) corrosion takes place, has been measured to be established immediately for steel embedded in a grout mixture made with a BFS-blended cement. For carbon

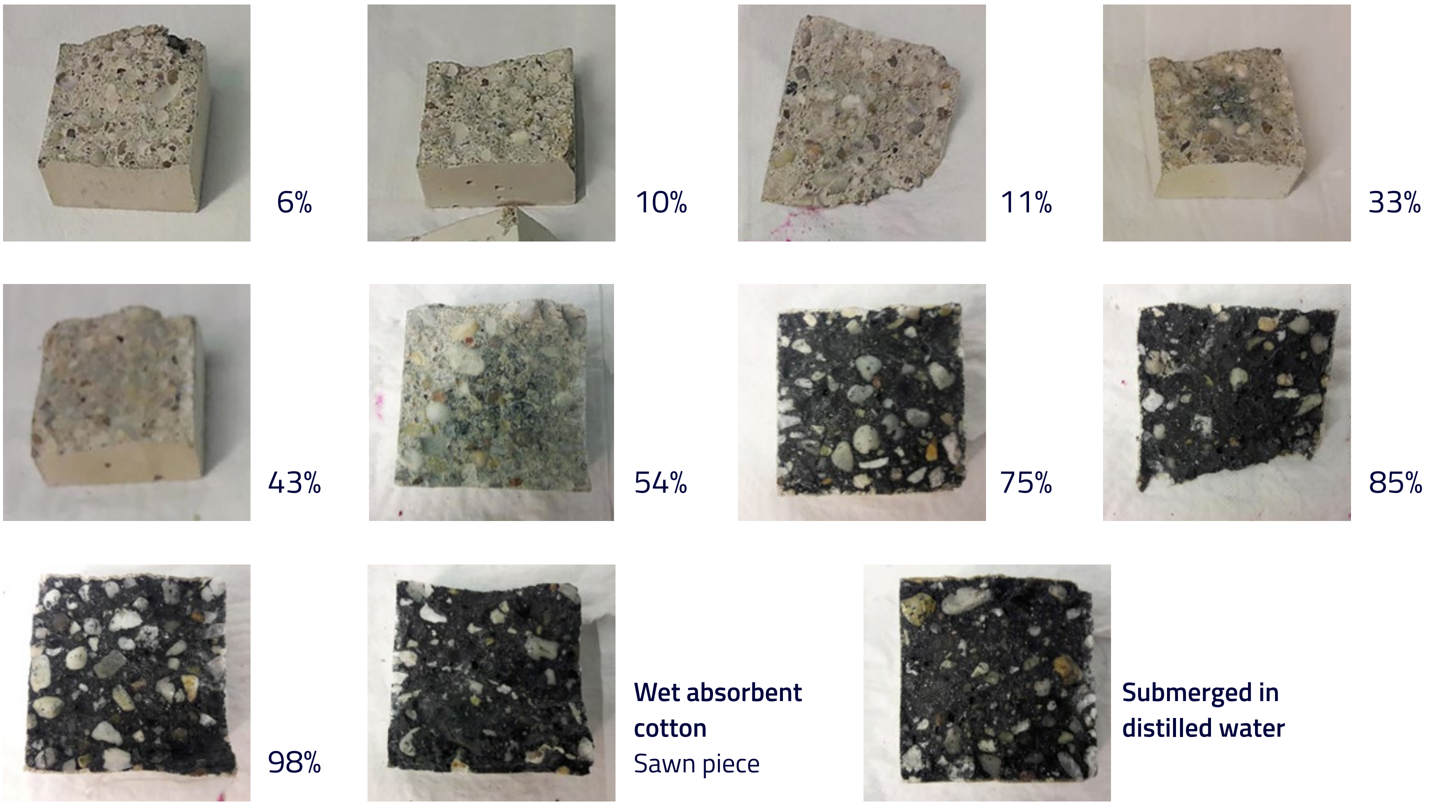


Figure 6-3: Extent of oxidation of cubical specimens of COVRA's waste package mortar as a function of the relative humidity at 20 °C (Mladenovic et al., 2024; Neeft et al., 2021).

steel surrounded by mixtures of OPC/Pulverized Fly Ash (PFA) and OPC/lime, it may take some time before entrapped oxygen is sufficiently consumed to establish the rest potential of steel under anaerobic, alkaline conditions. Passive corrosion requires the presence of magnetite, the type of iron oxide that is thermodynamically expected to be present under anaerobic, alkaline conditions. This iron oxide has been detected by Raman spectroscopy on the surface of carbon steel embedded in BFS-blended cement, but not within the same experimental period on surfaces of steel that have been embedded in mixtures of OPC/PFA and OPC/lime (Naish et al., 1991). These experimental results imply that anaerobic corrosion of the carbon steel overpack may be assumed to occur immediately if the concrete buffer is made with a BFS-blended cement: i.e., there is no aerobic corrosion period in the post-closure phase.

Iron, like uranium, is a redox sensitive element. The low solubilities of redox sensitive elements in concrete under reducing conditions minimize the potential inflow of redox sensitive radionuclides from the concrete into the clay rock. This feature is not yet used in the post-closure safety assessment.

6.2 The waste materials and their role in the EBS

In OPERA, the waste inventory was grouped into families, each of which is conditioned for disposal in a specific manner. Families are groups of radioactive wastes from the same origin which are similar in nature, have identical or closely related conditioning characteristics, and belong to the same category of the current waste classification scheme. In post-closure safety assessments, this grouping into families facilitates calculation of the radionuclide release rates from the waste forms (the so called 'source term'), but is necessarily a simplification. Criteria for grouping the wastes include the available information on their physical and chemical characteristics, radionuclide content, degradation mechanisms and the potential contribution to the source term. This means that small volumes of different wastes have been grouped into one family (e.g., compacted waste in 200 litre drums) in cases where the impact on the source term was limited (Verhoef et al., 2016). Figure 6-4 shows the waste families (Verhoef et al., 2016) with quantities taken from the most recent Dutch inventory (Burggraaff et al., 2022). The radionuclide inventory for each waste package in Appendix 6 is the same as that published in the OPERA Safety case.

The radionuclides in the waste forms, the long-term behaviour of the solid waste forms and the transport mechanisms of the radionuclides in the other engineered barriers, together determine which radionuclides can enter the clay host rock and the timescales on which this can occur. The long-term behaviour of the system, in particular how the waste forms react with, and dissolve in, pore waters in the EBS, is influenced by physical and chemical processes that can attenuate and delay releases, thus limiting release rates and also spreading releases over time.

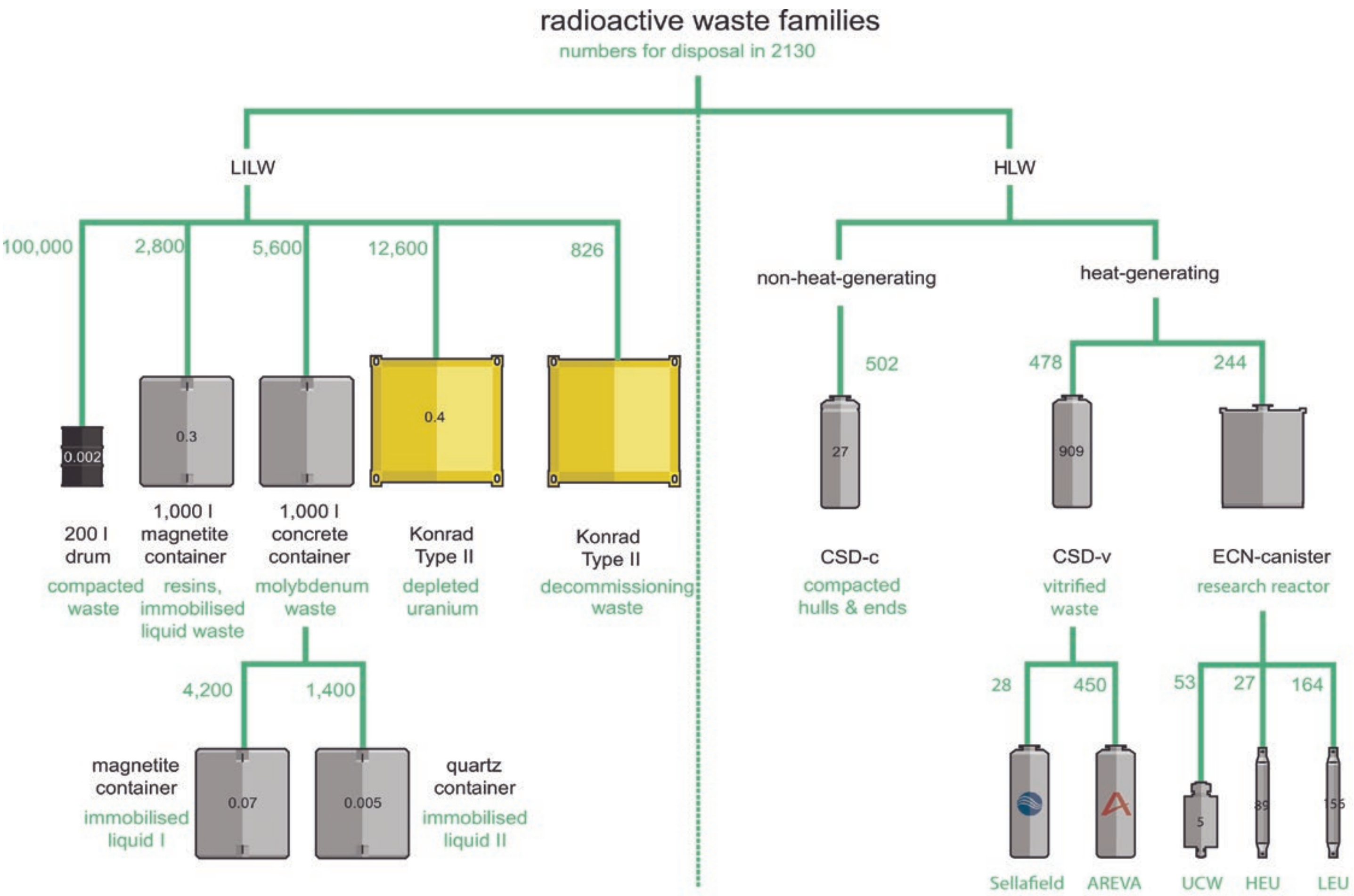


Figure 6-4: The Waste Scenario 1 inventory broken into waste families, showing the expected number of packages in green and the activity of each package in TBq in 2130 in black (assuming 130 years decay). LILW is to be disposed of as shown. HLW canisters will be overpacked as discussed in Chapter 4. Numbers of packages are updated from the figure in the OPERA Safety case

6.2.1 LILW

LILW has been grouped into the five waste families shown in Figure 6-4:

- 200 litre drums with a large variety of waste forms but low content in radionuclides;
- 1000 litre concrete containers with a homogenized cementitious waste form of high alkaline waste from production of medical isotopes;
- 1000 litre concrete containers with a homogenized cementitious waste with resins associated with the cleaning of water from nuclear plants;
- Konrad container with a homogenized cementitious waste with depleted uranium;
- Konrad container with dismantling waste.

6.2.1.1 200 litre drums

Solid waste in 90 litre containers is collected from some two hundred organisations, ranging from nuclear power plants and research establishments to numerous types of industry and hospitals. The waste includes materials from dismantling of nuclear and other installations and consists mainly of contaminated materials, such as organic cellulose-based materials (cloth, paper, tissue), sludges, metals (steel, aluminium), plastics (halogenated, non-halogenated), glass, concrete, inorganic adsorption material, salts etc. On receipt at COVRA, the 90 litre containers are perforated and compacted. The resulting pucks are embedded in concrete in 200 litre containers, as shown in Figure 6-5. This compacted waste is the second largest waste family by volume.

COVRA's waste package mortar has been studied in EURAD-1 ACED and MAGIC. Figure 6-2 and Figure 6-3 show the evolved mechanical strength and its reducing capacity. The values for permeability and diffusion coefficient are shown in Table 6-1.

Assumptions for current and future post-closure safety assessments

The variability of the chemical and physical characteristics of the waste forms are very large in this waste family. In OPERA, a simple instant release model for radionuclides was assumed for the post-closure safety assessment (Schröder et al., 2017b).
The amount of gases that are estimated to be produced in the GDF is however large (Filby et al. (2016), and instant release of radionuclides is associated with the instant release of these gases.

In future post-closure assessments, the geometry of the multi-barrier system with its low permeability concrete and clay media will be used to determine the range in possible alteration rates of the waste form with their associated release rate for radionuclides and gas generation rate.

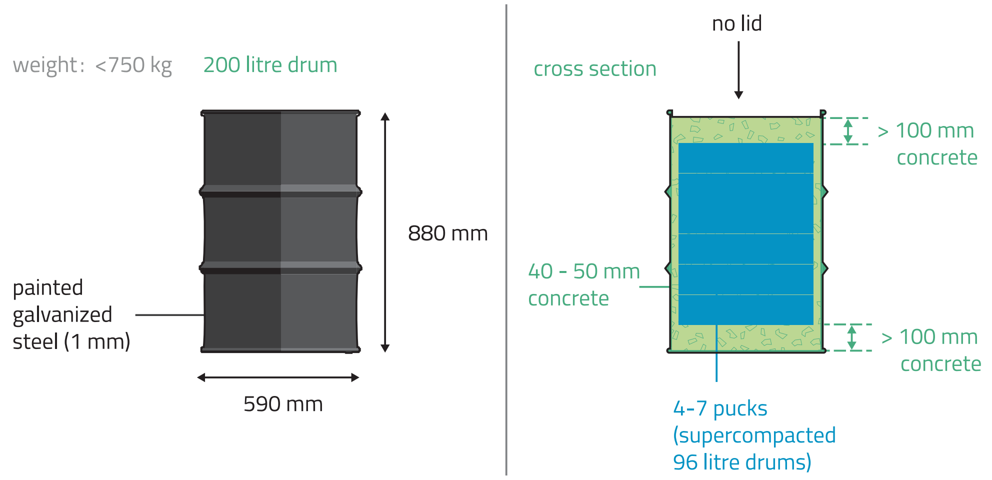


Figure 6-5: Schematic of a 200 litre drum with compacted pucks of LILW. Figure 6-3-10 in the OPERA Safety case

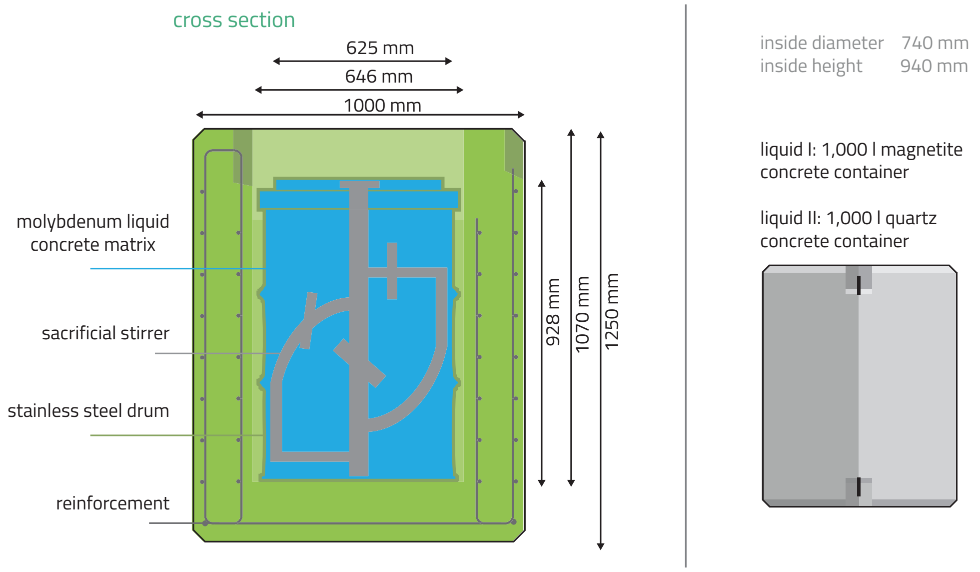


Figure 6-6: Schematic of 1000 litre concrete container. Figure 6-3-11 in the OPERA Safety case.

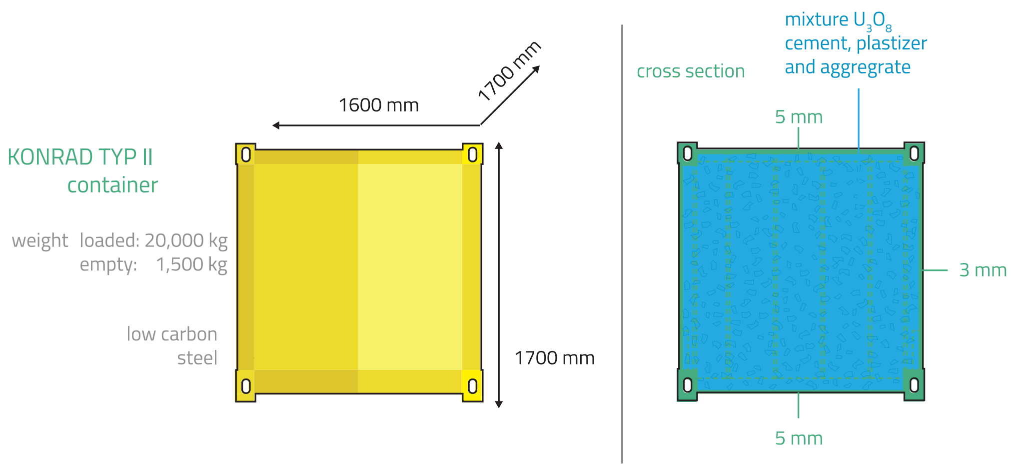


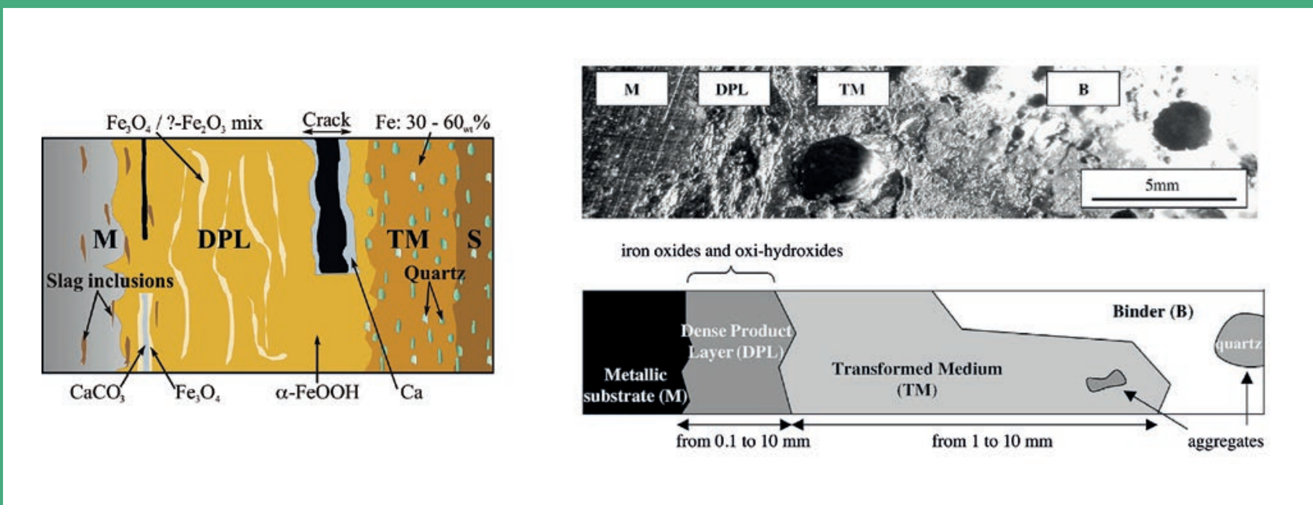
Figure 6-7: Sachematic of Konrad type II container. Figure 6-3-5 in the OPERA Safety case.

Box 6-2: Long-term behaviour of waste forms in contact with water and the porous barrier materials

Interfacial reactions between waste materials and other engineered barriers determine the alteration rate of the waste form. These reactions are complex and coupled. Post-closure safety assessments, such as that performed in OPERA, often conservatively simplify these by using experimental results in which waste forms are exposed only to solutions representing pore waters (i.e., no other EBS materials present), with the water often in large volumes compared to the waste materials. Such experiments tend to overestimate mobilisation rates as they are not representative of post-closure conditions where waste forms interface other solid, porous media and only limited quantities of pore water are available, as discussed in this Text Box.

The long-term behaviour of a waste form depends upon the rates of dissolution and dissipation of its constituents in solution, such as iron, silicon, calcium etc. The interaction between the waste form and pore water results in the formation of solid interface phases: these Product Layers are composed of minerals that are hydrated and/or oxidized constituents of the waste form. This occurs with glass (Conradt et al., 1986; Ferrand et al., 2023), metals such as steel (Mibus et al., 2015) and Zircaloy (Necib et al., 2018a). Product Layers have a so-called passivation capacity; they can attenuate the release of radionuclides from the solid waste form by acting as a diffusion barrier and also by uptake mechanisms such as sorption. A local equilibrium between this solid surface phase and a dissolved phase will develop, depending upon the solubility of the solid phase, which can be very small. The dissolution rate of the solid waste form is then controlled by the diffusion rate of dissolved species away from its surface. The dissolution rate becomes larger if flowing water is in contact with the waste, as this transports dissolved substances away from the interface, thereby reducing the concentration in the water in contact with the waste form. Local equilibrium must be maintained by further dissolution of the solid phase.

In the actual geometry of the multibarrier system, the waste form interfaces with an engineered barrier that is a porous medium, rather than directly with a liquid. Diffusion rates for dissolved species in water alone are larger than diffusion rates in a porous medium, where the dissolved species need to find their way through connected pores and thereby have an increased transport length. The mobilisation rate of dissolved radionuclides from a waste form can therefore be smaller in a porous medium than in water. However, the dissolved constituents of the waste form are usually charged species and the surfaces of minerals in the porous medium may also have a charge. Sorption occurs if this charge is opposite to that of the dissolved species. This reduces the concentration of dissolved species , so that the dissolution rate of the solids can also be larger in a porous medium if sorption occurs. The larger anaerobic corrosion rate of steel in clay rather than in water (see (Neeft et al., 2022) is an example of the impact of sorption of dissolved iron species by clay minerals that have a negative surface charge.
This iron sorption may not only be important in clay.
For concrete pore water at pH below 11.7, the surface charge of C-S-H minerals in concrete changes from positive to negative (see Figure 6-1). The potential features and processes of each waste form interacting with pore water and with the internal surfaces of minerals in the porous medium therefore needs to be known, in order to assess its long-term behaviour.



Main corrosion pattern found for steel in soils (clays) and cementitious materials. Pattern observed in steel-soil (clay) interface that is several hundred years old (left) (Neff et al., 2004) and a steel-binder (cement) interface that is 350 years old (right) (Chitty et al., 2005).

In addition to a Product Layer, another solid layer may develop at interfaces between two solids by reaction with the dissolved constituents: a so-called Transformed Medium (TM). Archaeological analogues of steel embedded in clay soils (Neff et al., 2004) and in concrete (Chitty et al., 2005) are examples of porous media in which TMs have been found. The figure below shows the main corrosion pattern of the metal (M), a steel or iron, the Dense Product Layer (DPL), which comprises oxidized and hydrated products from the metal, and the TM.

A TM (iron-affected clay or iron-affected concrete) may have properties that differ from the unaffected properties of the solid porous medium, such as strength, distribution in size of pores and swelling or self-healing potential. An example is the alteration of swelling clay minerals in bentonite (used as a buffer material in hard rock disposal concepts) into non-swelling sheet silicates when affected by iron corrosion products (Savage, 2014). The understanding of these processes is used to define the necessary thickness of the buffer for the post-closure phase in which sufficient non-iron affected bentonite will be left. The properties of a TM formed in concrete are less studied. For iron-affected concrete, currently the only available information is on a layer described as 'loosely bound material' (Atkins et al., 1991) on a more than 30-year-old concrete adjacent to steel.

These processes are not yet taken directly into account in the safety analyses, but the thickness of the concrete buffer that retains a high strength is assumed to decrease in the post-closure phase.

6.2.1.2 1000 litre concrete containers

The third largest volume of waste arises from the production of medical isotopes. It includes processed liquid molybdenum waste from irradiated uranium targets (see section 6.2.3). The highly alkaline waste stream is mixed with a cementitious mortar in 200 litre drums using a sacrificial stirrer (see Figure 6-6). These drums are packed in a 1000 litre concrete container. COVRA waste package mortar is used to fill the void volume around the 200 litre drums inside the 1000 litre concrete container. This container, together with the waste package mortar, provides radiation shielding during storage and emplacement in the GDF.

The aggregate used in manufacturing the concrete containers is magnetite (see Figure 6-4: there are 4200 containers of molybdenum waste and 1000 with spent ion exchange resins). Magnetite has a higher density than siliceous aggregates, so a smaller concrete thickness is needed for the waste package to provide the required level of shielding against gamma radiation compared to containers made with siliceous aggregates, which also reduces the storage volume. The possibility of using aggregates with a higher density to obtain more shielding for a given thickness of material has also been investigated for the HLW disposal package (see section 6.3).

The fourth largest volume of waste arises from the treatment of nuclear reactor water with ion exchange resins. This waste is also processed with cementitious mortar in 200 litre drums, which are also packaged in 1000 litre concrete containers.

Assumptions (to be) made for the post-closure safety assessment

In OPERA, instant release of radionuclides from the 1000 litre containers at the time of closure of the GDF was assumed for the safety assessment (Schröder et al., 2017b). The packages are conservatively modelled as a cementitious fluid with a homogenous distribution of radionuclides. The chemical and physical characteristics of the cementitious waste forms can be determined through experimental non-radioactive investigations since both the waste forms for molybdenum waste and spent-ion exchange resin have a homogeneous distribution of radionuclides in the waste form. For these resins, some studies have been performed in the EC's seventh framework Carbon-14 Source Term project: e.g., Capouet et al. (2018). In a future post-closure safety assessment, this homogenisation allows a more gradual release of radionuclides to be used.

6.2.1.3 Konrad type II containers

Konrad containers are manufactured from sheet steel with a thickness of at least 3 mm. The maximum in weight is 20,000 kg (Lange et al., 1992). Two types of waste are envisaged to be disposed of in Konrad type II containers (see Figure 6-7): dismantling waste and depleted uranium. Dismantling waste, which is the smallest category of waste by volume, consists of metallic parts, mainly steel, and concrete. The two main types of steel are carbon steel and stainless steel. In many nuclear engineering applications, these steels are used together, i.e., carbon steel for mechanical strength and stainless steel for high chemical resistance against corrosion. The waste characteristics for dismantling waste were not described in OPERA and these characteristics still remain to be determined. The largest LILW family by volume is depleted uranium, generated by URENCO in the uranium enrichment process. The uranium-fluoride tails that are not economically feasible for re-enrichment become waste. These tails are processed in France into U_3O_8 for safe storage. U_3O_8, which arrives in mm sized granules at COVRA's premises may contain some traces of UF_6. Conditioning with a cementitious matrix provides sufficient calcium for the uranium-fluoride to react into stable insoluble minerals (Kienzler et al., 2013). The granules were envisaged to be mixed with a cement paste and poured into Konrad containers (see Figure 6-7) in OPERA (Verhoef et al., 2014). Further study is needed to assess whether fluoride-traces are likely to be present after a storage period of at least 100 years. If they are not expected to be present, the necessity for additional processing is reduced..

Assumptions made for the post-closure safety assessment

Only the Konrad containers with depleted uranium were included in the OPERA post-closure safety assessment. Uranium release into the clay was assumed to begin at 1,500 years after closure. This rate was solubility-limited (U_3O_8 is the most insoluble form of uranium oxide): the solubility of uranium in the EBS was assumed to be 10^{-5} mol U/l (Schröder et al., 2017b). This solubility limit in the EBS is smaller than the assumed solubility of uranium in clay pore water of 10^{-4} mol U/l (Schröder et al., 2017c). In future assessments, a more realistic approach will be used.

Approach for a future post-closure safety assessment

The amount of water needed to dissolve the >12 tonnes of U_3O_8 in each Konrad container is more than 4×10^9 kg[15], using a solubility of 10^{-5} mol U/l. To provide a realistic source term for the mobilisation of uranium will require estimation of the ranges of water inflows in the multibarrier system with the low permeable media (clay and concrete).

In addition, the assumed solubility limits should be substantiated. The solubility of 10^{-9} mol U/l at alkaline reducing conditions (Chapman and Flowers, 1986) is based on based experimental results and is four orders in magnitude smaller than the 10^{-5} U mol/l used in OPERA. Experimental results should be given a higher priority for deriving the best estimate in the post-closure safety assessment, also taking into account how the experimental conditions may differ from those in the multibarrier system. Other information can be obtained from measurements on clays. The minimum measured uranium concentration in clay pore water in Mol is 0.200 µg/l and the maximum 3.5 µg/l (De Craen et al., 2004): i.e., a minimum of 8.4×10^{-10} mol U/l and a maximum of 1.5×10^{-8} mol U/l. The maximum is almost three orders of magnitude smaller than the default value considered in OPERA.

6.2.2 Spent research reactor fuel

6.2.2.1 Description of the waste

Spent research reactor fuel (SRRF) mainly arises from the production of medical isotopes in Petten. Highly Enriched Uranium (HEU: 93% ^{235}U) fuel was used in the Netherlands until 2006.The HEU fuel assemblies contain 23 vertically arranged, parallel, curved fuel plates with a height of 625 mm. Each HEU plate consists of a layer of aluminium-uranium-alloy with a thickness of 0.51 mm, encapsulated in aluminium cladding with a thickness of 0.38 mm for the inner plates, and 0.57 mm for the outer plates.

Since 2006, only Low Enriched Uranium (LEU: 19.75% ^{235}U) fuel has been used in Dutch research reactors. The fuel is uranium-silicide and has thicker LEU elements: 0.76 mm. The larger thickness of fuel plates for LEU results in fewer fuel plates per fuel element. Figure 6-8 shows cross sections of the assembled HEU and LEU fuel elements. The neutron absorbers (^{10}B in an aluminium matrix for HEU and cadmium wires for LEU) are clearly visible. For each HEU fuel assembly, two flat side boron plates, together containing 1000 mg ^{10}B, were used. The length of both the HEU and LEU fuel inside the fuel assembly (see fuel element in Figure 6-8) is 600 mm (Ahlf and Zurita, 1993; Dodd et al., 2000; NRG, 2012; Thijssen, 2006).

The fuel elements are transported to COVRA in special containers. The number of fuel elements in a transport container is limited by the heat generation. The maximum thermal output to comply with the current license for the transport container (MTR-2) is 25 W per fully irradiated element (NRG, 2012), i.e., 825 W, since the MTR-2 is able to contain 33 elements. There is a difference in thermal output between HEU and LEU; after 130 years the output per element is 0.76 W for HEU (NRG, 2005) and 1.23 W for LEU (NRG, 2012). Figure 6-9 shows that the heat output after 130 years is orders of magnitude smaller than at the start of the storage period.

Each fuel element is removed from the transport container and placed in a borated stainless steel basket inside an ECN canister to store the waste at COVRA's premises. The boron and also the spacing between the fuel elements are important to assess the criticality. Spacers separate fuel elements in the basket by 11 cm.

The lid of the stainless canister that contains the basket and SRRF elements is made at COVRA's premises in HABOG. A steel ring is screwed onto the outer circumferential side of the container in order to prevent mechanical damage to the weld during lifting. A gas system is positioned just below the mushroom of the lid, designed to remove air in the canister and fill the canister with helium. The canister is filled with helium in order to facilitate leakage detection during storage in the double sided wells in the HABOG surface facility.

6.2.2.2 Assumptions made in the OPERA post-closure safety assessment

Three features of the SRRF waste form that can determine the release rate of radionuclides from the fuel are discussed below: criticality, surface area of the cladding and alteration rate of aluminium. These three features were not addressed in the post-closure safety assessment in OPERA, which used a simple, conservative, instant release model for the radionuclides in the fuel, starting after periods of 1,000 years, 35,000 years or 70,000 years (Schröder et al., 2017b). These values are based on the period in which pH of the concrete pore water which determines the corrosion rate of the carbon steel overpack is 12.5 i.e. portlandite (a cementitious mineral see Figure 6-1) is present. This period is estimated to last from 1,000 years until 80,000 years (Kursten and Druyts, 2015). The alteration rate of the SRRF is highly dependent on the pH. This assumption of a failure time when the pH of

15. The density of the U_3O_8 granules is 2664 kg/m³ (Verhoef et al., 2014). Each Konrad container has a volume is 4.6 m³, thus holding more than 12 ton U_3O_8 (>4×10^4 mol U). The mol of U divided by the solubility of 10^{-5} U mol/l gives the figure of 4×10^9 kg water.

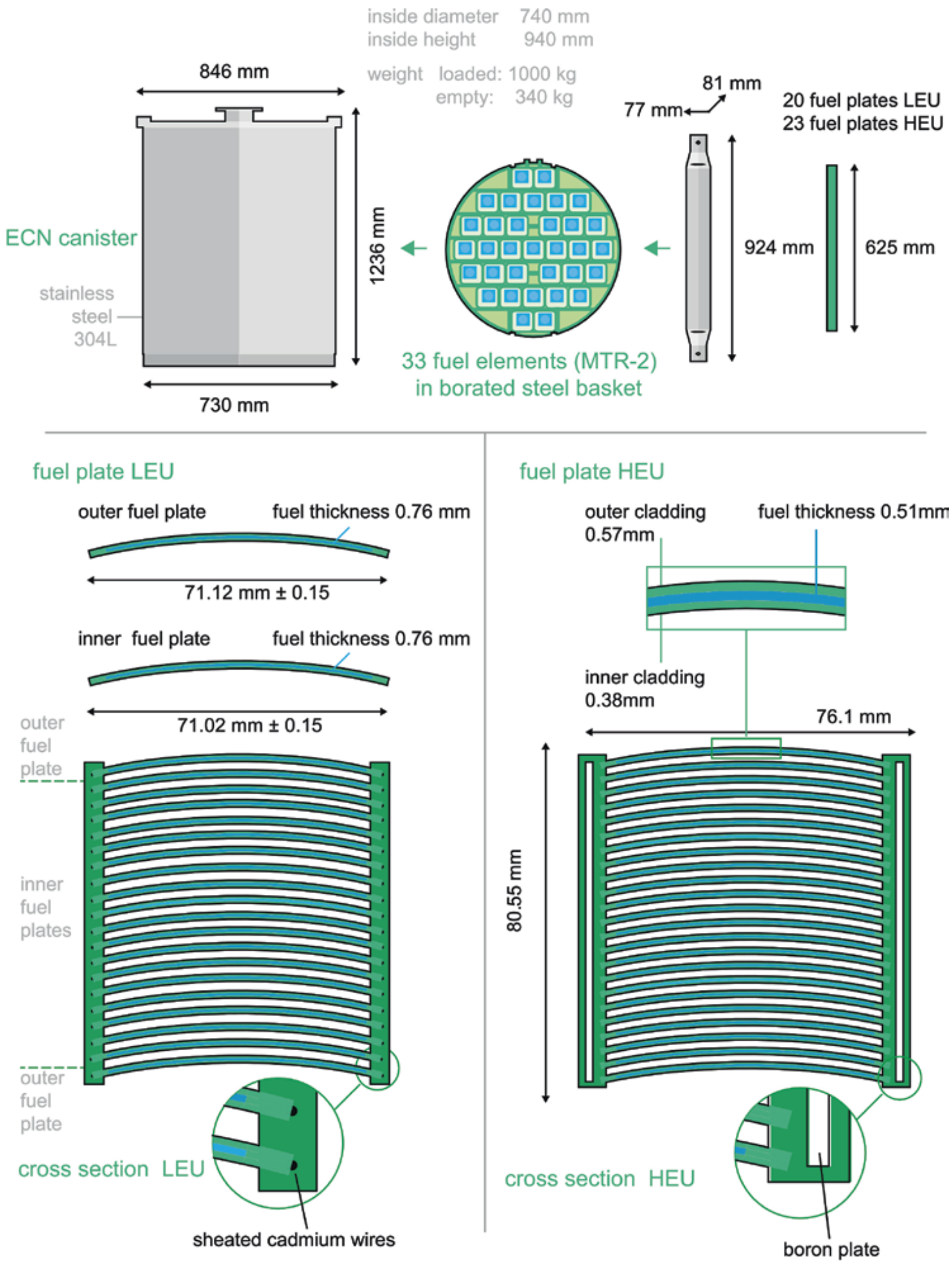


Figure 6-8: Schematic of spent research reactor fuel and the ECN canister. The canister is to be put in a HLW disposal package. Figure adapted from the OPERA Safety case by inclusion of the boron plate for HEU.

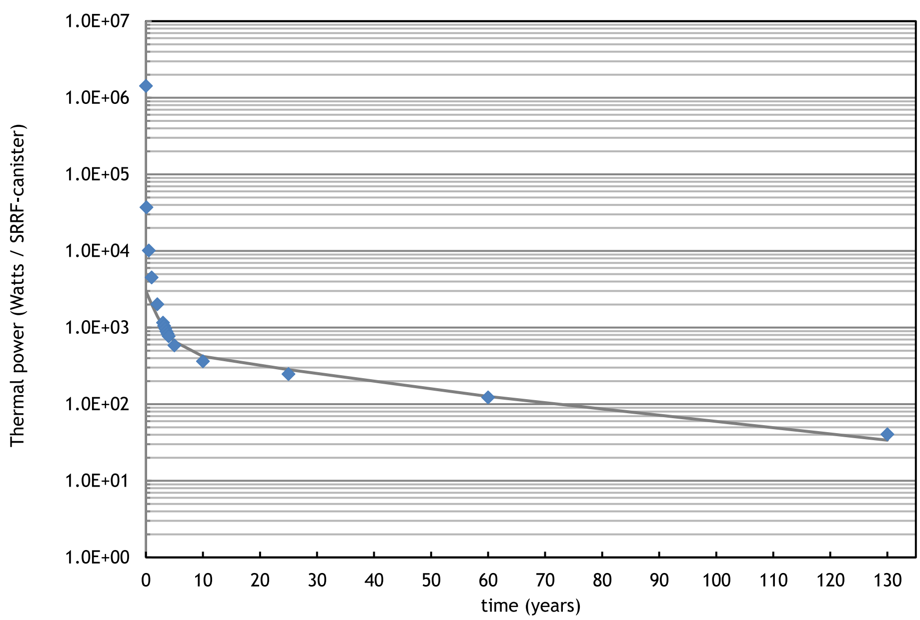


Figure 6-9: Evolution of the thermal power of SRRF (LEU) canister with 33 fuel elements (NRG, 2012)

concrete pore water is 12.5 allows to use the same alteration rate of any waste form at different times. Concrete in the multibarrier system was assumed to have no mechanical strength in OPERA, the carbon steel overpack alone was assumed to sustain the mechanical load in the underground. If the strength of the concrete buffer would be included, the formation of a crack in the carbon steel overpack and stainless steel canister would be caused by a loss in thickness of two materials with strength: carbon steel (from the overpack) and concrete (e.g. from the buffer).

6.2.2.3 Criticality of SRRF

Spent fuel contains a build-up of fission products, which act as poisons (neutron absorbers) that minimize the neutrons available for fission of uranium or plutonium, making the fuel increasingly inefficient. With time after discharge of the fuel from the reactor, these fission products decay and are no longer available. If the fuel is disposed of in an environment where plausible scenarios can be envisaged in which sufficient moderator such as water becomes present, enough neutrons may again be available to sustain the fission process. This generates heat at a rate that is difficult to predict. In addition, any process (such as deformation or dissolution and reprecipitation) that leads to a major change in the disposition or geometry of the fuel, causing a change in concentration, can lead to criticality. Potential criticality is therefore an important feature to address for the post-closure phase.

Criticality thus depends on the enrichment in fissile uranium or plutonium, the amount and geometry of fissile material access to a moderator and removal of generated poisons. The possibility of spontaneous fission reactions starting up is illustrated by the 2.1 billion years old natural fission reactor found in 1972 in a uranium ore body at Oklo in Gabon: e.g., Trotignon (2004). A fissile uranium enrichment of about 3.8% is representative for nuclear power fuel (AREVA, 2007) and the natural fissile enrichment in the Oklo ore 2 billion years ago is estimated to have been 3.5% (Bentridi et al., 2011) - much higher than the natural enrichment remaining in ore bodies today, owing to radioactive decay. With sufficient circulation of clean water in the surrounding sandy formations to allow moderation and removal of poisons this allowed a fission reaction to take place. Fission products (oxides) together with sand grains then formed clay minerals, and a clayey envelope minimized the circulation of fresh fluid and stopped the fission process (Bentridi et al., 2011). In our multibarrier system, the clay formation and the EBS with cementitious materials will minimize any circulation of fresh fluid in the post-closure phase.

However, the enrichments mentioned above are much smaller than those in nuclear research fuel, 93% for HEU and 19.75% for LEU, so that further precautions against re-criticality may need to be taken. The amount and geometry of research fuel per disposal package was therefore proposed to be reduced to a diameter of 13 cm in the second research programme, CORA, to ensure sub-criticality (Dodd et al., 2000). This implies that repackaging of the stored SRRF would be necessary, since the stored waste has a diameter of 74 cm (see Figure 6-8). During CORA, the geometry for the criticality calculations was simplified to a sphere: i.e., taking no credit for the spacers between the fuel elements that are present during storage of waste. The spacing between fuel elements has been included in COPERA. In storage, this spacing is 11 cm and 33 fuel elements (N=33) are present in a single canister. In addition, the impact of the second feature to minimize criticality, i.e., the boron content in the steel basket, has been included.

For assessing the long-term criticality behaviour, it is not sufficient to consider only the original geometry of the waste since the mechanical load of the overburden is eventually assumed to be transferred to the waste form, especially if is assumed that the concrete, the carbon steel overpack and borated steel basket have no strength. The impact of the higher mechanical load is a reduction in spacing between the fuel elements. The boron incorporated in the steel captures sufficient neutrons to reduce the availability for fissile uranium and plutonium. But this boron is expected to be released during the corrosion process and is therefore not assumed to prevent criticality.

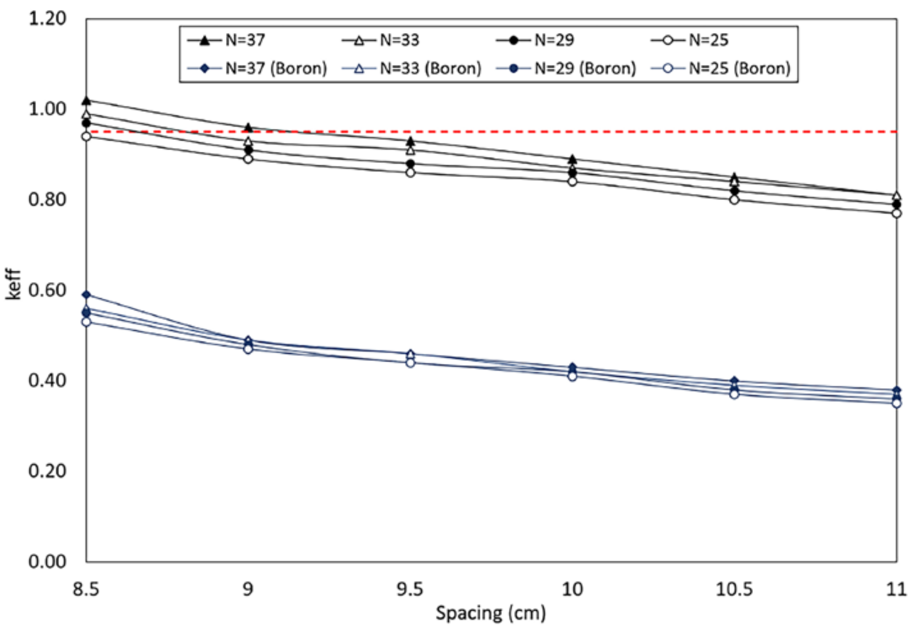


Figure 6-10: The effective multiplication factor (keff) as a function of spacing between the fuel elements (Koets et al., 2022).

Figure 6-10 shows the calculated effective multiplication factor (Koets et al., 2022). Criticality occurs if this factor (k_{eff}) becomes larger than 1. Criticality is calculated be impossible with the original geometry of the fuel elements, even without boron, but feasible if the elements become closer. The criticality further reduces if the boron is taken into consideration.

A solution considered in COPERA is elimination of the void volume inside the canister in order to prevent the spacing between the fuel elements being eliminated if the mechanical strength of the materials used for the disposal package is lost in the post-closure phase.

6.2.2.4 Aluminium alteration rates in SRRF

OPERA collected available corrosion rates of fuel materials, mainly for aluminium under alkaline conditions, since the data for the fuels themselves - UAl_x (HEU) and U_3Si_2 (LEU) - are scarce. For experiments on alteration mechanisms and rates, metals are usually cleaned and sometimes etched in order to remove as far as possible any oxide films, which are usually present on any metal surfaces. When the clean metal is exposed to water, it may take some time before an oxide film is formed that is representative for the solution chosen in the experimental set-up. These oxide films passivate the corrosion process so that the dissolution rate of the metal oxide is equal to its generation rate. The corrosion rate then becomes constant. It may take several days, months or even years before a steady state is achieved. The corrosion rates obtained from literature depend on how long an experiment has been running (Deissmann et al., 2016a). For example, for the first 1000 hours a corrosion rate for aluminium of 20 mm per year was measured, but the subsequent steady state rate was 10 μm per year (Fujisawa et al., 1997). Further details are provided in Text Box 6-4.

6.2.2.5 Surface area of fuel cladding

The surface area of aluminium will determine the associated hydrogen production from a canister, if it is assumed that access to water is the same as that used in the experiments from which rate data were derived. The surface area of the aluminium cladding in the HEU canister is about 95,000 cm^2, assuming both sides of each fuel plate are exposed as in Figure 6-8. The dimensions and numbers of LEU fuel plates in a single fuel element are smaller than for HEU fuel, with the surface area being 77,000 cm^2 per LEU canister.

6.2.3 Uranium filters

The COVRA inventory also includes uranium filters from the production of medical isotopes from irradiated HEU targets. As with spent fuel, these will be packaged for disposal in ECN containers. These are assumed to have the same characteristics as SRRF and, considering the relatively small number of packages expected, uranium collection filters (UCW in Figure 6-4) were not considered in the inventory for the calculation of the source term within OPERA. The current assumption is that the uranium collection filters have the same characteristics as SRRF, which is expected to be conservative.

6.2.4 Vitrified HLW

6.2.4.1 Description of the waste

The vitrified HLW is manufactured from the residues of reprocessing of spent fuel from which uranium and plutonium have been extracted. These residues are blended with a melted glass frit and poured into a stainless-steel container to solidify as glass. This waste processing ensures that the radionuclides are homogeneously distributed in the borosilicate glass matrix, which contains residual traces of plutonium and uranium, other actinides that have not been extracted, such as americium, and fission products. These vitrified waste products have been produced in Sellafield (UK) and are still being produced in La Hague (France). The largest amount in the Dutch inventory comes from France. Therefore, the French abbreviation for this waste product is used: Conteneur Standard de Déchets vitrifiés, CSD-v. Figure 6-11 shows the dimensions of the vitrified waste form and canister.

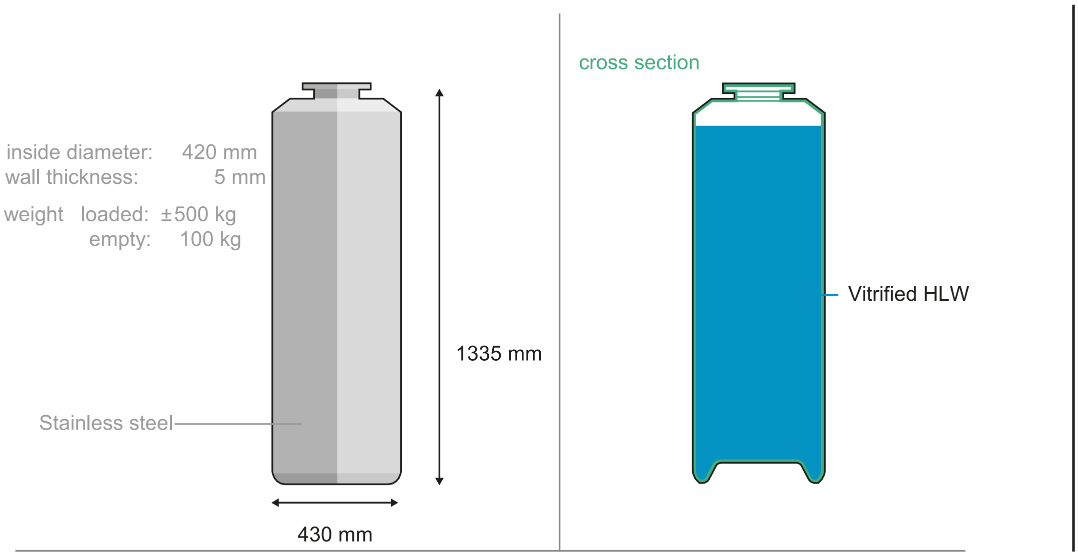


Figure 6-11: Schematic of vitrified HLW. Canister is to be put in a HLW disposal package.

Figure 6-12 shows the thermal power of each container and the main radionuclides responsible for the radiotoxicity of vitrified HLW. Early decay of many short-lived radionuclides with a half-life of less than 30 years means that their contribution to the thermal power will be negligible at the time of disposal. The radiotoxicity calculations were performed with the same data as used in Chapter 2. These data have been used in the Dutch RAS research programme on Recycling of Actinides and Fission products (Gruppelaar et al., 1998).

In the post-closure phase, as introduced for the safety concept in Chapter 3, the disposal package ensures that there is no contact between pore water and the vitrified waste form until the clay host rock is no longer significantly heated by the waste. When contact with water occurs, two characteristics of the waste form are assumed to determine the release rate of radionuclides from the evolved or altered disposal package: the alteration rate of glass and the glass cracking factor.

6.2.4.2 Alteration rate of glass

Many post-closure safety assessments consider solely the interaction between the vitrified waste and pore water for a description of glass alteration. The vitrified waste form is however contained in a stainless-steel canister, whose corrosion process and rate, along with the presence of corrosion products, may have an impact on the rate of the glass alteration and the formation of the Product Layer at the interface. Below, we first describe glass alteration by the interaction between glass and pore water as was modelled in OPERA. We then consider the influence of the corroded stainless-steel canister.

Glass interacting with water and its attenuation of radionuclides

The interaction between glass and water induces the formation of a sequence of alteration layers, as explained by Lutz and Ewing (Milodowski et al., 2015). Water is consumed by the formation of these layers. Assuming only silicon-oxide as the main glass constituent, 1 mol of reacted SiO_2 consumes 1 mol of water.

The gel layer formed on the glass acts a diffusion barrier, comprising a hydration zone containing silicon. The behaviour of other elements initially contained in vitrified waste depends on their solubility. Boron and (to a certain extent), lithium, sodium and molybdenum are highly soluble and most of these elements will be released quickly into the evolved pore water in the concrete buffer.
A reaction layer or alteration layer contains the precipitated products of those elements that form insoluble hydroxides, e.g., iron, aluminium, zinc, titanium and magnesium. The precipitated phases are clay minerals or, in some cases, zeolites (Conradt et al., 1986). Sorption of dissolved cations such as caesium and of cationic complexes such as americium and plutonium takes place on these clay minerals. The amount of these cations and cationic complexes released is therefore a fraction of the initial amount (Van Iseghem et al., 1992). Lithium and sodium are also cations but have a low affinity for ion exchange (Helfferich, 1962), as explained in Text Box 5-2.

The dissolution rate of the silica in the glass is also affected by the silica concentrations in the adjacent EBS materials. The pore waters in the evolved concrete buffer as well as in the clay host rock are saturated with silica and this will restrict releases from the glass. Alteration rates of basaltic glass, a natural analogue for the borosilicate waste form, have been estimated to be 0.1 μm per 1000 years in silica saturated environments (Lutze et al., 1987). This implies a glass dissolution rate of 7.4×10^{-7} g m^{-2}day^{-1} assuming a density of 2700 kg m^{-3} for basaltic glass. The type of mineral controls the concentration of silicon in silica saturated environments. The silicon concentration is in equilibrium with a CSH mineral in evolved concrete (see Figure 6-1) after concrete has been depleted in portlandite, at which point the pH of pore water is lower than 12.5. The silicon concentrations in concrete pore water increase with decreasing pH; (Berner, 1992). The silicon concentration at a pH of 11.77 is 0.09 mmol/l which is similar to the silicon concentration of 0.1 mmol/l in Boom clay (De Craen et al., 2004). It is therefore cautiously assumed that the alteration rate of the vitrified waste form will be larger for release into cementitious pore water with a pH higher than 11.77 than for clay pore water.

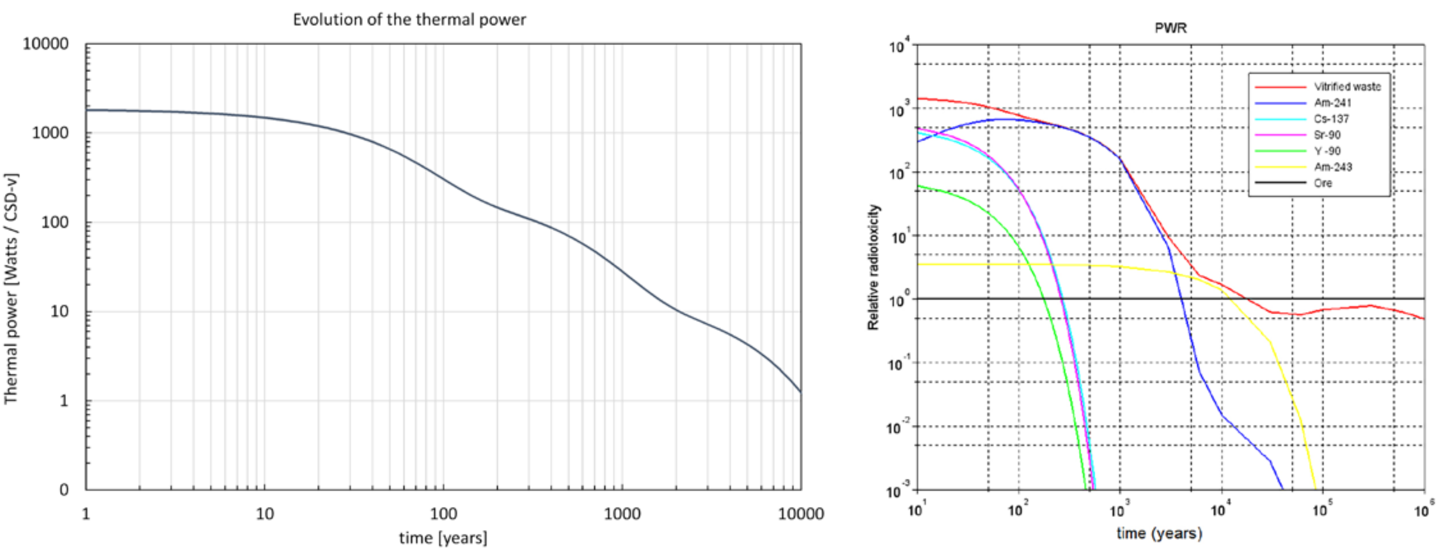


Figure 6-12: Thermal power (left) (AREVA, 2007) and radiotoxicity of vitrified HLW (right) from recycling of spent nuclear power fuel from a Pressurized Water Reactor (PWR) (Kloosterman, 2017).

Glass alteration in the presence of water and steel

Steel is present in the EBS in the vicinity of the vitrified waste form, which is in a stainless-steel container surrounded by a carbon steel overpack. The corrosion of iron has an impact on the glass alteration rate. The positively charged iron species formed during the steel corrosion process have an impact on the thickness of the layer of the precipitated phases with clay minerals. Also, the local increase in pH by the steel corrosion process increases the solubility of silicon due to the formation if hydroxyl ions in the corrosion process and insufficient dissipation. The research performed in EURAD-1 ACED shows that the glass alteration rates in the vicinity of (aged) iron, with both embedded in water saturated clay, is determined by the corrosion process of iron as iron silicates are formed (Gin et al., 2022). The glass alteration rate is then larger in the vicinity of iron. The vitrified waste form is however contained in a stainless steel canister (see Figure 6-11), whose corrosion rates are 10 times lower than those for iron and carbon steel (Swanton et al., 2015).

Stainless steel behaves differently from carbon steel and iron upon corrosion, since it releases chromium. Dissolved chromium is present as a negatively charged dissolved complex and is expected to have a less detrimental effect on the reduction in thickness of the glass alteration layer with passivating capacity than dissolved iron complexes (Neeft et al., 2022). Preliminary results in EURAD-1 ACED of an experiment in which stainless steel and glass are embedded in saturated clay confirm that the glass alteration rate in the vicinity of stainless steel is smaller than that measured in the vicinity of aged iron. The experiments have however run for a too short period to confirm that steady states have been achieved so that it is difficult to compare the glass alteration rates with steel to those rates obtained from glass directly exposed to a solution. In addition, the COPERA supercontainer design places carbon steel adjacent to stainless-steel and this will make transposition of the results of this type of experiment more difficult for a post-closure safety assessment in which there is contact between pore water and the vitrified waste form through the cracks in steel.

6.2.4.3 Cracking factor

At the time of disposal, the vitrified waste blocks may not be monolithic; cracks can develop during cooling after initial manufacture, or subsequently. The cracking factor is a parameter that is used in performance assessment studies to determine the surface area of glass that is exposed to pore water. The cracking factor is frequently obtained from leaching experiments (Ferrand, 2011), which requires more assumptions than does direct measurement of the total surface area.

Direct measurements of total surface area were determined in experimental studies with non-radioactive simulated HLW in the 1990s for the 3rd EC framework (RTD) programme. These studies also demonstrated with tomograms that it is possible to produce homogeneous glass blocks but with a very long cooling procedure in which the canister with poured glass remains undisturbed for many hours (Reimers, 1992). Full-scale tests with non-radioactive glass with very short cooling procedures show large glass shrinkage cavities as the inner part solidifies last and many circumferential cracks are generated by the large induced thermal gradient. A block of 391 kg of glass appeared to be broken in 11 pieces with a controlled cooling rate of 2.8 °C per hour in a full-scale test (Moncouyoux et al., 1991). The total surface area of these 11 pieces divided by the surface area of a monolith of glass leads to a cracking factor of 1.56. Shorter cooling periods lead to higher cracking factors. The cooling procedure that is representative for the vitrified waste form stored at COVRA's premises needs to be determined in order to determine the cracking factor just after its manufacture.

The heat and radiation produced by the radionuclides within glass can help to prevent or heal cracks as they act as glass network modifiers. The α-decay of actinides present in waste slightly diminishes the glass density; this appreciably improves mechanical properties, especially its resistance to cracking (Ribet et al., 2009). Defects are generated by the highly energetic alpha particles within the waste form, but the evolved helium diffuses so quickly at room temperature that these defects are annealed and no helium is trapped by defects within the glass (Chamssedine et al., 2010). There is no long-term α-damage. This lack in damage improves the resistance to cracking.

6.2.4.4 Assumptions made in the post-closure safety assessment

The assumed radionuclide release rate is currently determined by the cracking factor and a glass dissolution rate. Radionuclide releases are assumed to start after 1,000, 35,000 or 70,000 years in the post-closure safety assessment (Schröder et al., 2017b) since the lifetime assumed for the carbon steel overpack ranges from 1,000 years to 80,000 years (Kursten and Druyts, 2015). The glass alteration or dissolution rate used in OPERA was obtained from experiments employing cementitious pore waters that were unsaturated in silica. In the first days of such an experiment, the glass is dissolved at a fast rate until the solubility limit of silica within the experimental set-up is reached. The resulting alteration rates are so-called initial glass 'dissolution' rates and are usually obtained by measuring boron release, since boron is dissolved as an anion and therefore not incorporated in the clay minerals that are formed during glass alteration. The so-called long-term glass 'dissolution' rate is the steady state boron release and is considered representative for disposal conditions in the long-term. The rate at which the silica solubility limit is reached depends upon the amount of water in the system, so that the solid to liquid ratio has a high impact on the glass 'dissolution rates' obtained.

Table 6-2 shows that the glass 'dissolution' rates obtained at low solid to liquid ratios (probably[16] 2450 m^{-1}) and low dissolved silicon in the cementitious solution are more than 30 times higher than these rates obtained at a high solid to liquid ratio (302,400 m^{-1}, (Ferrand et al., 2023)) and with more dissolved silicon in the cementitious solution.

The third row in Table 6-2 shows the water consumption rates with the waste form dissolution rate as determined using the associated glass 'dissolution' rate. The best estimates in water consumption rates are used in Chapter 7 (section 7.1.3.1) in order to determine which glass alteration rates are possible when the water availability is limited by the thickness of the low permeability concrete buffer.

16. Glass 'dissolution rates' are determined from experiments in which a solid surface area is exposed to a volume of liquid. The solid to liquid ratio of the experiments referred to in OPERA (Deissmann et al. (2016b) has not been indicated. The database in Ferrand et al. (2023) shows that the experiments made in the past used low solid to liquid ratios and the more recent ones used high solid to liquid ratio. The solid to liquid ratio of the experimental data used in OPERA was obtained from this database.

	OPERA (italics available during COPERA)			Current knowledge
(Deissmann et al., 2016b)	Best Estimate	Lower Bound	Upper Bound	Analogue
Glass 'dissolution' rate [g m^{-2} day^{-1}]	6×10^{-3} (low SA/V) 1.8×10^{-4} (high SA/V) long-term	5×10^{-5} (low SA/V)	6×10^{-2} (low SA/V) 1.2×10^{-3} (high SA/V) initial	7×10^{-7}
pH at this rate	≤ 13.5 & 12.5 ≤ 13.7 ± 0.2	11.5	13.5 13.7 ± 0.2	Neutral cautiously assumed applicable until pH 11.7 since silicon concentration is similar (see section 6.2.4.2)
Water consumption rate [g/m year]	0.9 0.03	0.008	9	
Si at this rate [mmol/l]	≥ 0.3×10^{-3} ≥ 17×10^{-3}	?	0.3×10^{-3} 17×10^{-3}	0.1
Cracking factor	5.8*	5	100	1.56 (see section 6.2.4.3)
Glass package lifetime [years]	19000	6.2×10^{6}	260	
Waste form dissolution rate [$year^{-1}$]	5.2×10^{-5}	1.6×10^{-7}	3.9×10-3**	

*Surface area of glass block without cracks is 1.6 m^2, surface area equal to SRRF is 9.5 m^2 results into cracking factor equal to 5.8. Considering the voids between every filament (see Figure 6-8) compared to a localised void at the top of the vitrified waste form (see Figure 6-11), it is unlikely that the surface area for vitrified waste is larger than for SRRF. **a value of 3.8×10^{-3} per year (Schröder et al., 2017b) was used instead of 3.9×10^{-3} in Deissmann et al. (2016b).

Table 6-2: Comparison of assumptions of the glass waste form in OPERA collected by Deissmann et al. (2016b) with recent data in italics (Ferrand et al., 2023)

6.2.5 Compacted hulls and ends

6.2.5.1 Description of the waste

Compacted waste Standard Residues (Collis Standard de Déchets Compactés: CSD-c) arise from reprocessing spent fuel from nuclear power plants. These containers contain metal parts from the spent fuel assemblies that have been cut up to extract the spent fuel, then rinsed and dried. The rinsing minimizes the content of fission products and actinides so that the heat generated by this waste form is negligible, i.e., the canisters can be put next to one another without additional cooling during storage. This negligible heat generation has been used in the optimization of the design of the package for disposal, which does not need the additional protection provided by the carbon steel overpack used for the heat-emitting HLWs to provide complete containment during their 'thermal period' (see Figure 4-1).

A canister of about 170 litres internal volume is filled with either hulls or end pieces. The hulls are pieces of the fuel cladding which is made of Zircaloy; other metal parts are usually made of Inconel. End pieces are solid stainless-steel sections. Drums with other waste arising from reprocessing fuels, such as pumps, stirrers and filters, primarily comprise stainless steel. All drums are compacted to produce pucks that are loaded into CSD-c canisters with similar outer dimensions to those used for vitrified waste, and are then welded closed (see Figure 6-13). There is about 20% void space in the canisters.

The waste form contains radionuclides from two different sources: contamination from fuel residues and activation products. Radionuclides from fuel contamination are assumed to be present on the surfaces of the metal fragments, except for caesium and iodine, which can diffuse into the cladding (IAEA, 1985; Inoue et al., 1981). Activation products in the fuel cladding and other metal parts are assumed to be homogeneously distributed throughout the metals and their release rate into porewaters will be controlled by the corrosion rate of the metals.

6.2.5.2 Assumptions made in past and future post-closure safety assessments

The features of the waste form that determine the release rate of radionuclides into the evolved or altered concrete buffer are the surface area of the cladding and other metal parts, and the alteration rates of Zircaloy, Inconel and stainless-steel. These were not addressed in the post-closure safety assessment in OPERA, which used a simple, conservative model of instant release of radionuclides after the outer container fails at 1,000 years, 35,000 years or 70,000 years (Schröder et al., 2017b).

In reality, there will not be an instant release of radionuclides. The corrosion rate of Zircaloy at a pH of 12.5 is 0.002 μm per year (see Text Box 6-3), which is very small, even smaller than the 'dissolution' rate of glass. In a future post-closure safety assessment, the water consumption rate required to sustain the measured Zircaloy corrosion rate will be included in the calculations to determine the associated radionuclide release rate. The amount of Zircaloy in a single CSD-c is about 390 kg. The corrosion of this amount would require access to 80 kg of water. As discussed earlier, the permeability and diffusion values of the clay host rock and concrete in the multibarrier system will strongly restrict the access of large amounts of water to the wastes.

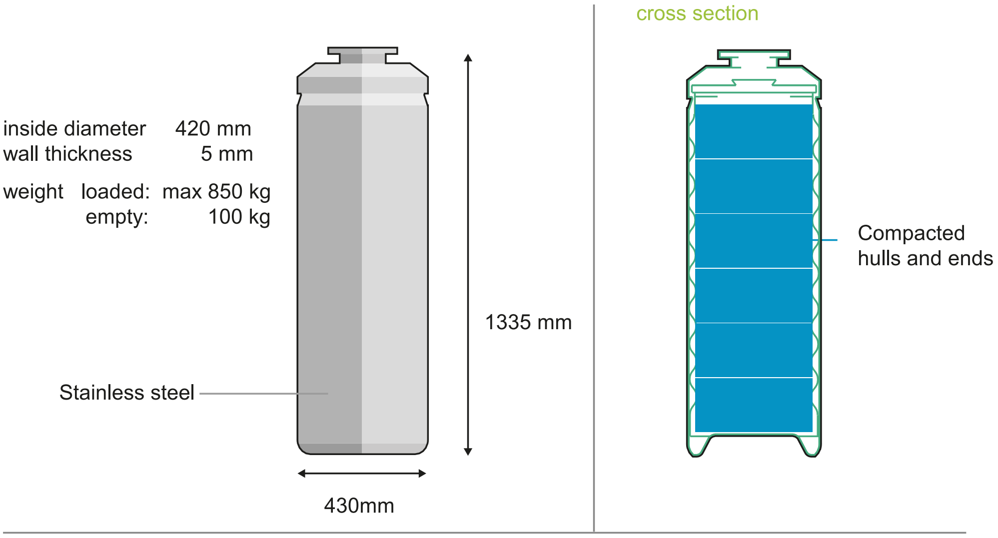


Figure 6-13: Schematic of the CSD-c container for compacted hulls and ends. Canister is to be put in a HLW disposal package.

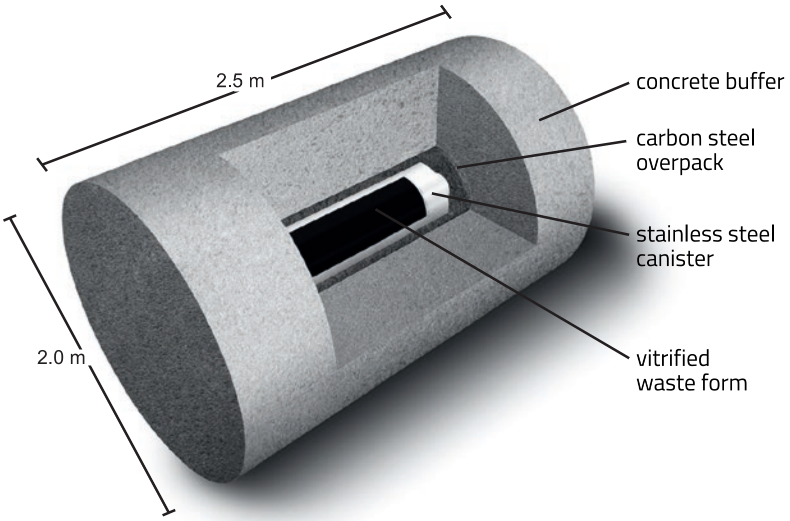


Figure 6-14: Supercontainer with concrete buffer and carbon steel overpack for vitrified HLW

6.3 The design of a disposal package for HLW

The safety concept in section 3.4.8 shows that the packages should provide sufficient radiological shielding to allow them to be contact-handled during emplacement in the GDF. As in the case of the 1000 litre containers (see Figure 6-6), concrete and steel are also foreseen for shielding the HLW disposal packages.
Figure 6-14 shows the supercontainer providing sufficient radio-logical protection during handling in the GDF (see section 6.3.3).

The safety concept introduced in section 3.4.8 also implies that the design of the disposal package for those HLW types that emit heat should prevent contact between pore water and the waste form for as long as the waste significantly heats the clay host rock in the post-closure phase. This simplifies the post-closure safety assess-ment because the values for diffusional properties of radionuclides at the natural in situ temperature of the host rock at disposal depth can be used. To prevent this contact with water, a metal barrier is employed, for example, steel.

At COVRA's premises, HLW is contained in stainless steel canisters; but these have not been designed to sustain the mechanical loads that will occur after closure of the GDF due to the overburden pressures. A carbon steel overpack 30 mm thick is foreseen. In the supercontainer, this overpack is surrounded by a concrete buffer. Steel embedded in concrete exhibits very small, uniform, and predictable corrosion rates. Figure 6-15 shows the hierarchical set of requirements in the RMS leading to the determination of the design specifications for this disposal package for heat generating HLW. The justification for the requirements and an explanation of their contribution to radiological protection and containment is provided in the next subsections.

Microbes are present in many of the initial constituents of concrete (Vidal et al., 2024). The potential for microbial activity can, however, be limited due space restrictions in the low permeability concretes of the waste package and buffer, which have pores with a maximum diameter up 0.1 µm (see Text Box 6-1). The proposed backfill of foamed concrete does have a sufficiently large pore size for microbial activity, but there is also a chemical effect in concrete

L1-DCRE-01: The permitted additonal dose for radiological workers is 20 mSv per year.

L1-NPRA-04: Waste shall be enclosed by a series of engineered barriers

L2-COV-01: The additional for radiological workers shall be less than 6 mSv per year.

L2-COV-04: Materials for which broad experience and knowledge exists, shall be used.

L3-D-NPRA-02: Waste shall be retrievable during the operational phase of the GDF through until its closure.

L3-D-IAEA-02: ..In the case of heat generating waste: the engineered containment shall retain is integrity until the heat production will nog longer adversely effect the performance of the multibarrier system.

L4-CD-PACK-RADPR-01: Contact handled waste packages are foreseen to be emplaced in the GDF.

L4-CD-PACK-CONTA-01: The physical and chemical properties of materials used for the package shall prevent contact between the waste form and pore water until the clay host rock is no longer heated.

L5-CD-PACK-RADPR-01: Gamma and neutron contact dose rate for each package shall be less than 0.12 mSv per hour (max. 15 minutes handling per day) and at 1 metre less than 0.0075 msV per hour (max. 4 hours handling per day).

L5-CD-PACK-CONTA-01: The package shall sustain a mechanial load of 10 MPa (500 metres) for 1200 years.

L6-CD-PACK-RADPR-01&CONTA-01: For HLW cooled for 130 years

1) the carbon steel overpack shall have a thickness of 30 mm (density 7850 kg m^{-3} & yield strength 600 MPa & uniform corrosion rate of 0.1 µm per year)
2) the outer diameter of the concrete buffer shall be 2 metres (thermal conductivity > 0.1 W / mK density 2350 kg m^{-3})

Figure 6-15: Hierarchical set of requirements in the RMS contributing to radiological protection and engineered containment, used in the design of the HLW supercontainer.

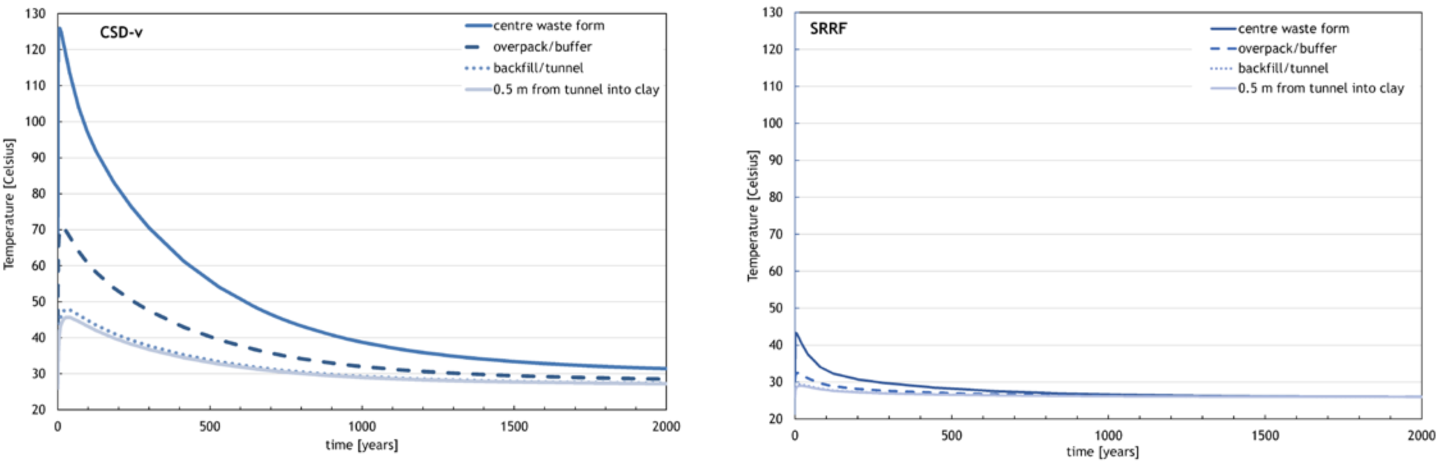


Figure 6-16: Calculated temperature at several locations in the multibarrier system with vitrified HLW (left) and SRRF (right). For both types of heat-generating HLW: concrete buffer diameter 2.0 m, tunnel outer diameter 5 meter. Length concrete buffer 2.5 m for vitrified HLW and 2.0 m for SRRF (see Chapter 4). Thermal powers as shown in Figure 6-9 (SRRF) and Figure 6-12 (vitrified HLW). Thermal properties as listed in (Neeft et al., 2021), except for the waste form for SRRF for which a thermal conductivity of 0.5 W $m^{-1}K^{-1}$ was used in the modelling.

that limits microbial activity. Microbial life in high pH environments requires a mechanism to keep a neutral cellular life and a proton motive force across the cell membrane to preserve proteins and produce adenosine triphosphate (ATP), a carrier of energy.
The concentration of H+ is very small at high pH and an insufficient supply is generally assumed to be available above pH 12 (although microbial communities have also been described growing at up to a pH of 13.2 (Wouters et al., 2016)). Microbial films have been formed within foamed concrete near the outer surfaces of foamed concrete specimens (Vidal et al., 2024). The current COPERA design does therefore not employ the stainless steel outer envelope of the HLW supercontainer that was considered in OPERA, so its susceptibility to microbial corrosion in contact with the surrounding foamed concrete backfill is now not an issue.

6.3.1 Determination of the period of the thermal phase

The HLW disposal tunnel will be backfilled with foamed concrete after emplacement of the disposal packages, as explained in section 4.7. The heat emitted by the waste form is dissipated into the concrete buffer, backfill, disposal tunnel and clay host rock. The emission of heat by the HLW reduces as a function of time as the radionuclides responsible for the heat emission decay.
The temperature in the centre of the waste form reduces due to this decay and to dissipation of heat (see Figure 6-15). The materials used as engineered barriers have their own restrictions for their performance. The temperature of concrete should remain below 100°C in order to prevent evaporation of water (Neeft et al., 2022). Figure 6-16 shows that, in the supercontainer for heat-emitting HLW, the calculated temperature at the interface between the overpack and the buffer is at most 70°C. Consequently, concrete can be used as a buffer in this design of the multibarrier system.

The end of the thermal phase is defined as the time at which the temperature near the interface between clay and the tunnel no longer significantly exceeds the natural in situ temperature of the host rock. As shown in Figure 6-15 for clay at 0.5 m from the tunnel, this takes a long time for vitrified HLW, whereas the thermal power of SRRF is so small that it can be questioned whether it is worthwhile defining a thermal phase with the current geometry of the EBS. The parameters to calculate transport, such as diffusion rates, vary by 2 to 3% per degree (Vanýsek, 2015). Consequently, a difference of 2°C between the clay interfacing the gallery and clay further away is considered negligible. In addition, the natural thermal gradient also induces a variation between the top and bottom of the clay of about 3°C for a clay formation with a thickness of 100 m. This natural variation in temperature is neglected in safety assessments. The thermal period for a multibarrier system with vitrified HLW then becomes 1,200 years, with the waste characteristics used in this safety case.

The period of the thermal phase together with the corrosion mechanisms and rates determine the additional required thickness of the carbon steel overpack to provide physical containment. Although microbially induced corrosion, radiation induced corrosion and chemically induced corrosion can occur, only chemically induced corrosion of the carbon steel overpack needs to be taken into account for the post-closure phase, with the design presented in this safety case.

Microbially induced corrosion can be neglected if a good bond is established between steel and concrete, since the size of the pores in the concrete buffer is too small for microbial activity. The use of pre-oxidized steel and ribbed steel surfaces facilitates the tightness of this connection. In addition, the radiation rate from vitrified waste is, even after cooling for 130 years (see Figure 6-16), too high to enable the existence of sulphate reducing microbes that can be responsible for radiation enhanced corrosion (Abrahamsen et al., 2015; Bruhn et al., 2009).

Radiation enhanced corrosion can also be excluded by design and by extending the storage period of the waste. The design presented in this safety case - a carbon steel overpack 3 cm thick embedded in a concrete buffer - excludes radiation enhanced corrosion. Radiation can enhance corrosion if radiolysis of water generates O_2 resulting in a more aerobic (oxidising) environment in the vicinity of the metal overpack. Aerobic induced steel corrosion rates are larger than anaerobic corrosion rates of steel (Crossland, 2005; Swanton et al., 2015). The outer radiation rate can be determined by the activity of the radionuclides in the vitrified waste form, the penetrating power of gamma rays that are released upon decay of these radionuclides and the steel thickness. The radioactivity of the waste is initially dominated by ^{90}Sr and ^{137}Cs (see, e.g., Figure 6-12).
The guaranteed maximal activity contents of these two radionuclides in a canister is 6600 TBq for ^{137}Cs and 4625 TBq for ^{90}Sr (AREVA, 2007). Only ^{137}Cs and its daughter, however, emit gamma rays of sufficiently high energy to contribute to the radiation dose rate at the surface of a 3 cm thick carbon steel overpack.

Because current Dutch policy is to have an operational disposal facility only in 2130, the vitrified waste will be cooled for over 100 years before emplacement. Consequently, the radiation dose rate will have significantly reduced at the time emplacement occurs. Based on research by Smart et al. (2017) for steel embedded in concrete, Figure 6-16 shows that a carbon steel overpack of 3 cm has a radiation dose rate that would allow radiation enhanced corrosion for a disposal package with an age of 60 years but not for 130 years.

The available literature shows that radiation enhanced corrosion and microbially induced corrosion can never occur at the same time, i.e. microbial activity occurs only at radiation doses rates lower than those that enable radiation induced corrosion. Radiation can kill microbes; the decrease in the number of viable bacteria as a function of increasing dose is determined and expressed as the D10 value - the total dose required to reduce the viable population by a factor 10 (Abrahamsen et al., 2015). The D10 values for relevant bacteria ranges between 0.5 and 1.57 kGy (Stroes-Gascoyne and West, 1997). For vitrified HLW, the doses received are many times higher than this. Consequently, microbes should not present a corrosion problem at any time, provided that the reduction in viable populations by radiation is not outweighed by the increase due to growth through consumption of nutrients and electron acceptors and donors. In practice, there are also several other arguments which make microbially induced corrosion highly unlikely, e.g., the high thermal load at the start of the post-closure phase, the small connecting pore throats that limit transport of food and energy sources, the drying of the concrete buffer at the start of the post-closure phase by the heat emitted by the waste and the high pH of the concrete pore water.

6.3.2 Providing mechanical support

A void volume is present above the vitrified waste form in the canister (see Figure 6-11), where the stainless-steel thickness is too small to prevent fracturing of the canister under the loads

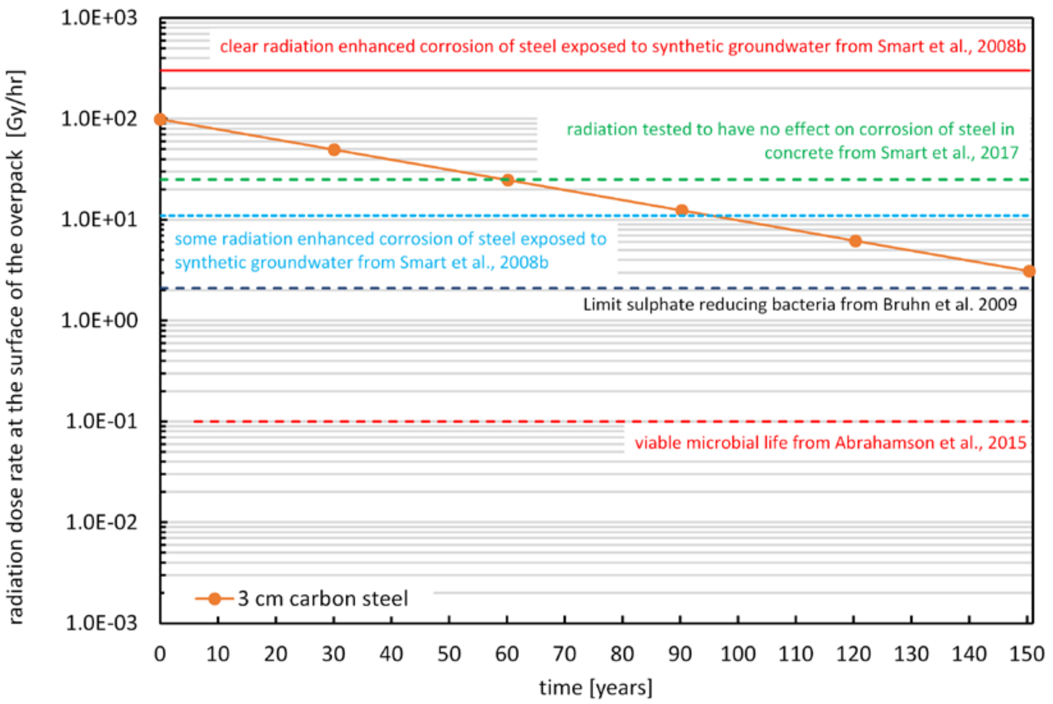


Figure 6-17: Radiation dose rate as a function of the time after CSD-v fabrication, calculated with Microshield (Neeft et al., 2022). Activity of ^{137}Cs at 0 years is assumed to be 6600 TBq, which is the maximum content as provided by the waste producer (AREVA, 2007).

expected. In the Belgian supercontainer concept, the concrete buffer does not contribute to mechanical support in the post-closure phase; the load is borne by the carbon steel overpack.
In the CORA programme, a mechanical analysis was performed for the vitrified HLW package (Barnichon et al., 2000); this has also been done in COPERA. Figure 6-17 shows the calculated von Mises stress for a carbon steel overpack for both heat-generating types of waste. As in CORA, a thickness of 30 mm was considered in COPERA to be sufficient for CSD-v, since the von Mises stress is at least 3 times smaller than the yield strength of carbon steel.

6.3.3 Determination of the thickness of the concrete buffer

As explained at the start of this section, for steel embedded in concrete, the corrosion rates are very small, uniform and predictable. The smallest corrosion rates are expected if the pH of concrete pore water is higher than 11.7. At a pH lower than 11.7, ion exchange of dissolved iron with the negatively charged cementitious minerals cannot be excluded (see Figure 6-1) so that the corrosion rate may become larger. Modelling studies in OPERA showed that the pH in the vicinity of the carbon steel overpack remains 12.5 for a period of at least 80,000 years (Kursten and Druyts, 2015).
The modelling was performed for concrete made with CEM I, which has a larger content of portlandite than the concrete buffer made with CEM III/B. Concretes made with a blended cement are called low-permeability concretes due to the refined pore structure (Atabek et al., 1991; Atkins et al., 1991; Jackson et al., 2017). Consequently, the ingress and egress of dissolved species can be smaller in these than in concrete made with CEM I. The thickness of the concrete buffer is therefore currently assumed to be controlled by operational radiation protection and mechanical requirements, rather than by chemical requirements.

The carbon steel overpack contributes to shielding, but the radiation dose rate of 7 to 8 Gy per hour (see Figure 6-16) is too high to allow the overpack to be contact-handled in the GDF. Figure 6-18 shows that the calculated gamma dose rates for the two types of heat generating HLW: CSD-V and SRRF, for concrete buffers with two different densities, are higher than the 0.1 mSv/h dose rate design requirement in Figure 6-15.

The main contributor to the dose rate is ^{137}Cs (and its daughter ^{137m}Ba) and Appendix 6 shows that this activity is almost ten times higher for CSD-v than for SRRF. However, the diameter of the waste canister for SRRF is almost twice as large as CSD-v so that the concrete buffer thickness for a supercontainer disposal package with SRRF is smaller for the same outer diameter of the supercontainer for CSD-v (see Figure 4-1).

The thickness of concrete can be reduced by the use of aggregates with a higher density. However, neutron shielding should also be taken into account with these types of HLW. These calculations are not made for this safety case but calculational results for PWR assemblies are available. These calculations have been used in the current choice of the type of aggregates in the concrete buffer. These calculations were carried out for concrete using depleted uranium granules as aggregates rather than quartz, to attenuate neutron and gamma dose from 24 PWR spent fuel assemblies. For this spent fuel, the required thickness for neutron shielding is larger than gamma shielding when using depleted uranium (Quapp, 1999). For now, a concrete buffer with aggregates with a similar density to quartz (i.e., calcite) is therefore proposed in order to obtain sufficient neutron shielding.

In the post-closure phase as well as in the operational phase, there will be a thermal gradient within the concrete buffer surrounding heat generating HLW. The thermal stresses associated with this thermal gradient induce compressive stresses in the buffer near the overpack and tensile stresses near the outer diameter of the buffer. The highest temperatures are expected in the operational phase, since the air surrounding the supercontainer has a lower thermal conductivity than the backfill. Prior to backfilling a disposal

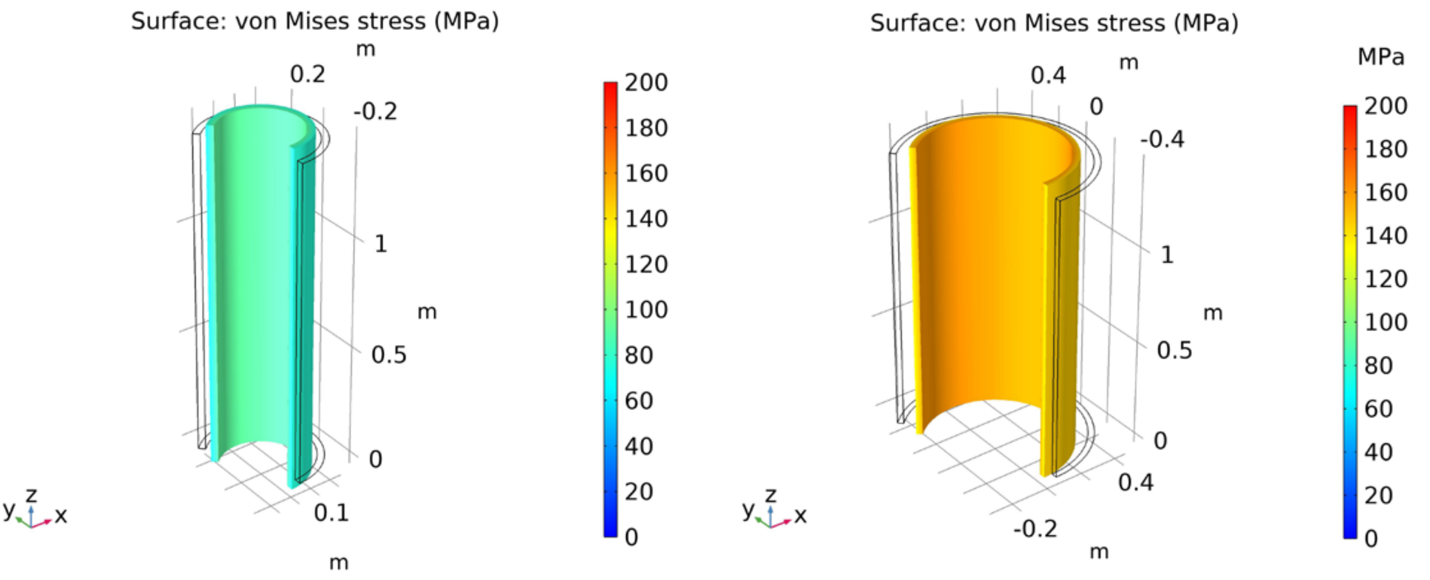


Figure 6-18: Calculated von Mises stresses for an overpack for CSD-v (left) and SSRF (right), both with a thickness of 30 mm.

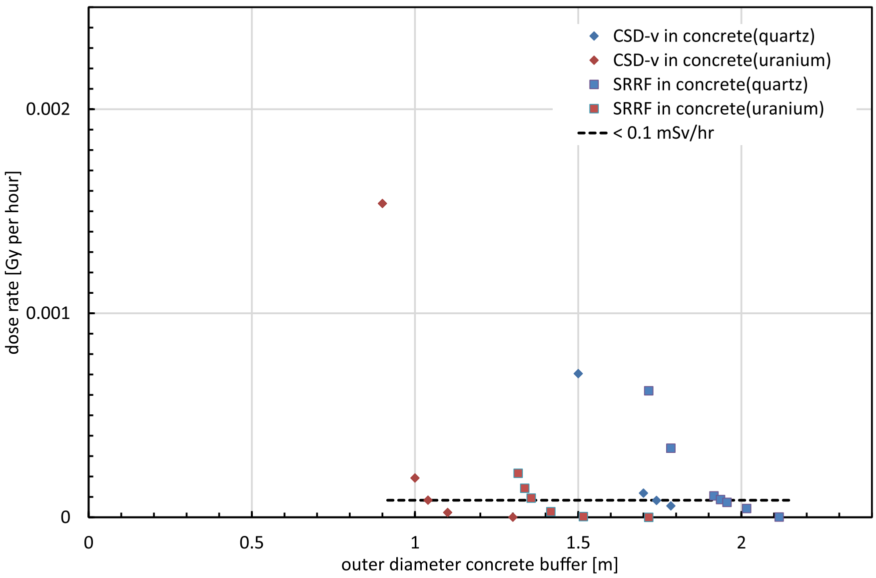


Figure 6-19: Calculated gamma dose rates for encapsulated heat generating HLW with a stainless steel canister, carbon steel overpack and a density of the concrete buffer of 2350 kg m^{-3} with siliceous aggregates and 5600 kg m^{-3} with depleted uranium granules as aggregates.

tunnel, the rate of emission of heat from the supercontainer into air depends on the velocity of air flowing over the surface of the supercontainer and the relative humidity of air. Thermo-mechanical calculations have been made for a supercontainer encapsulating vitrified HLW. The heat output is a very important input for these calculations and vitrified HLW was assumed to be cooled for 130 years. Although the temperature of the concrete buffer in the supercontainer was calculated to remain below 100°C, the calculated tensile stresses near the outer surfaces of the buffer may be too high (Neeft et al., 2021). The design of the disposal package for HLW is therefore still in development. A reinforced concrete container will also be considered in a future study of a disposal package for HLW in order to have sufficient tensile strength. The rebars are encapsulated in 'engineered' impermeable concrete which prevents microbially induced corrosion of steel.

Box 6-3: Gas generation and transport in the multibarrier system

In the normal evolution scenario of a multibarrier system with engineered and natural barriers, the movement of radionuclides at long times in the future takes place by diffusion in the pore waters of the undisturbed clay. However, gases can be generated in the EBS and these can potentially lead to a built-up of pressure that might cause disturbances in the clay host rock, affecting the movement of radionuclides in the pore waters. This box examines the mechanisms that can lead to gas generation and considers how any gases produced might be transported out of the engineered barrier system (EBS) without unacceptable deterioration of the clay host rock containment properties.

Gas can be generated in the wastes and the materials of the EBS by:

1. alpha decay of radionuclides, leading to helium production;
2. radiolysis of porewaters, leading to hydrogen and oxygen production;
3. microbial degradation of organic materials, generating CO_2 and CH_4, which may possibly include small quantities of radioactive gases, in particular C-14;
4. anaerobic corrosion of metals in the wastes and the containers, which generates hydrogen and is the principal gas source in a closed GDF.

For the wastes and packages in the GDF, the first two of these processes lead to negligible gas generation and thus have no impact on radionuclide movement away from the GDF.

For the third mechanism, microbial activity in both the near-field Paleogene Clay and most concretes of the EBS is expected to be limited due to pore size and tortuosity restrictions, and those microbes present are expected to remain in a dormant mode. An exception is in the porous backfill.

The fourth mechanism is expected to be the main contributor to the quantities of gas generated. Anaerobic corrosion rates can be obtained from experiments in which a piece of metal is exposed to a cementitious solution and the hydrogen generation rate is measured (for example in work performed in the European Commission Carbon-14 Source Term (CAST) project (Norris and Capouet, 2018)). The hydrogen evolution is used as an indicator for the corrosion rate. The hydrogen generation rate reduces with time due to the formation of a metal-oxide layer. After about 1000 days, the hydrogen generation rate becomes constant, corresponding to the dissolution of this oxide layer. This has been shown for steel (Swanton et al., 2015) and for aluminium (Fujisawa et al., 1997). The metal beneath the corrosion layer is then consumed at the same rate and produces hydrogen.
The table below shows the long-term chemical corrosion rates of the metals present in the waste and EBS.

The gas generation rate at disposal tunnel scale has been determined by the surface area times the corrosion rate and the production rate of stoichiometrically formed hydrogen as a function of the assumed corrosion reaction, as described, e.g. by Yu and Weetjens (2012). Stoichiometrically: $^4/_3$ hydrogen molecule is generated and $^4/_3$ water molecule is consumed for each corroded iron atom in the case of the formation of magnetite in the anaerobic corrosion process. Iron and aluminium are examples of metals in which the generated hydrogen can be free to escape during the corrosion process. This is not the case when hydrides are formed in the corrosion process. Hydride formation in many metals has been known for many decades, e.g., in zirconium (Zr) leading to ZrH_x (Lacher, 1937) and 90% of the generated hydrogen in anaerobic corrosion is picked up by Zircaloy hydrides (Sakuragi, 2017).

The EURAD-1 GAS Work Package investigated the diffusion of gases in saturated and unsaturated clay host rocks and in fabricated clay samples, providing a state of the art study of the potential gas transport mechanisms (Levasseur et al., 2021). Preliminary calculations were performed for Dutch waste with the EBS geometry considered at that time, for fully saturated cases (see figure below). The calculations used the expected hydrogen diffusion values in concrete for the EBS, or larger values similar to those found in clay that might be appropriate for degraded concrete. The analyses assume that the gas is fully dissolved. The major gas formed during radiolysis is hydrogen. These calculations (see figure left, below) confirm that radiolysis of pore waters will be a negligible source of generation of gas if shielding by the carbon steel overpack is included: i.e., a distinction between the curves (shielding magnetite and no radiolysis, magnetite) can hardly been seen. In the OPERA design, an outer stainless-steel envelope of the HLW disposal package was interfacing the foamed concrete backfill and the thickness of

Long-term corrosion rates of metals in cementitious pore solutions, in micrometer per year

Metal	Carbon steel	Stainless steel	Zircaloy	Aluminium
Long-term	≤ 0.1	≤ 0.01	≤ 0.002	≤ 10
Reference	(Deissmann et al., 2021)	(Mibus et al., 2018)	(Necib et al., 2018b)*	(Fujisawa et al., 1997)

*corrosion rate determined from release rate of non-radioactive nickel and chromium due to uptake of hydrogen of hydrogen by Zircaloy in the corrosion process (Sakuragi, 2017).

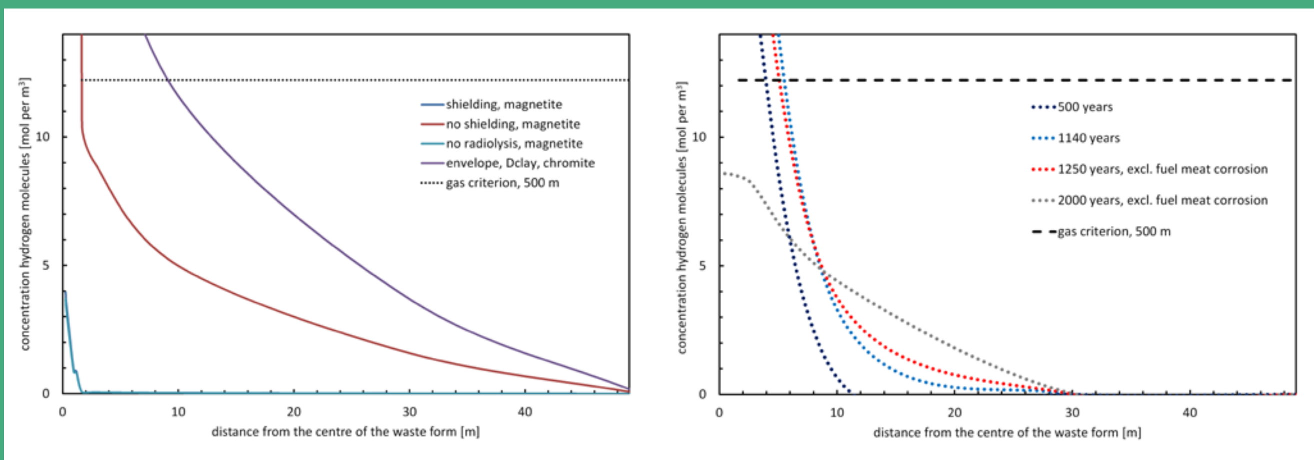
the foamed backfill was significantly smaller than in the COPERA design. As explained in section 6.1.4, microbial activity cannot be excluded. For microbial corrosion of a stainless envelope surrounding the concrete buffer (assuming a corrosion rate of 1 µm per year), the gas solubility in Paleogene clay host rock would be exceeded. The carbon steel overpack allows much more hydrogen dissipation, so that the hydrogen solubility in the Paleogene clay would not be exceeded, even if the same corrosion rate is assumed. In COPERA, the stainless envelope has therefore been excluded from the design of the HLW disposal package in order to prevent disturbance of the clay host rock in the post-closure phase.

There is no gas generation upon alteration of the vitrified waste form. The designs of the EBS for CSD-v and SRRF are similar leading to similar calculational results. Additional calculations have been performed for the case that the carbon steel overpack and stainless-steel canister is completely corroded, by which 100% of the surface area is in contact with the waste forms.

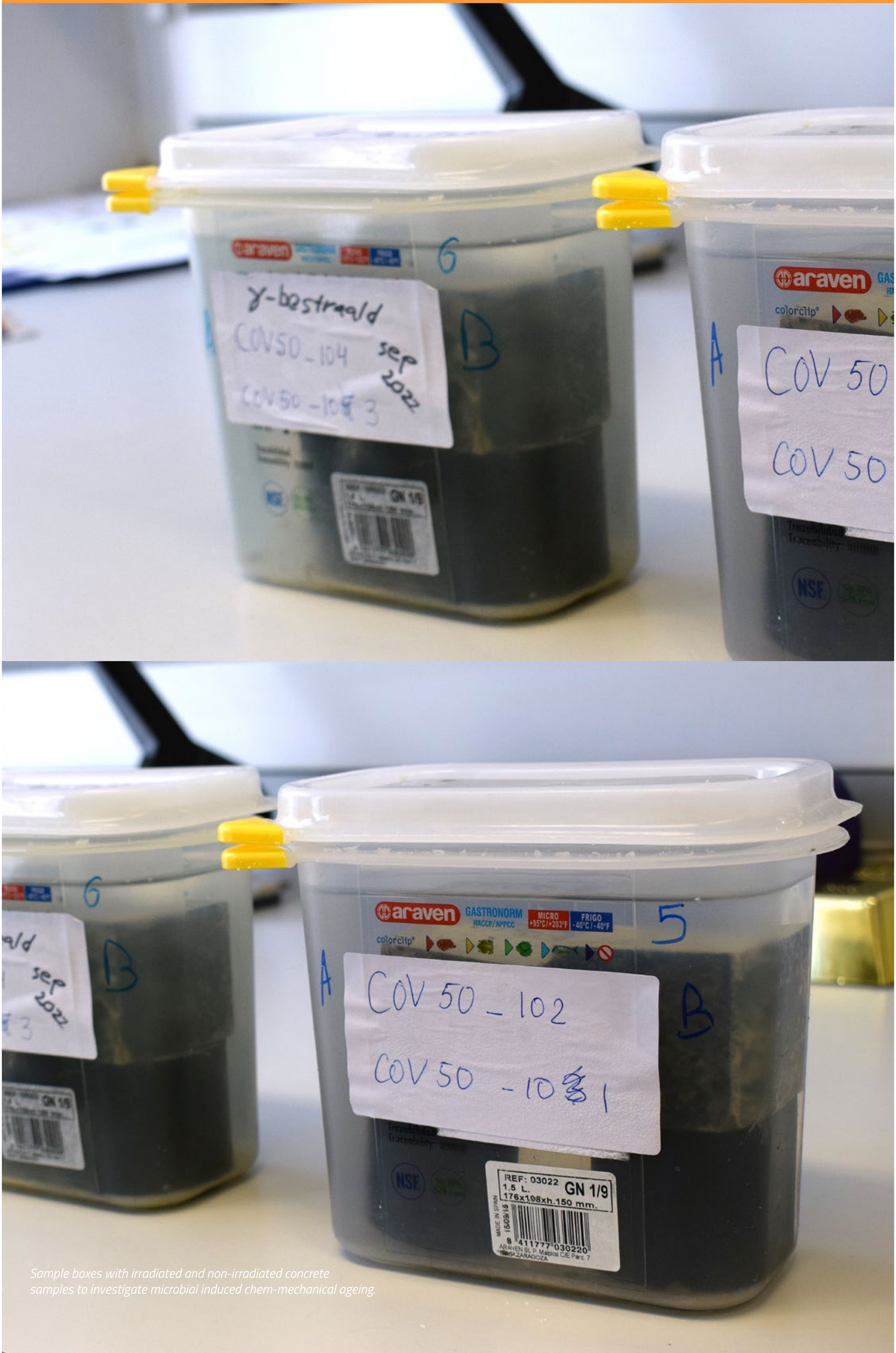
For SRRF, the major gas production occurs through corrosion of aluminium in the cladding. The corrosion rate is higher at high pH, but the concrete buffer loses its high pH by egress of alkalis and ingress of bicarbonate and magnesium. Assuming achievement of a neutral pH when there is contact between pore water and the aluminium cladding is the most optimistic calculation for SRRF. Even if a corrosion rate of 0.5 µm per year (Gustafsson, 2011) is assumed, i.e., 20 times smaller than shown in the table above, the gas solubility in clay would be exceeded so that perturbation of the clay host rock is expected. It is important to note that all of these corrosion calculations assume that there is a sufficient supply of pore water from the clay host rock to sustain the reactions.

In the EURAD-1 ACED project, more detailed consideration was given to water consumption in the anaerobic corrosion process. Recent work by Stefanoni et al. (2018) shows that the corrosion rate of rebars in reinforced concrete decrease with decreasing saturation degree (see section 6.1.3). The anaerobic corrosion process consumes water, and the potential consumption and supply rates of water are therefore also believed to control the generation of gas within the EBS at the disposal tunnel scale. Several attempts have been made to measure the hydrogen evolution rate from steel embedded in concrete directly, in order to obtain a reliable quantitative corrosion rate, but the slow transport of hydrogen diffusion within the concrete delays the arrival of hydrogen at the detection device too much (Kaneko et al., 2004). Accurate steady state anaerobic corrosion rates of metals in porous media are therefore absent and, in a laboratory setting, only the corrosion rates of metals directly immersed in a solution can be calculated by measuring hydrogen generation.

More complicated calculations in which the corrosion rate varies as a function of the potential water consumption rates are needed in order to assess whether gas generation by the anaerobic corrosion of metals in the EBS would perturb the clay host rock. Such type of calculations have been done in the EURAD-1 ACED project (Blanc et al., 2024) but need to include the specific geometry of the multibarrier system in question. For now, the prioritization of research devoted to gas experiments in clay rocks has diminished, since the perturbation of the clay host rock can by excluded by design and using realistic representative water consumption rates for the anaerobic corrosion of steel and aluminium in the EBS.



Stationary 1D-axisymmetric calculations for a multibarrier system with vitrified HLW (left) and transient calculation with SRRF (right) (Levasseur et al., 2021). At the time when these calculations were performed, the porosity and saturated diffusion values presented in Table 6-1 were not available. From these results, the hydrogen diffusion values could be estimated for the concrete buffer and foamed concrete. The saturated hydrogen diffusion value used was, for the concrete buffer: 7×10^{-12} m²/s, 3×10^{-10} m²/s for the backfill and 9×10^{-10} m²/s for clay host rock.



Sample boxes with irradiated and non-irradiated concrete samples to investigate microbial induced chem-mechanical ageing.

7. Evolution of the Multi-Barrier System

Our understanding of the properties and behaviour of the natural and engineered barriers underlies the concept of isolation and containment provided by the multibarrier system. Safety assessment, as presented in detail in Chapter 8, quantifies this evolving behaviour in order to forecast the performance of each component of the system and of the whole multibarrier system.

The information to quantify performance is subject to different types and levels of uncertainty. Safety assessment addresses this in a number of ways: by making conservative simplifications, assuming poor performance, using pessimistic parameter values and omitting potentially beneficial processes if they are not well-enough quantified. The results of safety assessments are thus designed to be conservative, so that it is expected that these assessments will make pessimistic forecasts of system performance. Nevertheless, it is essential for system engineering optimisation purposes to also make best estimates of how we expect the system to behave, acknowledging the uncertainties along the way. This allows a balanced view to be taken between a realistic assessment (somewhere close to expected behaviour) and a simplified but robust assessment showing the system is safe, even with considerable in-built conservatism. This balance is essential in order to take informed decisions later in the programme on GDF design optimisation and, eventually, on acceptable site characteristics. For example, this approach avoids over-engineering system components or rejecting otherwise acceptable GDF sites.

In this Chapter, we assemble information from previous Chapters on system understanding and the design of the GDF to compare best estimate behaviour of the multibarrier system with the assumptions made in the safety assessment. This is done in the form of a narrative describing the 'normal evolution' of the engineered and natural barriers of this multibarrier system, together with parallel commentary on how this is simplified in the quantitative assessment presented in Chapter 8. In the narrative in this safety case, the focus is on the engineered barrier system (EBS) and the clay host rock in the vicinity of the EBS. The 'normal evolution' scenario is designed to describe what we consider to be the most probable evolution of the multibarrier system. The normal evolution scenario contains a range of cases (or realisations) to encompass the expected range of variability and uncertainty in key parameters that affect system behaviour.

Alternative evolution scenarios comprise a set of cases in each of which the normal evolution scenario is changed in a specific way. They represent conditions that are physically conceivable but are considered much less likely to occur than those of the normal scenario, including, for example, unexpected or highly unlikely processes. Many of these scenarios were already identified in the OPERA Safety Case (Verhoef et al., 2017).

A third group of 'what if' scenarios covers processes or events for which no direct drivers are apparent or which are of extremely low probability, or situations that are not physically reasonable (e.g., omitting a key component of the EBS). They are used to test and illustrate the contribution to the containment and isolation provided by the individual barriers in the multibarrier system. These scenarios represent entirely hypothetical situations but highlight key sensitivities and points of focus for optimising the GDF. An example would be testing system behaviour assuming that no overpacks exist, to assess the specific contribution of the disposal overpack to system safety. Many of these scenarios have also been identified in the OPERA Safety case (Verhoef et al., 2017).

Specific scenarios that address human intrusion are also included in this safety case.

7.1 Normal evolution

In this section we focus on the evolution of the multibarrier system at disposal tunnel scale with disposed of vitrified waste, using new experimental results that have been obtained in EURAD-1 ACED (Gin et al., 2022) since our previous OPERA safety case analysis. To facilitate the evolutionary story, we look at four different time periods after closure of the GDF (see Figure 7-1):

- Closure to near the end of the thermal phase (1,000 years);
- From near the end of the thermal phase (1,000 year) out to 10,000 years, a period in which the clay host rock is no longer heated by the waste and the carbon steel overpack in the disposal package is still intact;
- From 10,000 years out to an ice age at 100,000 years, during which clay pore waters come into contact with the wastes;
- After a first ice age (100,000 years) to 1,000,000 years.

7.1.1 Closure to 1,000 years

7.1.1.1 Expected behaviour

In the operational phase, the tunnel liner begins to deform in response to the lithostatic load of the overlying geological formations, slowly taking on an oval-shaped form owing to the anisotropy of these stresses (Dizier et al., 2023). This type of deformation is expected to continue into the post-closure phase and becomes more pronounced as the strength of concrete is progressively lost. Before closure, ingress of clay pore water to the disposal tunnel occurs through the joints between the concrete segments of the liner. This water evaporates during emplacement of the waste packages, due to ventilation of the GDF but after backfilling is expected to slowly saturate the cementitious materials, including the backfill (foamed concrete), waste package concrete for LILW and the concrete buffer in the HLW disposal packages.
The low permeability clay limits the access of water to the engineered barriers in the post-closure phase. Figure 4-9 shows that the concrete liner prevents the clay from drying during the open, operational period of the GDF, so the clay host rock remains saturated. In the post-closure phase, the clay host rock is also expected to remain saturated. There is a limit to the water outflow from the clay into the EBS if the clay host rock is to remain saturated. This has been estimated (see Figure 7-2) for a single-level GDF located centrally in a clay host rock with a thickness of 100 metre

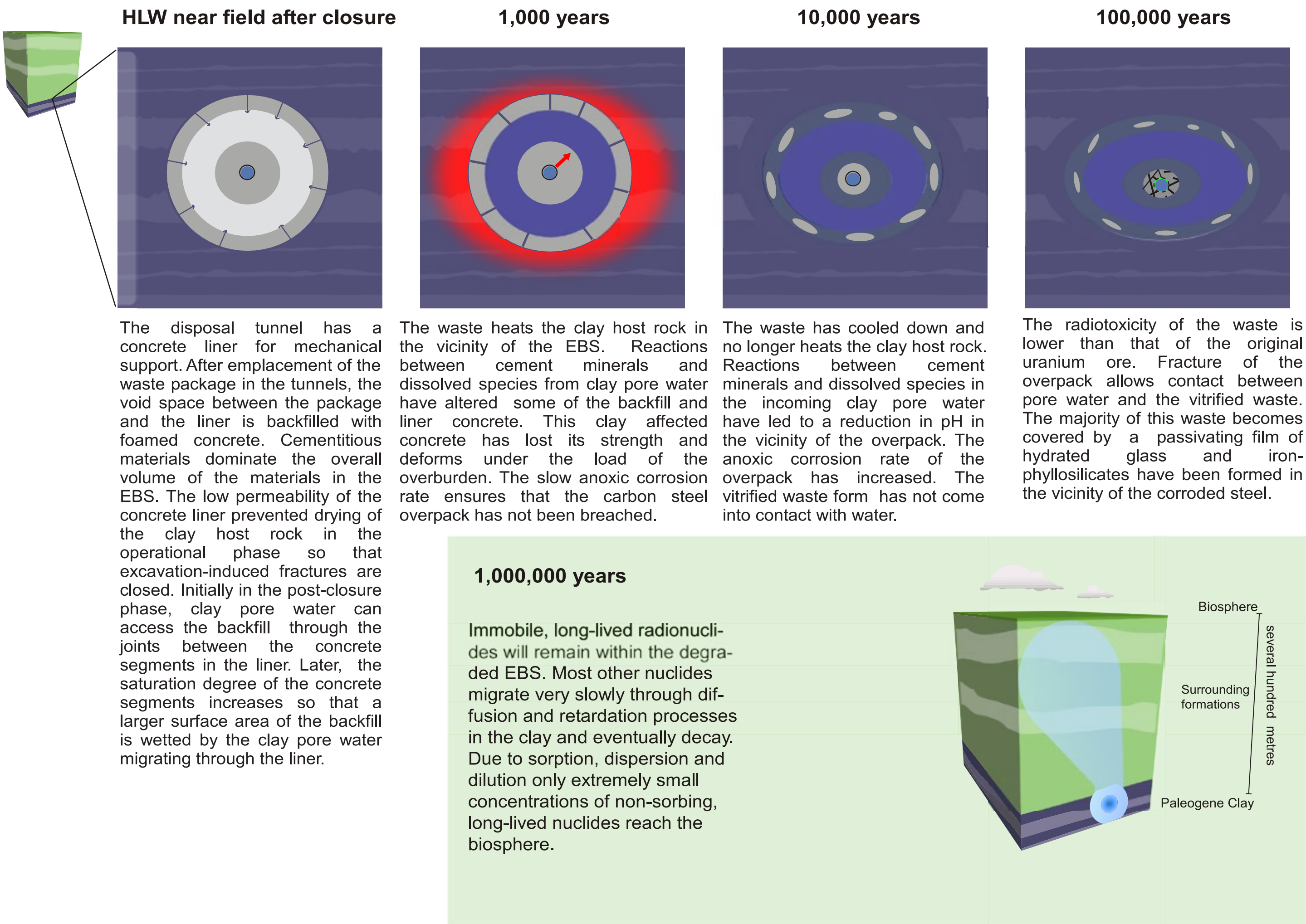


Figure 7-1: For each period, an illustration is provided of a cross section of a disposal tunnel surrounded by Paleogene clay. The vitrified waste form is indicated in blue.

and centre to centre distance between disposal tunnels of 50 m. A very small water inflow of 0.1 kg water per metre of tunnel per year is the highest possible flow of water leaving the clay, if it is not to become unsaturated. Inclusion of the anisotropy of clay slightly increases the maximum allowable water outflow that can occur without desaturation of the clay.

The choice of cement type will determine the extent to which oxygen entrapped in the manufacturing process will play a role in the early corrosion process of steel. Steel is used for all packages of waste i.e. LILW as well as HLW. All different types of concrete are assumed to be made with CEM III/B. Concrete made with CEM III contains pyrite, which provides a reducing environment for steel corrosion, meaning that passive anaerobic corrosion starts immediately (Atkins et al., 1991). In this narrative, we assume all cementitious materials in the EBS to be made with CEM III/B. Traces of carbon dioxide in air, initially entrapped in the pores of concrete are consumed by reactions with cementitious minerals. Entrapped nitrogen becomes dissolved as the saturation of the concrete continues. Hydrogen is formed by anaerobic corrosion of steel and the corrosion rate increases with the saturation degree of concrete (Stefanoni et al., 2018) and water is consumed.
There is a further water consumption reaction apart from anaerobic corrosion of steel: the transformation of calcium-siliceous hydrates (C-S-H phases) in concrete with ingress of CO_2 and bicarbonate into siliceous hydrates and calcite. The reaction of the portlandite in concrete with CO_2 ingress and then of the bicarbonate formed into calcite, produces water. So, there are cementitious minerals (e.g. C-S-H phases) in concrete whose reactions with CO_2 consume water and other cementitious minerals (e.g. portlandite) whose reactions with CO_2 produce water. The portlandite content for concrete made with CEM III/B is very small compared to the content of C-S-H minerals. For concrete made with CEM III/B, reactions with CO_2 for all cement minerals (portlandite and calcium-siliceous hydrates) have been assessed to consume more water than production of water (Blanc et al., 2024)

Leaching increases the porosity of concrete causing a reduction in the strength of the tunnel liner and backfill, waste package concrete for LILW as well as the concrete buffer in the HLW supercontainers. A reduction in strength is usually associated with a change in pore structure, leading to larger connecting pore throats, thereby increasing the permeability of the concrete and the diffusion values of dissolved species. The potential for leaching of concrete can be assessed by comparing the calcium concentration in clay pore water with its concentration in concrete pore water. The calcium concentration in clay pore waters with a similar salinity level to seawater is about 13.2 mmol/kg (Griffioen et al., 2017).
The calcium/silicon ratio of C-S-H phases in equilibrium with a calcium concentration of about 13 mmol/l ranges between 1.2 and 1.5 (Berner, 1992). For CEM III/B, the majority of cement minerals are C-S-H phases, with a calcium/silicon ratio assumed to be 1.1 (Neeft et al., 2022). Consequently, only a very small proportion of the cementitious minerals in concrete made with CEM III/B is available for leaching. Decalcification of calcium-siliceous hydrates by reaction with dissolved CO_2, bicarbonate and magnesium are the main alteration processes, but these cause only small changes in the porosity of concrete (Blanc et al., 2024). Reactions between cementitious minerals and magnesium in pore waters has been hypothesized to lead to a reduction in strength (Atkinson et al., 1985). However, after almost 8 years of exposure in experiments, a reduction in strength has not been measured for concrete made with CEM III/B and exposed to a clay pore water solution as saline as seawater (Vidal et al., 2024). Nevertheless, at present, we conservatively assume that ingress of dissolved species does lead to a reduction in the strength of concrete and consequent deformation of the tunnel liner, backfill and waste package concrete for LILW and the concreter buffer in the HLW supercontainers.

The tunnel backfill has the largest porosity of all the cementitious materials used in the EBS. The first alteration zones within the concrete backfill caused by ingress of dissolved species from clay pore water through gaps in the tunnel liner are expected to form after 1000 years, since saturating the backfill already takes 1000 years.

In OPERA, the concrete buffer for the supercontainer was chosen to be made with CEM I, which is Ordinary Portland Cement (OPC). Concrete made with CEM I lacks redox sensitive species and is therefore slightly oxidizing after manufacturing. The carbon steel overpack in the supercontainer is encapsulated in a concrete buffer made with this type of cement, first undergoes aerobic corrosion for a short period in time. In COPERA, this first aerobic corrosion is absent due to the choice of CEM III/B for manufacturing the concrete buffer. For heat-generating HLW, the corrosion rate of the

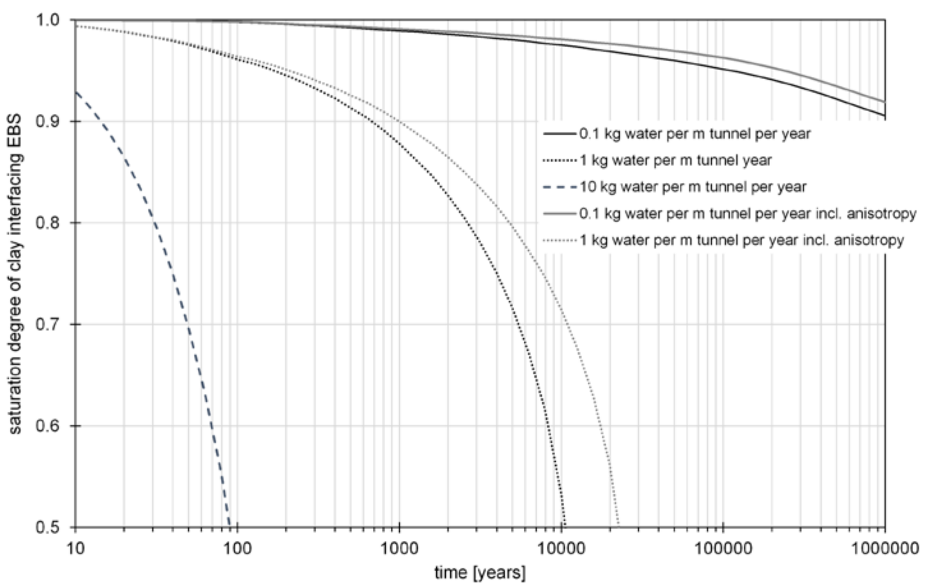


Figure 7-2: Evolution the degree of saturation of clay interfacing an EBS with a diameter of 5 m (see Figure 4-2) at different hypothetical inward flows of water per metre of tunnel length interfacing clay (i.e. an area of 5π m^2) using a saturated diffusion value for water in clay of 0.7×10^{-10} m^2/s perpendicular to the bedding plane and 1.4×10^{-10} m^2/s parallel to the bedding plane (Aertsens et al., 2023) and the relationship as defined by Millington and Quirk (1961).

carbon steel overpack is restricted by the restricted availability of water due to the low permeability of the concrete buffer to 1 μm per year (see Figure 7-3 with the thickness of the concrete buffer in the disposal package for vHLW and an initial saturation degree of concrete of 70%). At higher corrosion rates of carbon steel of 2 μm per year, the concrete buffer in the vicinity of the overpack would be dried after almost 5,000 years (saturation degree becomes 0% in Figure 7-3) so that the corrosion process would stop. The actual corrosion rates are lower, a maximum is set to 0.1 μm per year (Neeft et al., 2022), due to the high alkaline conditions which lead to formation of the insoluble corrosion product, magnetite.

The corrosion rate of 0.1 μm per year is sufficiently small to ensure that the concrete buffer in the supercontainer for vHLW remains saturated with the inflow of clay pore water and that hydrogen can leave the EBS without disturbance of the clay host rock. Initially, there is some movement of water from the backfill towards the tunnel liner since the size of the pores that were initially saturated in the backfill is larger than the pore size in the liner. At the joints between concrete segments of the liner, there is immediate inflow of water into the tunnel backfill. Later, the interfacial fluxes of water for all concrete materials decrease as they become further saturated. The calculated increase in the saturation degree of the concrete buffer interfacing the carbon steel is a conservative estimate, since the thermal gradient counteracting the inward flow is not included. There is no contact between pore water and the vitrified waste form during this period and the radiotoxicity of the waste has reduced by an order in magnitude after 1000 years .

This period is sometimes referred to as the 'thermal period' since the adjacent clay host rock will have a significantly higher temperature than the natural in-situ temperature, due to the heat generated by the waste. Some dissolved species, such as hydroxyl ions and dissolved alkalis from the concrete components of the EBS, migrate outwards into the clay host rock. The small porosity of the tunnel liner limits this egress of dissolved species and the impact of hyper-alkaline fluids diffusing out into the clays has been evaluated in natural analogue studies performed in Maqarin (Jordan), Cyprus and the Philippines, as well as in various modelling studies (Deissmann et al., 2021), so its maximum extent can be quantified (Savage, 2014). The reactions involved can cause swelling clay minerals such as montmorillonite to be transformed into non-swelling clay minerals such as illite, and C-S-H phases to be formed in the regions of host clay that are affected. This results in a decrease in clay porosity, hydraulic conductivity and swelling pressure (Savage, 2014). The primary safety function of the clay host rock, to limit transport of water towards the EBS and migration of radionuclides into the overlying rock formations, will only change locally, close to the tunnel liner interface, as a result of these mineral transformations. Also, the mechanism for the closure of fractures may have changed from seal-healing by montmorillonite to self-sealing by precipitation of minerals. If fractures have formed in the local host rock in this period, more time would be required for their closure and the associated reduction in hydraulic conductivity. Importantly, the alteration of clay by cementitious materials requires leaching of concrete and leaching processes are minimized with a proper choice of cement with which to manufacture concrete.

Elsewhere in the GDF in this first period, the LILW packages become increasingly saturated, having initially been dry, after a century of storage in a controlled atmosphere. LILW can include organic waste forms. Microbial processes can enhance degradation of organic waste, but they require water activities larger than 0.9 (Swanson et al., 2018), i.e., a relative humidity in air of 90%. However, there is also a wide diversity in the desiccation resistance of microbes. Sulphate reducing bacteria have been experimentally determined to require a minimum water activity of 0.96 to remain active (Stroes-Gascoyne and West, 1997). Lower relative humidity's are present during the dry storage of waste, in the operational phase of the disposal facility and early in the post-closure phase.
Microbial processes are therefore excluded during storage but corrosion of steel has already started, since corrosion of steel requires a lower degrees of saturation; the minimum degree of saturation for corrosion is 0.2 (Stefanoni et al., 2018). In the GDF, for LILW packages with steel outer surfaces, such as the 200 litre drums, the purely chemical corrosion processes become microbially enhanced as the backfill further saturates.

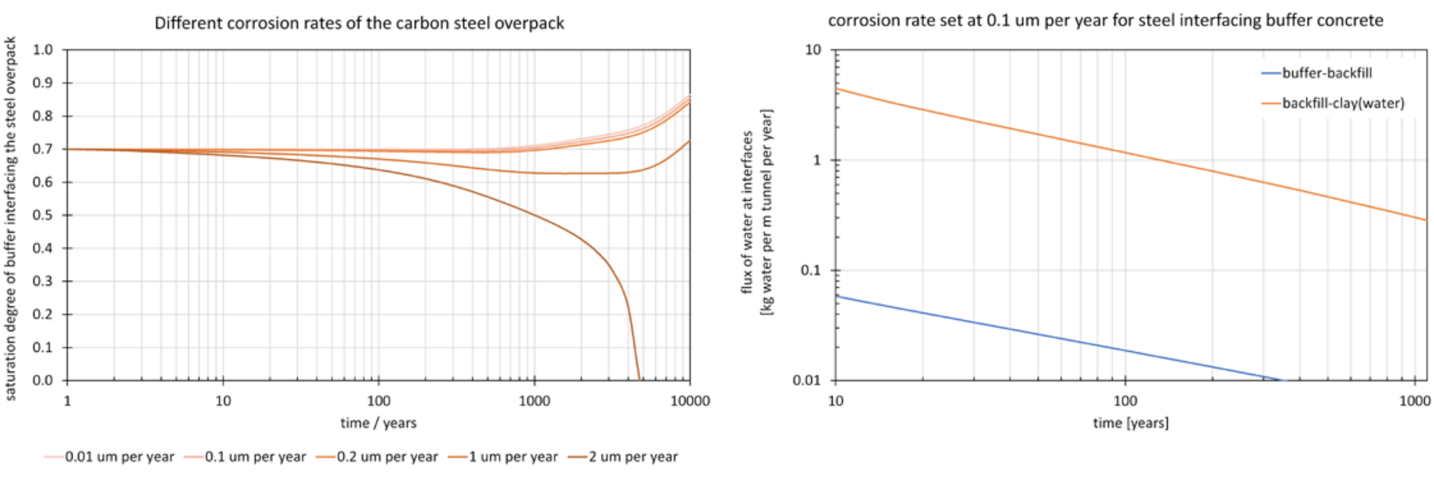


Figure 7-3: Possible corrosion rates of the carbon steel overpack in the vHLW supercontainer, which interfaces with the low permeable concrete buffer (left), fluxes of water into the backfill (21% porosity, (Blanc et al., 2024)) and buffer (13% porosity) if these concrete materials would initially be saturated for 70% and 50%. For disposal tunnel with vHLW: Interfacial areas for backfill-clay(pore water) and buffer-backfill are 4 π m^2 and 2 π m^2. No credit is taken for the low permeability of the liner, the backfill is conservatively interfacing clay pore water.

Water is consumed by anaerobic corrosion of metals such as steel but also by degradation of organic waste and by calcium-siliceous hydrates in concrete when reacting with dissolved CO_2 and bicarbonate from the EBS and host rock. In terms of the order of reaction, the consumption of water by anaerobic corrosion of steel is assumed to take place first, since this process requires a smaller degree of saturation than does the microbial degradation of waste.

Any mobilisation and movement of radionuclides within or from those waste forms that have contact with groundwater in this first phase will be at very slow rates, controlled by diffusion. The self-diffusion values for water are greater than the largest possible diffusion values for dissolved species, except hydrogen gas. Diffusion values for water are smaller in the cementitious backfill and waste package concrete than in the clay host rock. Calcium-siliceous hydrates in the cementitious components of the EBS have a pH dependent charge, allowing sorption of radionuclides to take place. This slow diffusive movement as well as potential retention together with the decay of radionuclides within the concrete, means that no significant flux of radionuclides into the clay host rock is expected in this early period.

7.1.1.2 Conditions assumed in the safety assessment

In the safety assessment, complete physical containment of the radionuclides within the HLW waste forms is assumed for the whole thermal period and there is no contact between the waste forms and water for the first 1,000 years.

For LILW, instant release of radionuclides from the waste forms is assumed to occur at the time of closure, except for depleted uranium. This is a highly conservative assumption, as can be seen from the preceding discussion, and is not the expected behaviour of the system. However, unlike HLW, where complete containment can be assured in this period, it is difficult to specify the rate at which LILW packages will degrade and a simple assumption of no containment function after closure facilitates the subsequent modelling of system behaviour. Radionuclides from LILW are therefore modelled as being available to migrate into the clay host rock immediately after closure of the GDF.

7.1.2 From 1,000 to 10,000 years

7.1.2.1 Expected behaviour

Depending on the diminishing strength of the concrete tunnel liner segments, it is expected that at some point in this period lithostatic load will be transferred to the cementitious backfill in the tunnels. Within the HLW disposal packages, the steel overpacks will continue to corrode, with a maximum corrosion rate of 0.1 µm per year, since the pH of the concrete pore water in the vicinity of the carbon steel overpack remains 12.5 (Kursten and Druyts, 2015). Complete containment of the HLW is maintained. By the end of this period, the radiotoxicity of the vitrified HLW has reduced by two orders of magnitude and is now only slightly greater than the radiotoxicity of uranium ore (see Box 2-1).

Elsewhere in the GDF, the LILW packages have become saturated and the concrete barriers, liner, backfill and waste package concrete have reacted with dissolved species from clay pore water leading to a reduction in pH of their pore waters. The anaerobic corrosion of metals, as well as microbial degradation of organic waste, both depend on the pH of concrete pore water and the permeability of the degraded concrete. The maximum flux of water available for these alteration processes is determined by the low permeability of the clay host rock, the thickness of the clay host rock, the diameter of the disposal tunnels and the distance between them. The hydrogen flux generated from the anaerobic corrosion of metals depends on the possible flux of water passing from the clay host rock and into the degraded concrete, and on processes that compete with water for anaerobic corrosion. The hydrogen can be converted microbially into CH_4 (methane) with CO_2 from clay pore water, but only in those areas of the barrier system where there are sufficiently large pores available: e.g., in the backfill made of foamed concrete.

A minority of organic waste degradation products are gaseous, such as CO_2 and CH_4. The CO_2 generated reacts immediately with cementitious minerals until no cement minerals are left or these minerals are covered with calcite rims which block further carbonation. The pH of the concrete pore water reduces in the vicinity of the organic waste by these reactions with CO_2. Methane can be converted into CO_2 by microbial processes, but the oxygen supply is too limited in the backfilled and closed GDF. Consequently, microbial oxidation of methane is only expected if it migrates to near the land surface: i.e., methane is expected to leave the degraded concrete and clay host rock unmodified.

In this period, the majority of the radionuclides have decayed to exemption levels and only those radionuclides that can be present in a gaseous form, such carbon-14 in methane, and non-sorbing, mobile, long-lived radionuclides can leave the degraded concrete. A small flux of radionuclides from the waste into the clay host rock is expected.

7.1.2.2 Conditions assumed in the safety assessment

Although the expected behaviour is that HLW does not come into contact with water and there are thus no releases in this period, the OPERA safety case considered an 'early failure' case (EBS-1) in which the carbon steel overpack is assumed to provide no containment after 1,000 years. In this COPERA safety case, a slightly larger period of 1,200 years is used, as the new calculations for vitrified HLW indicate that this is the time at which the clay host rock is no longer significantly by the waste. The radionuclide release rate from vitrified waste follows a 'dissolution' rate of glass derived from laboratory experiments in which glass is exposed to cementitious pore solution. Water consumption by the alteration of glass as well as by anaerobic corrosion of steel is neglected. Instant release of all radionuclides from spent research reactor fuel and compacted hulls and ends is then conservatively assumed to occur after 1,000 years (EBS-1 in OPERA).

After 1,500 years, the corrosion of the 3 mm thickness of carbon steel of the Konrad type II container is assumed to be completely corroded, based on corrosion rates of more than 2 µm per year (Filby et al., 2016). A passive film may not have been developed at such high corrosion rates since the corrosion rate continues to decrease (Kursten and Druyts, 2015). The table in Text Box 6-3 provides the long-term corrosion rates assuming that the rate is constant and no longer decreases (Deissmann et al., 2021).
The radionuclide release rate from the depleted uranium wastes is modelled to be solubility limited, for the base case. Radionuclides from LILW were already assumed to be instantaneously released from the whole EBS in the period from closure until 1000 years, so the clay host rock is assumed to be the sole containment barrier for LILW in this period.

7.1.3 10,000 years to 100,000 years

7.1.3.1 Expected behaviour

In this period, any changes in a disposal tunnel with heat generating HLW is expected to be entirely devoted to the HLW supercontainer composed of carbon steel embedded in a concrete buffer. The pH of the concrete pore water in the vicinity of the carbon steel overpack is expected to have been reduced, possibly to a value of < 11.7, by reactions with dissolved CO_2, bicarbonate and magnesium. Calcium-silicate hydrates are charged cementitious minerals that act as ion exchangers. The charge depends on the pH (Pointeau et al., 2006) and the cementitious minerals have a positive charge at a pH >11.7 and a negative charge at a pH <11.7 (see Figure 6-1), promoting the sorption of iron from the corroding HLW overpacks onto the altered concrete buffer. Sorption processes are usually faster than precipitation processes, suggesting that iron mobilised from the overpack will tend to sorb rather than form precipitated phases. Consequently, control of the steel overpack corrosion rate by the sorption rate of dissolved iron needs to be considered when the pH of concrete pore water reduces to < 11.7. Realistically, the corrosion and alteration rates are not expected to be high enough to allow access of pore water to the inner stainless-steel container and the vitrified waste form in this period. However, to be conservative, the higher corrosion rates found for carbon steel exposed to clay may be more appropriate at this stage, although the inner stainless canisters are still expected to remain out of contact with pore waters, even if the carbon steel overpack corrosion is faster: for example, if corrosion rate has increased from 0.1 µm per year (see Text Box 6-3) to 1 µm per year, a value measured for carbon steel interfacing with bentonite (Johnson and King, 2008).

The buffer interfaces the cementitious tunnel backfill, which is assumed to have already lost its strength by ingress of dissolved species from the clay host rock. The outer region of the concrete buffer in the HLW supercontainer also becomes affected by ingress of dissolved species from the clay host rock, but the rate of ingress is much smaller, due to the smaller porosity compared to the tunnel backfill.

The inner regions of this buffer are affected by the uptake of dissolved iron arising from the corrosion of the carbon steel overpack. The concrete buffer is also continuously compressed by the lithostatic pressure of the host rock. Although the unaltered buffer has sufficient mechanical strength to prevent deformation, it is assumed that the chemically affected parts lose their mechanical strength so that the remaining thickness of the unaffected concrete buffer becomes unable to withstand the lithostatic load. Consequently, the eventual failure of the remaining, now extensively corroded, carbon steel overpack occurs, due to a combination of chemical corrosion and the lithostatic load. This failure could be by corrosion penetration or by fracture, or a combination of both. The fracture of the carbon steel overpack also leads to immediate fracture of the stainless canister which is not designed to withstand high loads and contact between water and the vitrified waste form becomes possible. In the current supercontainer design for vitrified waste, the hydrogen generation rate by steel is too small to lead to perturbation of the clay host rock (see Text Box 6-3).

Fracture of the stainless-steel canister is envisaged to occur where there is internal void space near the top of the canister, above the vitrified waste form shown in Figure 6-11. There will be some helium gas present in this volume, produced by decay of actinides, since helium diffuses very quickly through the vitrified waste form at disposal temperatures (Chamssedine et al., 2010). This will then - like hydrogen gas - dissipate through the degraded EBS into the clay host rock. In the period up to 35,000 years, the radionuclide Am-241 is mainly responsible for the helium generation; in each vHLW canister, about 3.5 mol helium can be generated by this radionuclide. Assuming 4 mol and an empty volume of 22 litres at the top of the canister in Figure 6-11 leads to an additional pressure of 0.5 MPa. This pressure of helium is very small compared to the gas entry pressures of the concrete buffer of 49 MPa and foamed concrete of 12 MPa (see Text Box 6-1). But these entry pressures are for intact concrete and we assume that the concrete is degraded. Nevertheless, dissipation of gas ensures that helium pressures are lower at as it reaches the clay host rock, and intact Paleogene clays such as Boom Clay have an entry pressure of 4.9 MPa (Levasseur et al., 2021): i.e., still above the calculated helium generated pressure. Perturbation of the clay host rock by the helium gas collected in the empty volume in this period, is thus considered to be negligible.

There may be contact between the vitrified waste and pore water, passing from the clay through the degraded concrete tunnel liner, the tunnel backfill and the supercontainer buffer, with a pH higher than 10. An alteration layer is always generated on vitrified waste , which limits further dissolution, whatever pH it is exposed to.
This layer is composed of hydrated glass on top of unaltered glass, and clay minerals or zeolites on top of the hydrated glass.
The incorporation of less soluble elements such as aluminium, iron and zirconium from the waste, determines the stability of the clay minerals and zeolites formed in the alteration layer. However, these layers provide less effective protection from leaching under the high pH conditions that prevail in the EBS.

These layers may also provide less effective protection in the vicinity of steel. Chemisorption of iron occurs on the clay minerals that are generated by the alteration of the vitrified waste in the vicinity of corrosion products. The sorption of iron changes the newly formed clay minerals into iron phyllosilicates that are less protective. Experimental evidence shows that ion exchange reactions can be the dissolution rate controlling processes. Corrosion products have the same influence (Van Iseghem et al., 1992; Vernaz et al., 1996). Corrosion products such as chromite (Souza et al., 2012) and magnetite Kim (2012) in (Eisele et al., 2005) are also negatively charged at pH conditions representative for disposal and therefore also preferentially sorb the less soluble elements.

The glass alteration process requires water, whose influx into the fractured vHLW canister is restricted by diffusion through the altered concrete buffer, backfill, liner and clay host rock.
The consumption rate of water depends on the many different type of clay minerals that can be formed. Kaolinite has the smallest water content and its formation consumes the smallest amount of water (see Table 5-3). For each SiO_2 molecule from the glass, $^4/_5$ water molecule is consumed. Assuming an alteration rate of 3 µm per year (Gin et al., 2022) would generate a slightly smaller water consumption rate than used in Table 6-2. The water consumption rates have also been used in the calculation of the evolution of the degree of saturation of an initially saturated concrete buffer in Figure 7-4. First, it can be seen that the concrete buffer interfacing a carbon steel overpack with a corrosion rate of 0.1 µm per year remains saturated. A consumption rate of only 0.03 g water per m tunnel length with vitrified waste form per year also ensures that the concrete buffer remains almost saturated (see green line in

Figure 7-4). A thirty times higher consumption rate of water to dissolve glass, would dry the concrete buffer in the vicinity of the waste form (see red line in Figure 7-4). Dissolved silicon from the vitrified waste in the concrete buffer cannot migrate away and dissolution of the glass stops after 900 years. The dissolution reaction can only continue after sufficient water has been replenished.

Alteration of the vitrified waste also requires water. In experiments to study this, typically a piece of solid (S) glass would be put in a large container of liquid (L): e.g., an aqueous solution representative of concrete pore water. The representativeness of such experiments for the encapsulated vitrified HLW in a concrete buffer can be questioned and more realistic assumptions of how water can come into contact with the vitrified waste need to be made.

Contact between pore water and the vitrified waste form occurs after fracture of the steel overpack and the stainless-steel container. Only water that passes through the fracture is available for alteration of the vitrified waste: i.e., only a part of the surface area of the vitrified waste form may be in contact with pore water. If the strength of the concrete is reduced by the decalcification process, then the permeability of the concrete buffer may have increased. But the dissolved silicon concentration in equilibrium with the decalcified CSH minerals has increased. At a pH lower than 11.7, the silicon solubility of the decalcified CSH mineral is larger than the silicon solubility of glass (see Table 6-2). Consequently, the vitrified waste is not expected to dissolve in the vicinity of a decalcified concrete buffer.

If all steel in the overpack and canister has been completely corroded, the total surface area of the vitrified waste form can be in contact with pore water. An increase in water permeability of the concrete buffer is less likely in case of completed corrosion of the carbon steel overpack and stainless steel canister. The calculation in Figure 7-4 assumes completed corrosion of steel: i.e., the total surface of the vitrified waste form is in contact with water. It shows the degree of saturation as a function of time with the associated water consumption rate of the glass alteration rate used in OPERA at two locations of the concrete buffer: in the vicinity of the vitrified waste and 0.5 m away from it, assuming the same saturated diffusion value for water as obtained from waste package concrete in Text Box 6-1. Taking these effects into account indicates that smaller glass alteration rates than used in the OPERA programme are expected to be representative for a normal evolution scenario.

The vitrified waste will have achieved a similar radiotoxicity to uranium ore after about 20,000 years (see Box 2-1) and this continues to decline over the following tens of thousands of years in this period.

7.1.3.2 Conditions assumed in the safety assessment

For the HLW supercontainers, the calculated period in which the carbon steel overpack interfaces with the concrete buffer at a pH of 12.5 is from 1,000 years until 80,000 years (Kursten and Druyts, 2015). The alteration rate of the waste form depends strongly on the pH. For simplicity, release of radionuclides was assumed in which the pH was 12.5: after 1,000 years, 35,000 years and 70,000 years (Schröder et al., 2017b). In OPERA, all concretes in the multi-barrier system were assumed to have no mechanical strength; the carbon steel overpack alone was assumed to sustain the mechanical load from the underground.

The upper bound of the glass dissolution rate in OPERA was derived from experiments with a pH of concrete pore water of 13.5, rather than 12.5. In addition, the initial glass alteration rate was assumed to be representative for the long-term. The water consumption rates would then be 300 times larger than those used to produce the green curves in Figure 7-4, and 10 times larger than for the red curves.

The amounts of radionuclides released from the vitrified waste are derived from glass alteration rates obtained in laboratory experiments. Instant release is assumed for spent research reactor fuel and compacted hulls and ends.

The releases of radionuclides from depleted uranium are assumed to remain solubility limited. Radionuclides from LILW were already assumed to be instantaneously released in the period from closure until 1,000 years.

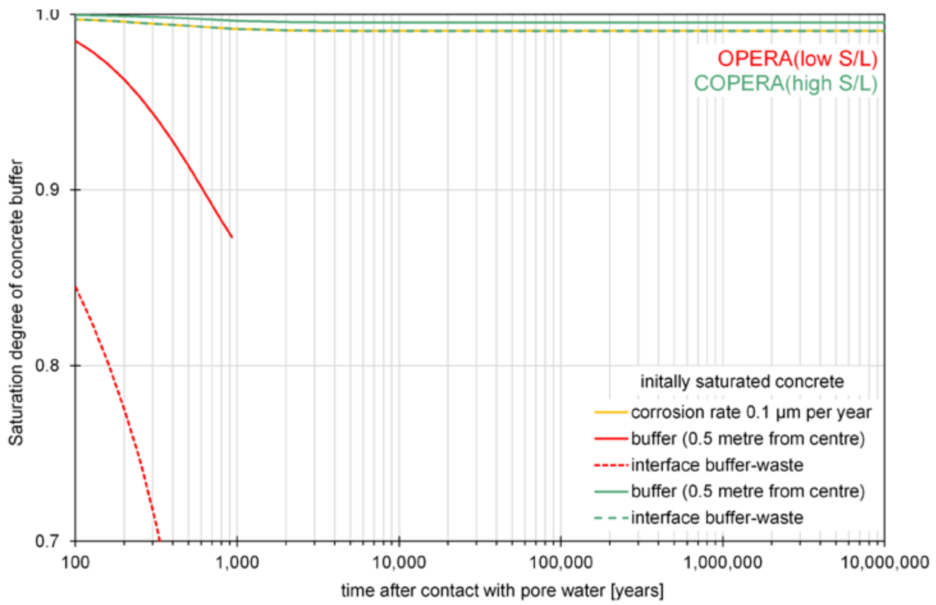


Figure 7-4: Evolution of the degree of saturation of the concrete buffer in the vHLW supercontainer (outer diameter 2 m, inner diameter 0.5 m) interfacing the carbon steel overpack with the geometry shown in Figure 4-2 and the relationship to determine diffusion values as a function of the degree of saturation as defined by Millington and Quirk (1961).

7.1.4 From 100,000 years until 1,000,000 years

7.1.4.1 Expected behaviour

Natural climate change, eventually leading to an ice age after 100,000 years is postulated (Rochelle and Long, 2009). The Dutch territory could be covered by an ice sheet, especially the Northern part of the Netherlands (see section 5.2.1), which would increase the mechanical load on the EBS. This makes it more likely that the carbon steel overpack in the HLW supercontainers becomes fractured if it was not fractured earlier. We assume that fracturing of overpacks could be staggered over many thousands of years so that access of pore water to the HLW waste forms and the start of release of radionuclides will be spread over time.

Several elements with safety-relevant radionuclides (caesium, niobium, uranium in section 5.1.6.4) have higher concentrations in the clay host rock than in (clay pore) water. These examples are illustrative of the impact of chemical containment processes such as sorption, and can be further analysed to determine the expected migration behaviour of radionuclides in the clay host rock.
This was done in the Swiss programme, showing that niobium-94 and caesium-135 hardly leave a bentonite buffer with a thickness of 0.7 m (NAGRA, 2002). Both bentonite and the Paleogene clays contain montmorillonite (a swelling clay mineral), but with a higher content and a higher cation exchange capacity, e.g., 100 meq/100 gram for bentonite compared to 18.5 meq/100 gram for Boom clay (see section 5.1.6.4). However, a bentonite thickness of 0.7 m is much smaller than the 100 metres thickness of clay host rock, so that caesium-135 and niobium-94 would also be expected not to leave the clay host rock. Half-lives are important in the determination of the migration distance. The calculated uranium migration distance of uranium-233 (half-life 159.2 years) is also too small to leave the 0.7 m thickness bentonite buffer, but uranium-235 (half-life 707 Ma) requires, in addition to the bentonite buffer, a 40 m thickness of Opalinus Clay for complete containment after 10^9 years (NAGRA, 2002), which is beyond the timescale considered in this narrative. Uranium-238 (half-life 4500 Ma) would take hundreds of millions of years to diffuse across Opalinus Clay.
Only the non-sorbing long-lived radionuclides selenium-79, chlorine-36, iodine-129 and (organic e.g., contained in methane) carbon-14 are released into the biosphere (NAGRA, 2002).

Although cementitious materials dominate the overall volume of the EBS, the alkaline disturbed zone in the clay host rock has been calculated to remain within three metres of the tunnel after 100,000 years (Wang et al., 2010). Consequently, the majority of a clay host rock with a thickness of 100 m remains unaffected, and unmodified natural physical and chemical containment properties are representative for assessing the safety of the disposal system.

After a million years, residual immobile and long-lived radionuclides will remain within the EBS and clay host rock in the vicinity of the EBS - including uranium-238, the main radionuclide of depleted uranium. Inexorable processes of geological erosion such as those taking place during the retreat of an ice sheet (see section 5.2.1) over hundreds of million years will ultimately disperse these residual elements into new sediments and rocks, as occurs with naturally occurring ore bodies.

7.1.4.2 Conditions assumed in the safety assessment

All barriers are assumed to be unaffected by climate change.
In OPERA, three different climate states (warm, moderate and cold - with and without ice cover) were used to calculate the travel times from Boom clay host rock at around 500 m depth to the biosphere. The travel time was calculated to be about 30,000 years for warm and moderate climates. Cold climates have larger travel times: 70,000 years with ice cover, 120,000 years without ice cover (i.e., only permafrost) (Schröder et al., 2017b; Valstar and Goorden, 2017). Conservatively, sorption and other chemical containment processes were excluded in the formations that surround the clay host rock. For the assumptions for the clay host rock, the parameters for sorption and diffusion used in OPERA lead to significantly larger migration distances of radionuclides such as caesium-135, niobium-94 and uranium-233, 235 and 238 than in the Swiss case (NAGRA, 2002).

7.2 Alternative evolution scenarios

Alternative evolution scenarios comprise a set of cases in each of which the normal evolution scenario is changed in a specific way. They represent conditions that are considered much less likely to occur than those of the normal evolution scenario, including, for example, unexpected or highly unlikely processes. Calculation of releases in alternative scenarios are foreseen after sufficient confidence has been obtained in the assessment basis and when region-specific normal evolution scenarios are available. In COPERA (2020-2025), priority has been given to improving understanding of the assessment basis of the safety assessment, e.g., by further identification and study of key processes in the normal evolution scenario and by further research projects for verification of the input used in OPERA.

7. 2.1 Failure to close the GDF adequately

A poor sealing scenario was considered in the Belgian programme for the second Safety Assessment and Feasibility Report SAFIR-2. It was assumed that the shafts, transport tunnels and disposal tunnels in the GDF are poorly sealed, e.g., due to construction errors, poor construction materials or errors in the design and testing of the facility and/or the seals. This might result in the formation of a hydrological connection between a sandy formation overlying the host rock and the transport and disposal tunnels.
If pore water pressure in the clay host rock is higher than in the tunnels, water can be squeezed into them, inducing flow upwards through the poor seals of the GDF and into the overlying sandy formation. However, the slow processes of degradation of the engineered barriers and mobilisation of radionuclides from the wastes would be the same as those in the normal evolution scenario. In the current design, all types of concrete, as well as the reconstituted clay that is to be emplaced in the transport tunnels, have a lower permeability than the clay host rock. But squeezing water into tunnels that allow flow is only possible if some engineered barriers have a higher permeability than the clay host rock. This scenario will be further assessed in the future on a region-specific basis. The pore water pressure in the clay host rock increases during the presence of an ice sheet and, during its retreat, which can be retained for some subsequent period due to its low permeability.

7.2.2 An Excavation Damaged Zone is not healed

For a GDF constructed in Boom clay, there is overwhelming evidence of self-healing of excavation induced fractures, as explained in section 5.1.6.3, so that the permeability of the host clay may be assumed to remain at the values of the intact, undisturbed conditions at the start of the post-closure phase. This is also assumed for the safety assessment. An increase in salinity of the pore water of a Paleogene clay is expected to make the clay stiffer, which enhances the feasibility of the construction of the tunnels in a GDF but may also increase the fracture density in the Excavation Damaged Zone (EDZ) and lead to an increase in its size. There is a preferential pathway to the shafts if the EDZ has a higher permeability than the permeability of the undisturbed surrounding clay host rock. Calculations in more than one dimension are required for assessing the impact of an EDZ, since the pathway is along the shaft.

7.2.3 Abandonment of the GDF

The disposal facilities and operations will be designed to be fail-safe during all steps of the disposal process. This means that, even in case of abandonment of the GDF without proper closure, the waste will not suddenly be released to the surface and present an immediate threat to the environment. Nevertheless, an abandoned and incomplete GDF will not provide the same level of containment and isolation as intended, and this possibility needs to be analysed. Unlikely events that might lead to abandonment of the facility include serious economic and regulatory malfunction, war or other national disasters and major mining or underground construction accidents, without proper response. Temporary abandonment would be a recoverable event. In a highly unlikely worst case, involving long-term societal breakdown, events could lead to permanent abandonment of the GDF, without proper closure.
Such an event was considered in the second national programme, CORA, where it has been assumed that abandonment could lead to flooding of unsealed galleries and earlier exposure of the containers and the wastes to larger volumes of water, compared to the normal evolution scenario, followed by flow and diffusion through the remains of the underground infrastructure (tunnels, shafts) and earlier release of radionuclides into an overlying aquifer or the biosphere. No engineered containment, apart from that provided by the vitrified waste form, was assumed in the calculation of the radiological hazard as a function of time (Grupa and Houkema, 2000). Consequently, the potential radiological exposure would be smaller with the present disposal concept in which the vitrified waste is encapsulated in a carbon steel overpack and concrete buffer.

7.2.4 Anthropogenic greenhouse effect on future climate change

This scenario considers changes in the overlying aquifers due to global warming of the atmosphere and the resulting radiological impact. The greenhouse effect may cause the present moderate climate to evolve into a warmer, more Mediterranean climate over the coming centuries. In the Belgian SAFIR-2 safety study, the greenhouse effect was assessed to have only a small impact on the disposal system, affecting mainly the biosphere and, to a lesser extent, the hydrogeological environment. The scenario indicated no direct impact on the clay host rock or the near field, and no radionuclides were released into the aquifer during the first 5,000 years. Therefore, that scenario was excluded from further study in SAFIR-2.

This scenario could lead to an increased risk of flooding of the GDF site as a consequence of rising sea-level, if no measures are taken to protect it during operations, or immediately post-closure. As a result, brackish water might infiltrate the shallow subsurface, or the GDF itself, if it has not yet been closed. An important difference from the abandonment scenario is the timing of radionuclide releases to the geosphere and the biosphere and the prevailing biosphere conditions at the time of release, as impacts might occur well after the greenhouse effect has come to an end. This scenario could also consider enhanced transport through the aquifer system compared to the normal evolution scenario and changing chemical conditions, especially in the aquifer system.

7.2.5 Faulting affecting the geological barrier

Site characterization will screen carefully for the presence of major faults transecting the GDF or the surrounding host rock. However, the possibility of undetected deep faults being present and being reactivated, propagating upwards through a Paleogene Clay to the surface, cannot be completely excluded at this stage, before any siting studies have been performed. The fault scenario considers the consequences of a tectonic fault through the host rock and the GDF, which has the potential to form a preferential flow path for radionuclide migration. Owing to the plasticity of the Paleogene Clays, a sharply defined fault plane might not be formed. Instead, the clay will deform plastically over a broader zone, resulting in a change in the hydraulic and mechanical properties of the clay within the fault zone compared to those of the undisturbed clay.
The SAFIR-2 study assumed that a fault forms through the GDF, affecting the containment and isolation capacity of the geological barrier. The potential changes in hydraulic properties in the faulted rocks and possible mechanical processes affecting the waste packages need to be evaluated.

7.2.6 Intensified glaciation

During the past Quaternary glacial periods, permafrost developed intermittently in large parts of northern Europe where periglacial conditions prevailed, being estimated to have reached depths ranging from a few tens of metres in the case of the Mol site in Belgium ((Marivoet et al., 2000) to 100-300 m in the Netherlands, Germany and northern England (Shaw et al., 2013). Future, deep permafrost development could have direct impacts at disposal depth, including possible impacts on the EBS if it were able to penetrate so deeply. Even if the GDF is at a depth greater than permafrost development, impacts on the host rock and indirect effects such as brine formation and migration, intrusion of freshwater from melting permafrost or gas hydrate formed beneath the permafrost layer (Rochelle and Long, 2009), and cryogenic pore pressure changes associated with volume change during the water-ice phase transition, could affect the integrity of the geological barrier. These processes might affect the transport processes of any released radionuclides. In addition, an intense glaciation with thick ice sheet development over the GDF site could lead to localised deep erosion. This possibility is discussed in Chapter 5.
The intensified glaciation scenario assumes the presence of a massive ice sheet producing meltwater, deep subglacial erosion and thick permafrost development in front of the ice sheet. Compaction of a Paleogene clay by glacial loading resulting in increased pore water movement was addressed as a what if scenario in OPERA and is put in this safety case as an alternative scenario.

This scenario was addressed in the second programme CORA (Wildenborg et al., 2000) by assuming a thickness of the ice sheet of 1 km, as explained in Chapter 5. Glaciation in the next 100,000 years is considered highly unlikely.

7.3 What if

A 'what if' scenario can be used to test and illustrate the contribution to the containment and isolation provided by the individual barriers in the multibarrier system and the robustness of the system. These scenarios represent entirely hypothetical situations but highlight key sensitivities and points of focus for optimising the GDF.

7.3.1 Early failure of a HLW package

Early failure of a HLW package, which might be caused by a defective overpack, was identified in OPERA. Early failure (EBS-1) after 1,000 years was assumed. In some national safety assessments, this scenario is a central case for analysis rather than a 'what if' case. Despite considerable advances in manufacturing quality control over recent decades, it could be difficult to ensure complete integrity of each of the large number of HLW packages in the GDF at the time of emplacement. In COPERA, a radionuclide specific calculation has been made for one long-lived radionuclide after overpack failure at the end of the calculated thermal phase of 1,200 years.

7.3.2 Nuclear criticality

Nuclear criticality leading to excessive heat production was identified in OPERA as a process that should be evaluated. Criticality is addressed for spent research reactor fuel in Chapter 6. The recent analysis of the Oklo natural reactor in Gabon (Bentridi et al., 2011) emphasizes the need for removal of fission products to continue the chain reaction. This removal is easy for a sandy formation but more difficult for a clay formation and concrete. The multibarrier system with the low permeable clay and concrete may already be optimized to prevent nuclear criticality.

7.4 Human intrusion

Although the footprint of the GDF is relatively small, future inadvertent drilling at the disposal site cannot be excluded as eventual loss of memory of the facility is assumed to be inevitable. At present, deep drilling projects would generally be guided by the results of geophysical surveys to establish the nature and geometry of the geological environment at depth. Such surveys are likely to identify an anomaly caused by the presence of the GDF and thus raise questions, if the presence of a GDF is no longer known about.
Even if inadvertent drilling proceeds, its risks are somewhat minimized by the design of the disposal facility and the waste packages, which are disposed of in segregated, small waste units. This compartmentalisation ensures that the likelihood of drilling intrusion (before the drillers identify and understand the hazard) would be limited to a single waste container and not the whole waste volume.

Drilling projects are expensive and only done if natural resources (minerals, hydrocarbons geothermal energy) are expected to be present. The associated risks are then determined by the technology used. So far, only the direct radiological risks for workers performing drilling activities have been considered and no radiological risks for the public. The human intrusion risks will be further assessed in a future safety case.

7.4.1 Clay as a resource

Although clays have numerous engineering uses, abundant clays outcrop at many places in the Netherlands and neighbouring countries to satisfy all current uses. Collecting clay at depth for these purposes is not necessary or economically feasible. Human intrusion to extract clay host rock is therefore considered unlikely.

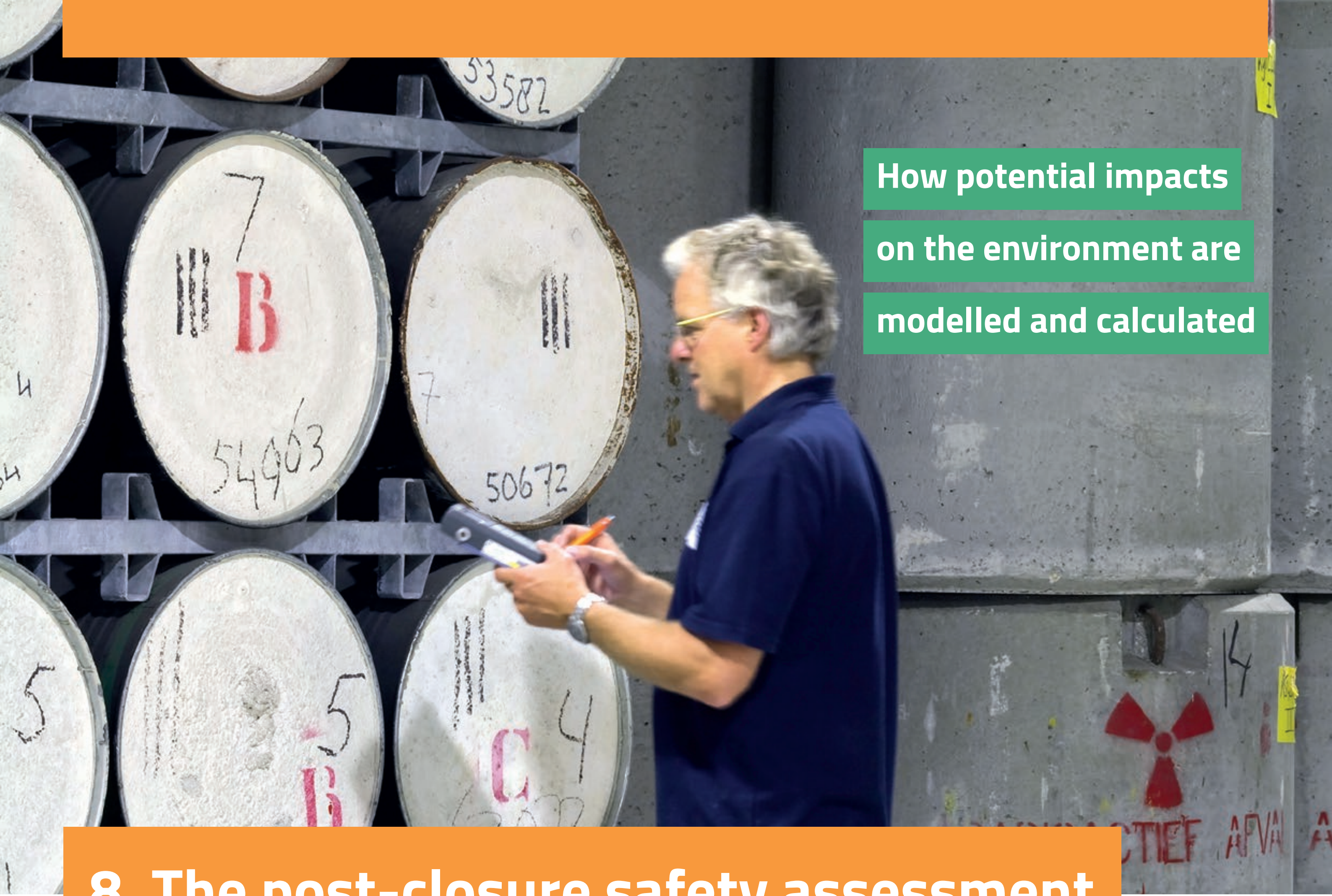
7.4.2 Extraction of groundwater

Extraction of groundwater from clay is not economically feasible due to its low permeability, but almost all the Paleogene clays are surrounded by Paleogene sands, which are highly permeable formations and can therefore be used to extract groundwater.
As explained in Chapter 5, Paleogene formations at suitable disposal depth generally contain more saline than brackish water, perhaps even as saline as seawater. If water supplies were scarce and desalination were ever to be used in the Netherlands, extraction of water from the sea is economically more feasible than extraction of saline water at depth.

7.4.3 Underground extraction and storage of heat

In the Netherlands, geothermal wells locally extract the heat from sandy formations at several km depth. After a few decades, the underground is locally depleted in heat at that spot and underground extraction of heat is no longer economically feasible.
It will take thousands of years to re-equilibrate the thermal state at such locations before extraction might be resumed. The use of underground formations to store heat (or provide refrigerated containment spaces) currently takes place at depth < 200 m, above the minimum depth for disposal of HLW but potentially of relevance for disposal of LILW. Similarly, the use of ground-source heat pumps for domestic and industrial heating is becoming common and also involves drilling boreholes to depths of up to 200 m.

The engineered barriers of the GDF utilise high strength concrete for the liner and concrete buffer. Drillers inadvertently hitting a GDF tunnel will be alerted by the difficulty of drilling through this concrete, an unexpected material in the relatively soft formations they would be expecting at this depth in the Netherlands.
Also, drill cuttings other than clay that arrive at the surface will alert the drillers.



How potential impacts on the environment are modelled and calculated

8. The post-closure safety assessment

A central part of a Safety Case is the modelling and calculation of the potential impacts of the disposed wastes on the environment, for long times into the future. This is the function of the safety assessment presented in this report, which involves developing conceptual and calculational models of all significant processes and quantifying the necessary parameter values needed to calculate the evolution of the multibarrier system as a function of time. This Chapter first briefly summarises the conceptual models that we have used to calculate the potential impacts of radionuclide releases in terms of radiation doses to people. The results of the safety assessment calculations are presented and compared with certain yardsticks in order to optimise the disposal concept, steer the development of knowledge and guide research.

In COPERA, a full safety assessment such as that presented in OPERA has not yet been carried out, since work to improve the modelling and to extend the database is in progress. Instead, selected results from the OPERA safety case (Verhoef et al., 2017) are presented again in this report and compared with updated results that illustrate the impacts on safety of updating key parameter values based on enhanced understanding of the processes involved. The comparisons have been done for those radionuclides which contribute most to the calculated potential doses: those from the two main forms of HLW in the inventory, vitrified HLW and compacted hulls and ends. Vitrified HLW has been studied in the EURAD ACED project and new results have led to a lower estimate of the alteration rate of the vitrified waste form. Taking account of the properties of representative concrete as elucidated in EURAD ACED and MAGIC may further reduce the estimated accessibility of water required for the degradation or alteration of the waste forms, leading to more gradual releases of radionuclides into the clay host rock than were calculated in OPERA. Paleogene clays contain natural radionuclides and chemical analogues of radionuclides in the waste. Preliminary studies on the transport and retention mechanisms of these naturally occurring elements have led to a revision of the data to be used in post-closure safety assessments.

8.1 Modelling approach

The model used in OPERA to calculate the movement of radionuclides in the multibarrier system and the potential health effects of released radionuclides as a function of time distinguished four compartments: the engineered barrier system (EBS), the clay host rock formation at 500 m depth, an aquifer system and the biosphere (Meeussen and Grupa, 2017; Schröder et al., 2017b). The parameters that determine the calculated concentrations of radionuclides in the natural and engineered barriers are shown in Figure 8-1. The biosphere acts as a receptor for any radioactive elements released from the engineered barriers, which move upwards through the natural barriers in a normal evolution scenario.

In COPERA, updated properties of concrete are used to assess the potential water consumption rates of the waste and the engineered barriers as they degrade, leading to radionuclide releases. This approach has been applied to assessing the behaviour of the principal HLW material in the GDF inventory - the vitrified HLW supercontainers. We have considered how vitrified waste glass is transformed into clay minerals and how metals are transformed into metal-oxides, metal-hydroxides or metal-hydrides. The water to do this needs to be supplied by the clay host and transported through other engineered barriers to the waste form, as discussed in Chapter 7. The time and rate at which water can access the waste control the time for radionuclides from the waste form to enter a dissolved state. Safety assessment calculational results imply that, if the alteration of waste forms were to consume more than 0.1 kg water per metre length of disposal tunnel per year, then the clay host rock and the concrete would dry out and any alteration process would stop.

8.1.1 Concentration gradients drive diffusive transport

If radionuclides are dissolved and mobilised from the waste into solution in porewaters in the barrier system, then the concentration gradients in the fluids present in different parts of the multibarrier system will determine the direction in which they can migrate. However, some of the chemical elements in the waste are also present naturally in pore waters in the host clay and surrounding formations. Their initial natural concentrations were conservatively assumed to be zero in OPERA (Meeussen and Grupa, 2017). This maximises the outwards concentration gradient and implies that radionuclides from the waste would move towards the natural barriers. However, as discussed in Chapter 5, the concentrations of some naturally occurring elements that have radioisotopes in the wastes are not zero, so that a more realistic model will include a lower driving force for diffusion of radionuclides out of the waste. In this study, we initially neglect this fact, as was done in OPERA, but then compare the impact of the updated modelling and data on the performance of the engineered barriers. Later, in Section 8.4.1, we elaborate further on the impact of the natural concentrations of relevant elements in the clay host rock.

8.1.1.1 Radionuclide concentrations in the OPERA EBS

For LILW, the concentration of man-made radionuclides in the EBS is determined by assuming that all radionuclides are available to enter solution immediately after closure of the GDF and then dividing the radionuclide content in the waste form by the pore volume in the EBS (Schröder et al., 2017b). Almost all the radionuclides thus enter into solution in the pore waters of the cementitious materials, which are assumed to be saturated with water just after closure of the GDF. Exceptions are uranium, thorium and neptunium, since the solubility limit of these elements would be exceeded in this instantaneous release model. (A solubility limit of 10^{-5} mol/l was used for uranium in the EBS in OPERA, more than 1000 times larger than obtained by experiments (Chapman and Flowers, 1986)).

For depleted uranium wastes, spent research reactor fuel and non-heat generating HLW, a similar approach as that used for LILW

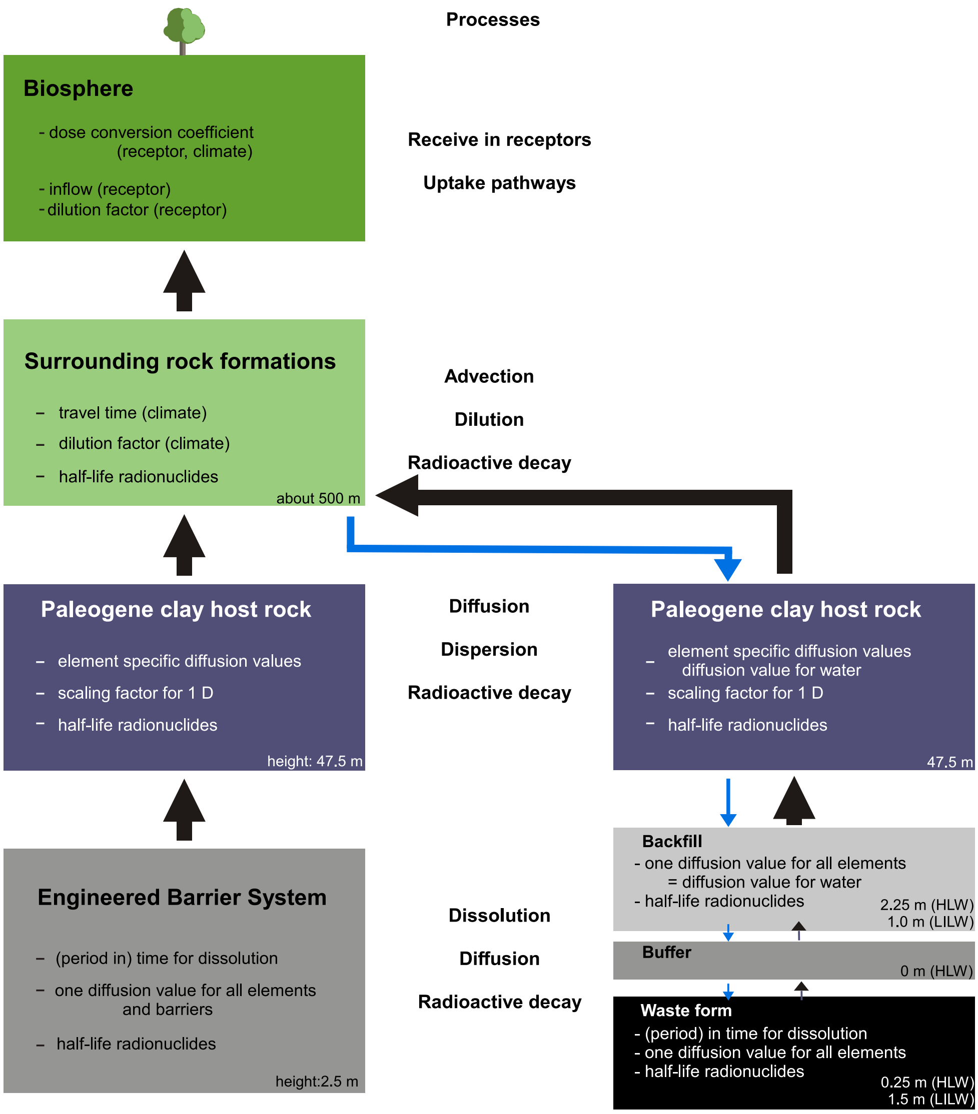


Figure 8-1: Schematic overview of parameters and processes used in the components and subsystems of the safety assessment model (left: OPERA), (right: updated approach developed in COPERA). Black arrows show modelled transport of radionuclides, blue arrows show modelled transport of water.

is applied, except that radionuclides are assumed to be available for dissolution only after the assumed failure time of their respective disposal packages (Schröder et al., 2017b).

For vitrified HLW, the concentrations of radionuclides in the EBS are determined by assuming an alteration rate for the glass combined with an estimate of the area of glass surface available for leaching, after the assumed failure time of the supercontainer. Only in the case of vitrified HLW, is the release of radionuclides assumed to be time dependent rather than instantaneous.

8.1.1.2 Radionuclide concentrations in the COPERA HLW EBS

For HLW, the same approach is used to that in OPERA, except that the volume in which the radionuclides are assumed to be dissolved is assumed to be in a single volume with all EBS but also to the volume of the waste form. Radionuclides released from the waste form, then enter the cementitious materials (supercontainer buffer, tunnel backfill and tunnel liner).

For vitrified HLW, time dependent alteration of the glass is assumed, and is determined by results obtained from laboratory experiments in which a non-radioactive glass analogue for the vitrified waste form is exposed to a solution. The deduced glass alteration rate depends strongly on the ratio between the solid and solution, as explained in Chapter 6. The glass alteration also consumes water. In section 7.1.3.1, the evolution of the degree of saturation of the concrete buffer was calculated with the glass alteration rate used in OPERA leading to a water consumption rate of 9 gram per m tunnel length per year and the lower glass alteration with an associated consumption rate of 0.03 gram water per m tunnel length per year (see Figure 7-4). Only the lower glass alteration rate does not lead to dehydration of the concrete buffer in the supercontainer and is therefore feasible on the long-term. Anaerobic corrosion of steel also consumes water. The water consumption rates that would lead to lower water diffusion values due to drying of the concrete are highlighted with the calculated results.

8.1.2 Geometry

Geomechanical constraints during the construction of the GDF at a depth of 500 m require the distance between the disposal tunnels to be 50 m. The thickness of the clay host rock formation is assumed to be 100 m, with the tunnels of the GDF situated in the middle of the clay host rock formation.

The model in OPERA is a one-dimensional pathway through different compartments. The vertical dimensions used in the pseudo 2D model are 2.5 m for the EBS (Schröder et al., 2017b) and 47.5 m for the surrounding clay host rock (Meeussen and Grupa, 2017). The volume of the EBS is however small compared to the volume of clay host rock, and there is thus extensive horizontal dispersion of any mobilised radionuclides as they migrate through the host rock. Concentrations of radionuclides at the outer regions of the clay will thus be much smaller than they are in the clay in the vicinity of the EBS. This horizontal dispersion is determined from steady state concentration profiles made in 2 dimensions. This dispersion in the clay host rock further away for the EBS is approximated in 1D dimension resulting into a pseudo 2D model.

The waste families are emplaced in five separated sections of the GDF: vitrified HLW, spent research reactor fuel, non-heat generating HLW, depleted uranium and LILW. In OPERA, the amounts of radionuclides moving into the clay host rock were calculated for each disposal section as a function of time and summed up to obtain the total radionuclide releases into the clay host rock and the outwards flux into the overlying aquifer system. The aquifer system geometry is based on the national hydrogeological model (LHM) for groundwater management. This LHM-model was extended for OPERA, since the hydrogeological base is not deep enough in most geographical areas of the Netherlands. In the radionuclide release calculations, a non-retarded particle is tracked from the Boom clay at 500 m depth towards the biosphere. Conservatively, sorption and other chemical containment processes were excluded in the formations that surround the clay host rock since the geometries and clay content of these formations will be site-specific. A range of travel times and dilution factors were calculated for three different climate states: Temperate, Mediterranean and Boreal (Valstar and Goorden, 2017; Valstar and Goorden, 2016).

8.1.3 Diffusion

Diffusion values for radionuclides in the different components of the EBS and the clay host rock are used in the safety assessment. The same diffusion value is used for all radionuclides in the EBS but the diffusion values in the clay host rock are element-specific. All radionuclides considered here are dissolved as negatively charged species (anions) or positively charged species (cations or cationic dissolved complexes). Self-diffusion values for water are the largest possible diffusion values for dissolved species, except for some gases such as hydrogen. For anions, the diffusion value is assumed to be reduced, because fewer pores are available for diffusion for these negatively charged species. For positively charged cations, ion exchange can take place with non-radioactive cations attached to negatively charged clay minerals and immobile dissolved organic matter. Ion exchange therefore results in a smaller amount of radionuclides being available for transport in clay pore water.

8.1.3.1 Diffusive transport as modelled in OPERA

A diffusion value of 3×10^{-10} m2/s was used for the EBS for all elements in calculating the influx from the EBS into the clay host rock (Meeussen and Grupa, 2017; Schröder et al., 2017b).

Element specific diffusion values were used to model transport in the clay host rock. Many radionuclides, such as radioactive isotopes of caesium and uranium, are cationic dissolved species that exchange with non-radioactive cations originally in the minerals and immobile dissolved organic matter. This ion exchange is incorporated in the model by reducing the diffusion value using retardation factors (K_d values) calculated by dividing the concentration of elements in the solid and immobile phases of the clay rock by the concentration of dissolved elements in pore water (Meeussen and Grupa, 2017; Meeussen et al., 2017; Schröder et al., 2017b). The assumed solubility of an element is therefore critical for the assumed diffusion value. The assumed solubility for uranium in clay was taken to be 10^{-4} mol/l as a default value. This solubility is higher than the uranium solubility of 10^{-5} mol/l assumed in the EBS. The uranium solubility of the EBS was the dominating factor (Schröder et al., 2017d) controlling the release of uranium from the multibarrier system into the biosphere.

8.1.3.2 Diffusive transport as updated in COPERA

The chemical evolution of the different barriers was assessed in EURAD-1 (ACED) at waste package scale. The transport properties of water in COVRA's waste package concrete and backfill concrete were investigated in order to understand the impact of differences in porosity and size of pores. The diffusion values derived for water in these cementitious materials when saturated at 20°C were 0.8×10^{-11} m²/s for waste package concrete and 1.6×10^{-11} m²/s for the tunnel backfill (Blanc et al., 2024). These diffusion values are at least an order of magnitude smaller than any diffusion value assumed for radionuclides in the EBS in OPERA. The water diffusion value for COVRA's waste package concrete is expected to be similar to the concrete buffer in the supercontainer since the content of aggregates is the same. The porosity and size of pores are smaller for the buffer than for the backfill, leading to smaller diffusion values for the buffer than the backfill. However, the HLW supercontainer buffer is conservatively assumed to have the same properties as the backfill. The higher diffusion value of 3×10^{-10} m²/s from OPERA was used for the waste form in order to observe the impact of treating the HLW EBS as a multibarrier system in which radionuclides leaving the supercontainer buffer need to pass through the backfill, with a lower diffusion value, before they enter the clay host rock.

The model we now use in COPERA is of a gradual release of radionuclides from the EBS over a period of time whose duration depends on the durability of the concrete and the evolution of its properties with time. These properties limit the transport of water to the waste form, imply lower diffusion values for dissolved species and permit potential ion exchange with cementitious minerals. This ion exchange potential depends on the evolution of the pH of the concrete pore waters. However, taking the impact of pH into account makes the safety assessment more complicated and this has not yet been accomplished.

We limit our quantitative update of the OPERA safety assessment to examining the impacts of updated radionuclide diffusion values. Diffusion values in the concrete barriers are sensitive to the distribution in size of pores, and this distribution may change over thousands of years after GDF closure, as explained in section 6.1.3. As explained in section 6.1.2, also the strength of concrete is sensitive to the distribution of size and pores, but the strength is not expected to decrease with appropriate choices for making the concrete and good engineering, which provides an argument that the diffusion values in concrete may not increase.

The OPERA diffusion values for the clay host rock are used, except in one case in which the radionuclide (niobium-94) was categorized as an anion, whereas the evidence obtained in Chapter 5 indicates that this radionuclide would behave as a cation. Diffusion values for cations (e.g., caesium, potassium) that would lead to their almost total decay in a clay host rock with a thickness of 100 m (see also Chapter 5) are highlighted.

8.1.4 Uncertainties in the modelling

The safety case needs to consider different kinds of uncertainties, including uncertainties in the understanding of the processes determining system behaviour, in the models, in parameter values and in the scenarios (e.g., (IAEA, 2012). Section 8.4.1 addresses the propagation of some uncertainties through the assessment of the overall performance of the multibarrier system:

- System uncertainty: arises from incomplete understanding or characterisation of the multibarrier system. The uncertainties related to the performance of the barriers in the clay host rock and barriers in the EBS are discussed in Chapter 5 and 6, respectively;
- Model uncertainty: relates to whether conceptual models adequately describe the behaviour of the multibarrier system and its components. The conservative subsystem conceptual models that we currently use clearly do not represent reality. These conservative models include those for degradation of the waste form and for transport of radionuclides.
  The conceptual model assumed in OPERA,- that unlimited water is available for alteration of the waste form - has a major impact on calculated doses that could result from releases from the multibarrier system. In the multibarrier system, the access to water at the waste form - EBS interface and the transport of water through the EBS to this interface are much lower than the experimental values applied in tests where samples are immersed in a solution. Radionuclides are present only in very small concentrations in all waste forms, apart from depleted uranium and spent fuel. For radionuclides present in trace amounts, their complete dissolution may be feasible because the resulting concentrations in pore water are below the solubility limit for the relevant elements. For uranium in depleted uranium and spent research reactor fuel, the solubility limit of uranium will prevent complete dissolution, unless the turnover rate of water in contact with waste massively increases. The time it takes to transport water to the waste package thus needs to be calculated. Evaluations of the volumes and turnover rates of water in contact with waste materials may explain the differences between experimental leaching results obtained in the laboratory and the observed corrosion behaviour found in studies on archaeological and natural analogues. Reducing this gap is especially important for the calculation of radiation doses in a normal evolution scenario;
- Parameter uncertainty: relates to the accuracy of all parameter values used in the safety assessment. This uncertainty can be related to the measurement technology and sampling methodology. It is often addressed by considering a wide range over which the parameter value may vary. Using different range widths allows different levels of uncertainty to be represented in normal, alternative or what-if scenarios.

  Natural variability and heterogeneity mean that single parameter values may be inappropriate, and some models need to incorporate a wide range of values for certain parameters. Natural variability is not the same as uncertainty and can, to some extent, be constrained by thorough characterisation of the barriers in the multibarrier system. The Dutch Paleogene clays have been little investigated experimentally in the laboratory or in-situ, but the geochemical characterisation work in OPERA (Koenen and Griffioen, 2014) and the modelling of its results (see Chapter 5) allows some constraints on the range of possible diffusion values that determine how radionuclides migrate. The preferred approach is to use realistic data and assumptions where possible. Pessimistic data can be selected from the realistic range in data values in order to perform conservative assessments.

  Numerical uncertainties and variability are commonly dealt with by performing sensitivity analysis, in which relevant parameters are varied throughout their potential ranges.

This can be done through deterministic modelling of multiple cases or by probabilistic modelling, using probability distributions of parameter values. In the latter case, the correlations between parameters must also be considered in order to avoid physically unrealistic combinations. Unrealistic combinations in OPERA led to some unrealistic element-specific diffusion values in clay, as explained in Chapter 5. Given the lack of characterisation of clay host rocks in the Netherlands, it is appropriate at present to employ deterministic modelling. After sufficient characterisation of all the barriers in the system and understanding their behaviour in a normal evolution scenario, it becomes feasible to determine probability density functions of parameter values for a safety assessment.

- Scenario uncertainty depends on how well the features, events and processes are understood, and whether these uncertainties are representative for normal, alternative, what-if and human intrusion scenarios. Scenarios are described in Chapter 7.

8.2 Treatment of the biosphere

The biosphere acts as the receptor for any radioactivity that moves upwards from the geosphere and the safety assessment needs to model biosphere processes that determine how people might be exposed to radionuclides that have left the multibarrier system. However, in the timeframe from 10^4 to 10^6 years after closure of the GDF (the period in which radioactivity might reach the biosphere), long-term natural changes in climate will occur and the range of possible biospheres and human behaviour is too uncertain for reliable modelling. Consequently, for this time period, hypothetical critical groups of people living in reference biospheres are usually proposed as a basis for modelling potential exposures to radioactivity.

The characteristics of these groups and biospheres are chosen to represent circumstances under which the highest doses could arise, given our knowledge of present-day habits and biospheres. Standard practice is to estimate radiation doses to people conservatively, by defining the most highly exposed individual, usually taken to be a member of a subsistence community taking water from a well for drinking and for use by cattle and for crop irrigation, and food from local sources, including rivers or lakes. Information on present day conditions represents the largest and most reliable database for environmental transfer of radionuclides (IAEA, 1999) and it is standard practice to use one or more reference biospheres, based on temperate climate conditions (IAEA, 2003a).
Three biosphere receptors for radionuclides released from the GDF were considered in the safety assessment in OPERA: a well, a river and soil, as shown in green in Figure 8-2. The ingestion of radionuclides from well water takes place through drinking this water or by eating irrigated crops, or agriculture animals that have ingested the water or the crops, as well as their products (milk, eggs). Irrigation of the soil using well water can also lead to external radiation and can lead to inhalation of gaseous radionuclide decay products. River water can also be used to irrigate soil and eating freshwater fish leads to another exposure pathway.

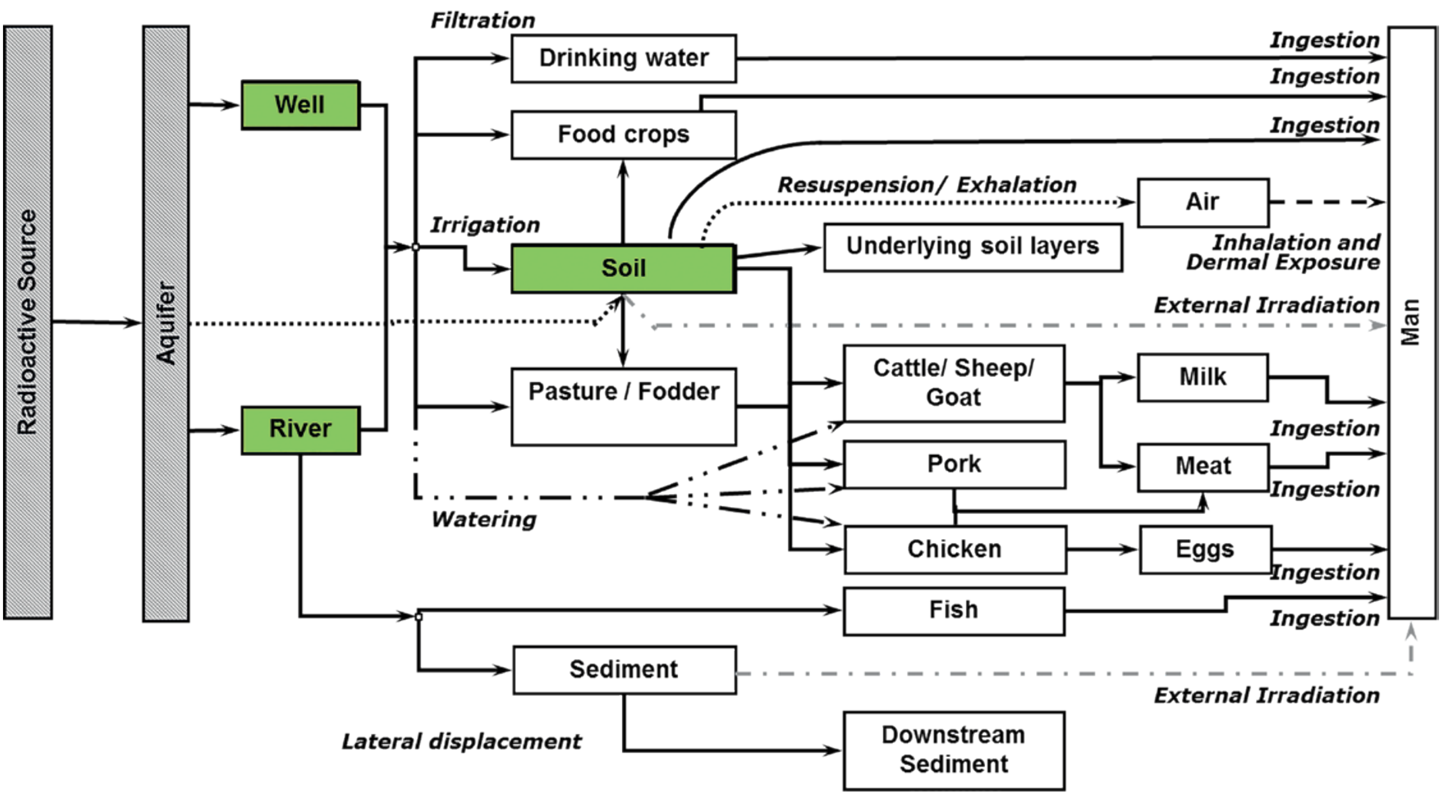


Figure 8-2: Schematic illustration of the compartments considered in the biosphere model. Well, river and soil are the biosphere receptors (green). The grey shaded compartments (radioactive source, aquifer) provide the flux of the radionuclides into the biosphere and are not part of the biosphere model (Grupa et al., 2017); Figure 8-5 in the OPERA Safety case.

For the safety assessment calculations shown in the OPERA Safety case and also in this COPERA safety case, only the (local) irrigation water well has been used, as this is the scenario in which the highest doses would arise, given our present knowledge of habits and biospheres. In the model, a shallow well was assumed, with an abstraction capacity of 14,000 m^3 per year (Schröder et al., 2017b). No dilution is assumed for this well case. The consumption rates of drinking water, crops and food by people, and their exposure time to contaminated soil, are assumed to be climate dependent. Consumption rates and exposure times in Belgium were chosen for the temperate (moderate) climate state, those in Spain for the Mediterranean (warm) climate state and those in Sweden for the Boreal (cold) climate state. Dose conversion coefficients (Sv/year per Bq/m^3) were obtained for this well scenario and these three climates states (Grupa et al., 2017)

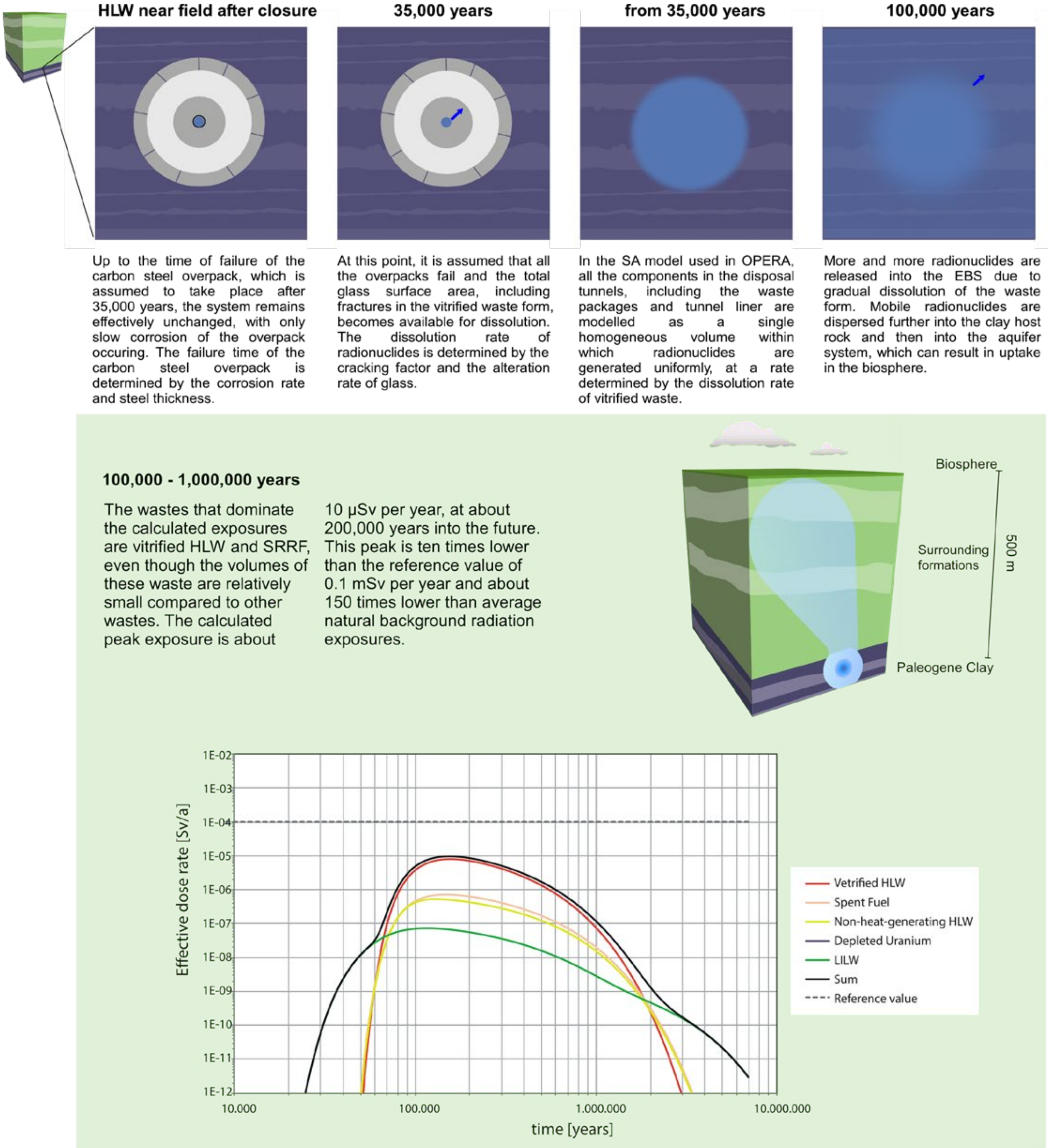


Figure 8-3: Contributions of each waste type in OPERA to the effective dose rate, aggregated for all radionuclides. Release of radionuclides from the EBS to the clay host rock begin at 35,000 years for HLW and 1'500 years for depleted uranium; adapted Figure 8-6 from the OPERA Safety case.

8.3 Yardsticks for judging post-closure performance

The aim of geological disposal is to ensure the safety of people and the environment at all future times. Calculated radiation doses indicate the level of protection afforded by a system with multiple barriers; they can be used for judging safety levels and to guide the optimisation of the multibarrier system. Concentrations of radionuclides throughout the multibarrier system, and particularly in the biosphere, can also be used to assess the functioning of the multibarrier system. For interpretation of the calculated results, the dose rates and concentrations are compared to present-day measures of safety and to naturally occurring radiation doses or radionuclide concentrations: these comprise our reference values or yardsticks.

8.3.1 Calculated radiation doses

As explained in Chapter 3, the Dutch radiation protection decree has not yet set specific limits in the Netherlands for potential releases from a geological disposal system. A dose limit of 0.1 mSv per year was used in the OPERA safety case and is also used in this safety case.

8.3.2 Concentration of natural radionuclides throughout the multibarrier system

Uranium, thorium and radioactive potassium are present in concrete and the clay host formation, as explained in Chapters 4, 5 and 6. Chapter 3 showed that the average radiological exposure of 1.7 mSv per year from natural radionuclides currently received by the Dutch public is mainly from these three radionuclides, which have a primordial origin. In the multibarrier system, the volume of the waste is much smaller than the volume of the surrounding clay host rock formation. More natural radionuclides and chemical analogues of radionuclides in the waste can be present in a surrounding volume of the clay host rock than in the waste in the GDF. The fluxes of these natural radionuclides, as well as chemotoxic elements from the clay, into the aquifer system will be determined in a future safety case, when sufficient information has been obtained from the geochemical characterisation of the Paleogene clays and surrounding sands. This will allow comparison of the calculated health effects of natural radionuclides compared to those resulting from radionuclides that might be released from the waste.

8.4 Calculated safety assessment results

The main results of calculations for the normal evolution scenario were presented in the OPERA Safety case. This section reproduces the peak exposures calculated in OPERA for radionuclides from vitrified HLW and compacted hulls and ends and then discusses how the knowledge developed since OPERA can have an impact on the peak exposures from these types of waste. For heat-generating HLW, the carbon steel overpack in the supercontainer has a thickness that provides sufficient mechanical support and prevents failure due to corrosion for a long period of time. For HLW, the period of 35,000 years was chosen as a base case (default value, DV) after which release of radionuclides occurs (Schröder et al., 2017b). The waste package for depleted uranium has a smaller thickness of carbon steel than the HLW overpack and the base case (DV) was deduced to be 1,500 years (Schröder et al., 2017b).

8.4.1 Calculated radiation doses in the OPERA base case

Figure 8-3 shows the calculated dose rates over ten million years in the OPERA safety assessment model. Vitrified HLW contributes most to the calculated dose. The calculated doses for depleted uranium do not appear in this figure, as they are too small over this entire period.

Figure 8-4 shows the six radionuclides contributing most to the effective dose rate. The predominant origins of these six radionuclides are (Rosca-Bocancea et al., 2017):

- vitrified HLW (CSD-v) for Se-94;
- spent research reactor fuel and non-heat generating HLW for I-129;
- non-heat generating HLW (CSD-c) for Nb-94;
- LILW for Re-186m, Cl-36, K-40.

With the exception of Nb-94, only radionuclides with half-lives larger than 100,000 years leave the multibarrier system. In OPERA, Selenium (Se), niobium (Nb), chlorine (Cl) and iodine (I) are assumed

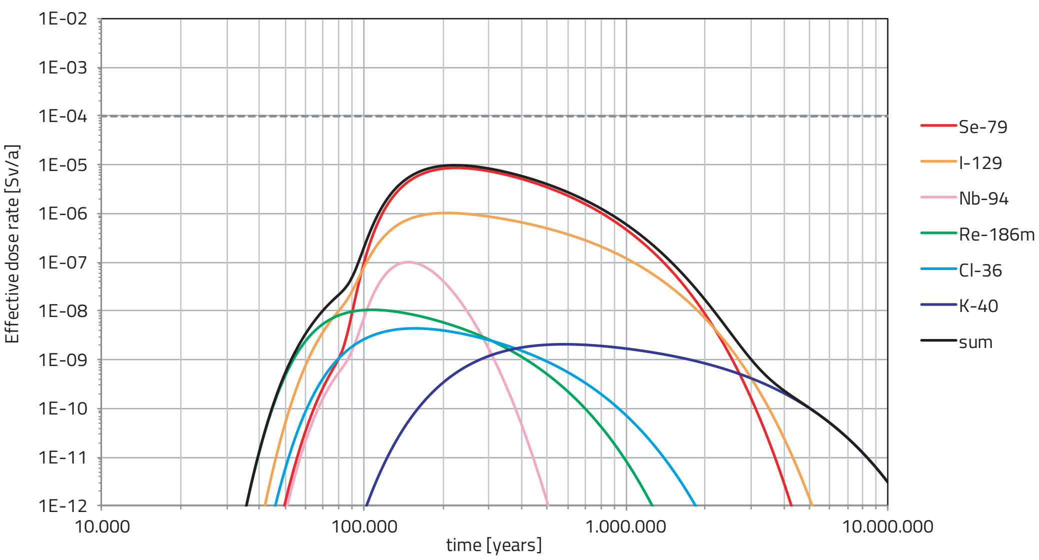


Figure 8-4: OPERA calculations of contributions of six radionuclides contributing most to the effective dose rate. Release of radionuclides from the EBS to the clay host rock set at 35,000 years for HLW and 1,500 years for depleted uranium (Rosca-Bocancea et al., 2017); Figure 8-7 in the OPERA Safety case.

to be dissolved in pore waters as anions. These anions have lower diffusion values than neutral species such as water but there is no retardation by ion exchange with clay minerals or other immobile phases in the clay host rock. Iodine and chlorine are halogens, which are always present as anions. Although Nb was assumed to be an anion in OPERA, the increasing concentration of niobium with increasing clay content in Boom clay samples, as presented in Chapter 5, suggests that Nb behaves similarly to other transition metals that are present as cations in clay host rock and thus Nb was assumed to act as a cation in COPERA. Elements that are dissolved as cations can have a smaller diffusion value than anions if they are retarded by ion exchange. Due to lack of data, rhenium (Re) was conservatively assumed not to be retarded. The retardation factor assumed for potassium (K) is so small that its assumed diffusion value in clay host rock remains larger than the diffusion value for water.

8.4.1.1 Comparison of radioactive potassium in clay host rock and LILW

Radioactive Potassium-40 (K-40) is only present in compacted LILW (see Appendix 6) and its average concentration in the waste form is 84 Bq/kg, assuming an average weight of 500 kg for the 200 litre drums. This is smaller than the concentration of K-40 in concrete (227 Bq/kg; (Smetsers and Bekhuis, 2021) and is also smaller than the concentration of K-40 naturally present in the clays. In fact, the K-40 emitted gamma-ray together with uranium and thorium, is used to determine the thickness and depth of clays by the logging in the borehole (e.g. (Vardon et al., 2022)).
The presence of K-40 in the clay host rock was not taken into account in OPERA. In Chapter 3 it was pointed out that K-40 is mainly responsible for the external gamma radiation of current populations from the soil (0.04 mSv per year) and provides the largest contribution to the radiation dose from foods (0.43 mSv per year). These effective dose contributions are 100,000 to a million times larger than the calculated contribution of K-40 to the effective dose rate from radionuclides in the disposed waste, so that these can be regarded as insignificant.

8.4.1.2 Impact of new COPERA data on Nb-94 doses from compacted hulls and ends

Radioactive Niobium-94 is a neutron activation product and is predominantly found in the inventory of reprocessing wastes in the CSD-c canisters. Non-radioactive Niobium-93 is a component of Zircaloy reactor fuel cladding, included in order to increase corrosion resistance and, in the reactor, this absorbs a neutron to become Nb-94. In OPERA, instant release of radionuclides from the Zircaloy is assumed to occur after 35,000 years. However, as shown in Chapter 6, the maximum long-term anaerobic corrosion rates for irradiated Zircaloy cladding exposed to cementitious solutions are less than 0.002 μm per year. Moreover, the 390 kg of Zircaloy in each CSD-c would requires 80 kg of water for its complete corrosion (see section 6.2.5). But the low permeability of the clay and concrete imply that the maximum possible inflow rate of water into the disposal tunnels, as calculated in Figure 7-2, is 0.1 kg per year per metre of disposal tunnel. The Zircaloy corrosion rate and the water supply rate both imply that the releases of Nb-94 would be spread over a long period of time, highlighting the high degree of conservatism in the instantaneous release model. In a future safety assessment, data on the cladding surface area that can potentially be exposed to water and the associated water consumption will be analysed, in order to determine a more representative alteration rate in the envisaged engineered barrier system.

A more representative alteration rate for the EBS would also allow more CSD-c containers to be loaded into each HLW package (see Chapter 4, seven CSD-c are proposed per package).

The calculated doses for Nb-94 in COPERA are strongly affected by the new modelling of diffusion and also by the assumption that Nb is in cationic form. Figure 8-5 reproduces the calculated Nb-94 OPERA results in Figure 8-4, together with doses calculated using the recently obtained diffusional properties of the tunnel backfill (Blanc et al., 2024), with a thickness of 2.25 m. Recent studies indicate that the backfill has about a 20 times smaller diffusion value than the value for diffusion used in OPERA. It shows that the peak dose rate would be reduced by a factor of ten. The peak dose is too low to appear in the figure if niobium is assumed to behave as a cation.

Another line of reasoning on why radioactive niobium is not expected to contribute significantly to the effective dose rate is the natural presence of niobium in clays. As explained in Chapter 5, the concentration of natural niobium in Boom clay is on average between 10 and 15 mg/kg, which is similar to the concentration of radioactive niobium (about 9 mg/kg) in CSD-c canisters, after 130 years of storage. The half-live of 20,000 years implies that two-thirds of the radioactive niobium has decayed after 35,000 years, when release is assumed to begin in the safety assessment base case. In practice, diffusive transport of niobium would tend to be from the clay host rock towards the EBS, if the concentration of niobium in the clay pore water is higher than in the pore water of the EBS.

8.4.1.3 Impact of new COPERA data on Se-97 doses from Vitrified HLW

Selenium-97 is a fission product and is predominantly found in the vitrified HLW. As explained in Chapter 6, the glass dissolution rate used in OPERA was obtained from experiments with a low solid to liquid ratio (low S/L in Figure 8-6). Experiments using a more representative solid to liquid ratio (high S/L in Figure 8-6) have a 30 times smaller dissolution rate. Figure 8-6 shows that the peak dose would be more than 50% lower, compared to the calculation performed in OPERA.

COPERA also makes calculations with instant releases, but with a thickness of 2.25 m for the tunnel backfill. These calculations are purely hypothetical and not representative for a normal evolution scenario since the complete alteration of 390 kg of vitrified waste in each CSD-v canister to permit instant mobilisation would require access to more than 110 kg of water (see section 6.2.4). Allowing for the 2.25 metre thick tunnel backfill in COPERA also significantly reduces the effective dose rates as calculated in OPERA.

Only dissolved radionuclides can leave the multibarrier system and two properties of the clay limit the concentration of dissolved species in clay pore water: reducing conditions and ion exchange. In the safety assessment, selenium was not assumed to be solubility limited whereas the formation of pyrite can significantly reduce the soluble content of selenium (Hoving et al., 2019). Selenium is not expected to be retarded by ion exhchange in the clay host rock, as it behaves as an anion, but the solubility of selenium is quite low in clay host rock due to the prevailing reducing conditions underground. The assumed solubility of selenium in the safety assessment may be more than hundred times larger than the actual solubility in clay host rock.

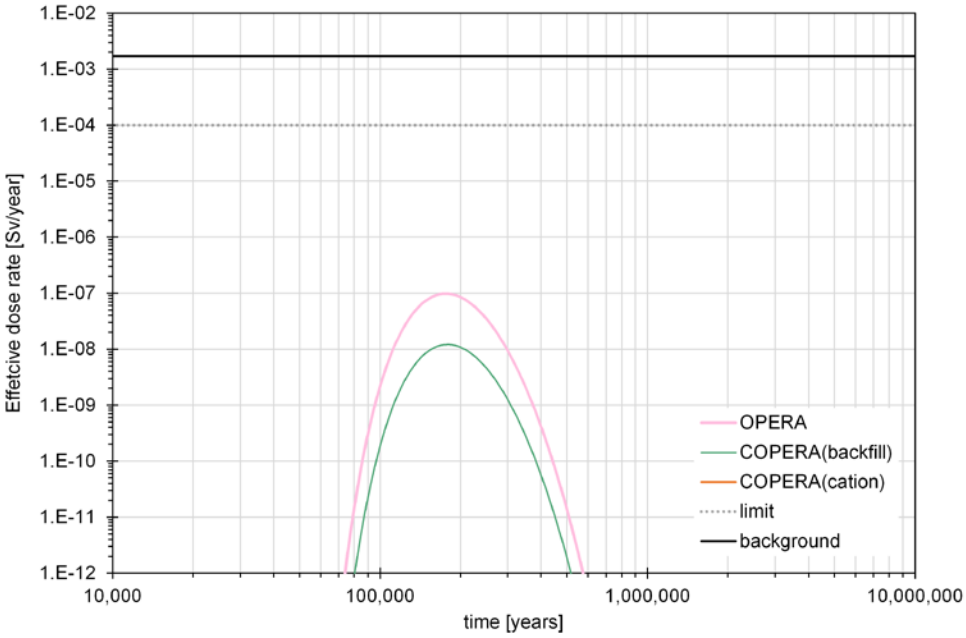


Figure 8-5: Contribution of the effective dose rate of radioactive niobium assuming instant release with different geometry for the engineered barrier system (a single CSD-c per HLW disposal package) and revised assumptions for the behaviour of dissolved niobium. Instant release after 35,000 years is a conservative simplification that implies unrealistically high water consumption rates.

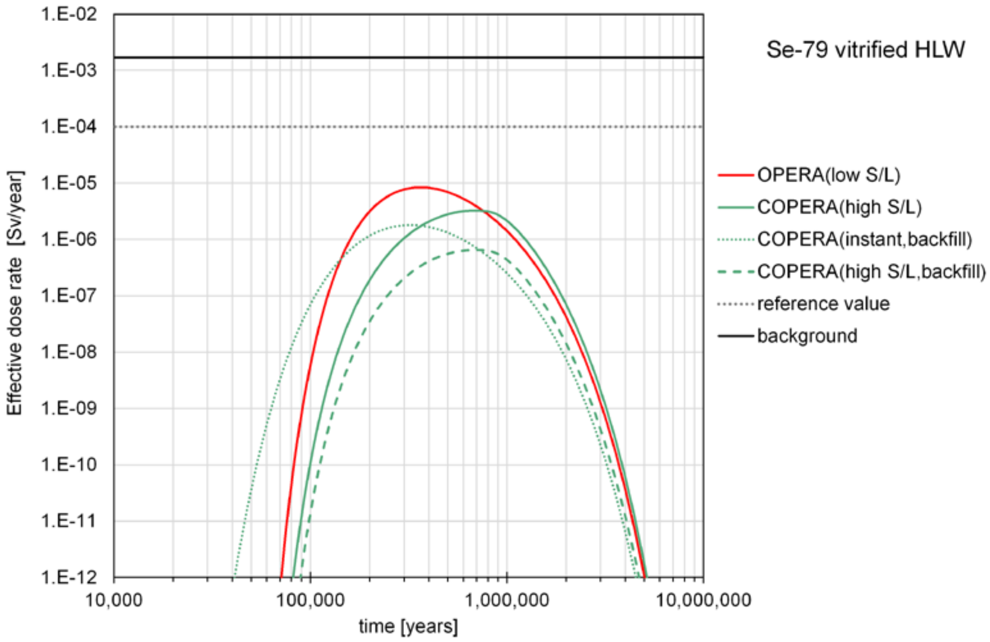


Figure 8-6: Contribution of the effective dose rate of radioactive selenium assuming release after 35,000 years (OPERA (low S/L and COPERA (high S/L)) and after 1,200 years (COPERA (instant, backfill) and COPERA (high S/L, backfill). Instant release after 35,000 years is a conservative simplification that implies unrealistically high water consumption rates.

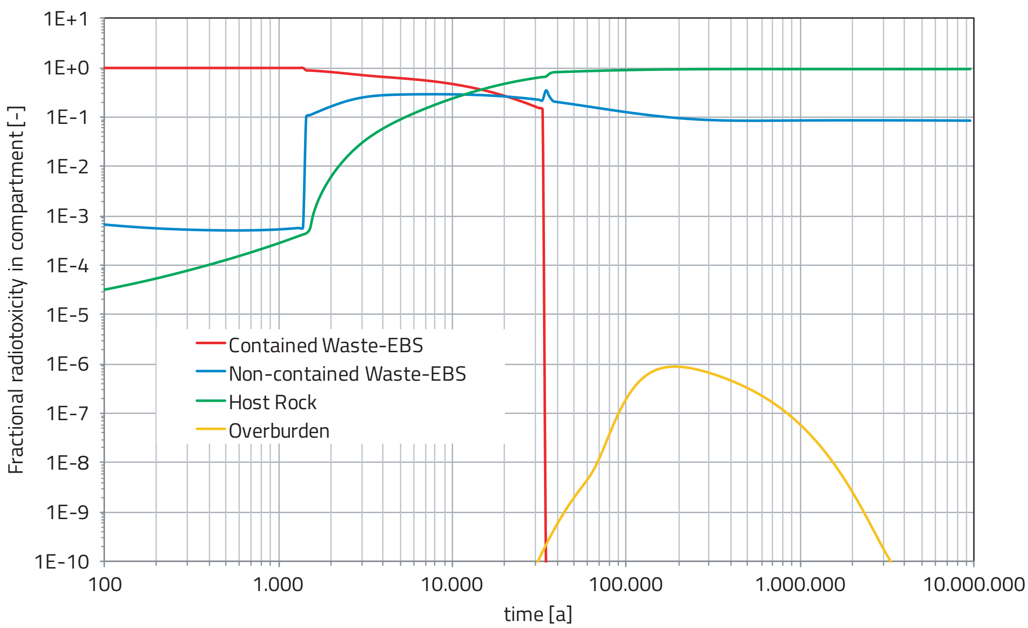


Figure 8-7: Fractional radio-toxicity of in the subsystem (EBS), clay host rock and overburden (Rosca-Bocancea et al., 2017). Figure 8-9 from the OPERA Safety case.

8.4.2 Performance of the multibarrier system

The disposal packages for HLW hold the largest fraction of the radioactive inventory and, in the safety assessment model, completely contain this radioactivity until their allocated time of failure. This time was set at 35,000 years in the base case in OPERA. All the HLW packages are pessimistically assumed to fail together and the radioactivity in non-heat generating HLW and spent research reactor fuel becomes instantaneously available for diffusion into the clay host rock. From this time onwards, as shown by the green line in Figure 8-7, which is taken from OPERA, the bulk of the total radiotoxicity in the system resides in the clay host rock.

Figure 8-5 and Figure 8-6 show the contribution of the effective dose rate of Niobium-94 and Selenium-79. In order to show the impact of updated data, the instant release of radionuclides after 35,000 years and the single CSD-c per HLW package from OPERA have been kept the same for the calculations in Figure 8-8. The EBS in OPERA had a height of 2.5 m (see Figure 8-1). The total height in EBS in COPERA was also 2.5 m and divided into the waste form (0.25 m) and backfill (2.25 m) (see Figure 8-1). Figure 8-8 also shows the additional effect of assuming niobium as an anion in the clay host rock data (green line: COPERA, anion) and cation (orange line: COPERA cation). Figure 8-8 shows that most of the radioactive niobium content decays within the EBS and is not released into the clay host rock. This recent calculational result in COPERA is similar to that in the Swiss safety case for Opalinus clay (NAGRA, 2002) (see section 7.1.4).

In order to show the impact of updated data, the instant release of radionuclides after 35,000 years and the single CSD-v per HLW package from OPERA have been kept the same for the calculations in Figure 8-9. The same dimensions are used (2.5 m in OPERA and for COPERA 0.25 m waste form and 2.25 m backfill) as the calculational result in Figure 8-8. One of the differences with the previous calculation is that the (vitrified) waste form gradually releases radionuclides in the safety assessment while instant release of radionuclides is assumed for CSD-c. As explained in Chapter 6, the glass dissolution rate used in OPERA for the base case is highly conservative. The rate at which selenium from the waste actually enters solution reduces when a more realistic 'dissolution' rate is used, as shown in Figure 8-9, which compares cases with two different dissolution rates.

8.4.3 Diffusion rates in the host rock

For the host rock, the calculated results described so far have used the median values for the element-specific diffusion values derived for clay host rock in OPERA. The physical reasonableness for the diffusion values for Re-186m and K-40 that behave as cations in clay host rock were addressed in section 8.4.1. The derived maximum diffusion values of these radionuclides is almost two orders of magnitude larger than iodine and selenium. Consequently, their peak dose rates calculated in OPERA are also more than an order of magnitude larger in Figure 8-10 than those shown in Figure 8-4. The appearance of Caesium-135 is caused by assuming virtually no ion exchange. As explained in Chapter 5, the trace amounts of chemical analogues of Caesium-135 measured in Paleogene clay would not be present with this maximum diffusion value.

8.5 Concluding perspective

With respect to the LILW inventory, the radionuclides in the wastes are assumed to be instantaneously available at GDF closure for release into the clay host rock, i.e., no engineered containment is assumed. Despite this assumption, a GDF located at the centre of a 100 metre thick clay formation at 500 m depth provides sufficient containment and radionuclides dissolved as cations are not expected to leave the clay host rock. Only radionuclides that are dissolved as anions and have half-lives longer than 100,000 years are calculated to reach the biosphere.

With respect to HLW, the principal contributor to the total radioactive inventory of the GDF, one radionuclide dominates the calculated radiological exposure in the base case, in which radionuclides from HLW are released from the supercontainers after 35,000 years. This radionuclide is Se-79 (predominantly from vitrified HLW) with a contribution of 8 μSv per year after 200,000 years. Recent results obtained by SCK CEN for vitrified HLW and calculating the associated water consumption for altering the vitrified waste form into clay minerals and zeolites show that the glass dissolution rate used in OPERA is unlikely to apply in the multibarrier system.
A more realistic glass alteration rate is more than 30 times smaller, leading to a contribution of less than 4 μSv per year after more than 600,000 years: i.e., a 50% reduction in peak exposure (see Figure 8-6). Although the solubility of selenium is low in clay host rock, due to the prevailing reducing conditions in the underground, the transport of selenium in the clay is assumed not to be solubility-limited. This assumption may not be realistic and the amount of dissolved selenium that leaves the clay host rock is expected to be smaller. Another feature indicating that the calculated dose rate can be smaller than currently calculated is that the concrete barriers have conservatively been assumed to have a diffusion value for selenium (an anion) equal to the diffusion value of water in the backfill, which is the most porous concrete barrier. Ion exchange in cementitious materials is highly likely to occur, as research on other anions and anionic dissolved complexes have shown (Pointeau et al., 2008). Inclusion of ion exchange would lead to lower diffusion values for selenium in concrete, resulting in lower concentrations of dissolved selenium at the interface with the clay host rock in a normal evolution scenario. Even without taking this possible reduction in diffusion value into consideration, the contribution of selenium reduces to below 1 μSv per year, for an assumed thickness of 2.25 m for cementitious materials with diffusion value of water with the most porous concrete barrier (see Figure 8-6).

With the exception of the vitrified waste form and depleted uranium, many radionuclides are present in smaller concentrations in the wastes than the non-radioactive isotopes of the same elements in the clay host rock. It is therefore useful to investigate how natural radionuclides and chemical analogues of radionuclides in the waste behave in Paleogene clay formations, since these studies also contribute to the confidence in the chosen speciation of a radionuclide.

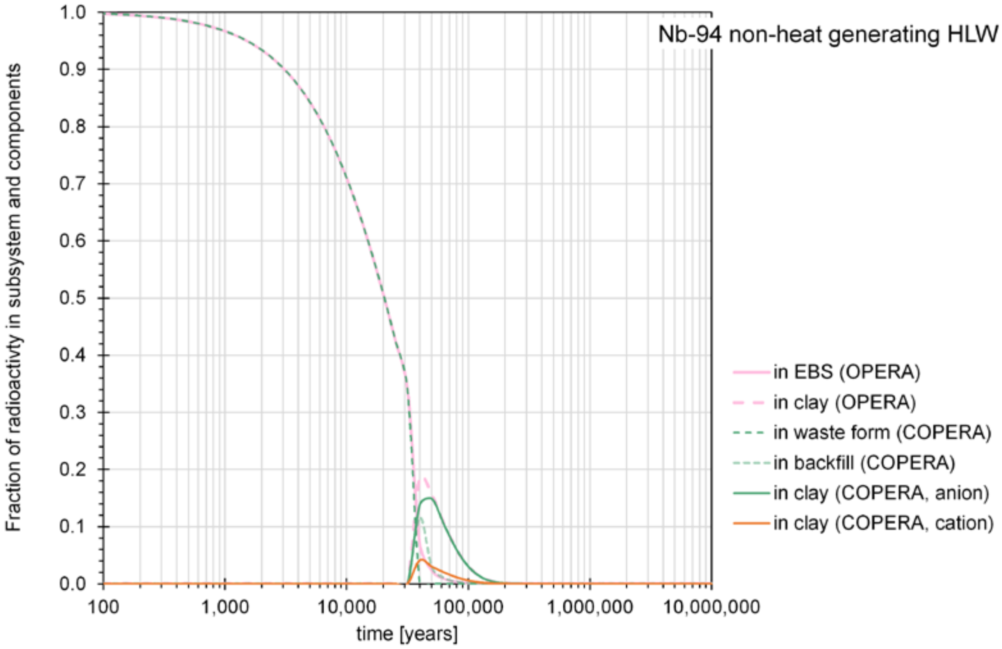


Figure 8-8: Containment of the radioactivity of Niobium-94 in the components of the MBS: in a single EBS volume(OPERA), in the waste form (COPERA) and all other EBS backfill properties (COPERA) and clay host rock of the calculated result in Figure 8-5.

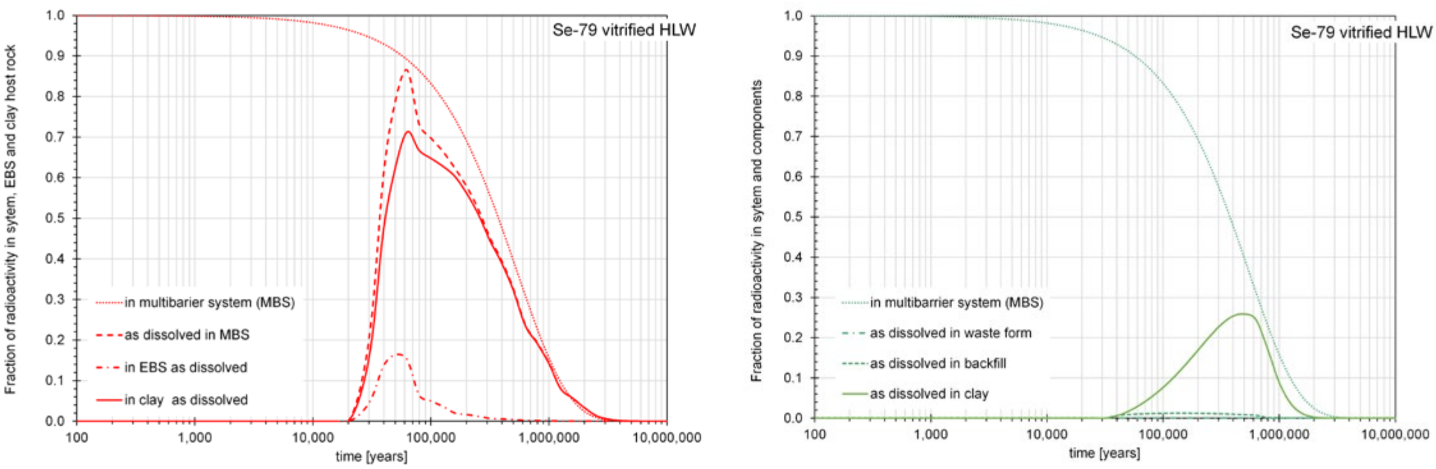


Figure 8-9: Containment of the radioactivity of Selenium-79 in the components of the MBS for two calculated results in Figure 8-6: OPERA(red, left) used the glass alteration rate obtained from an experiment with a low S/L ratio and COPERA (green, right) uses a glass alteration rate with a high S/L ratio. The calculated fraction as dissolved in the waste form (COPERA) is due to the small glass alteration rate and high diffusion values in the waste form, so small that it is not visible in this graph with a linear scale.

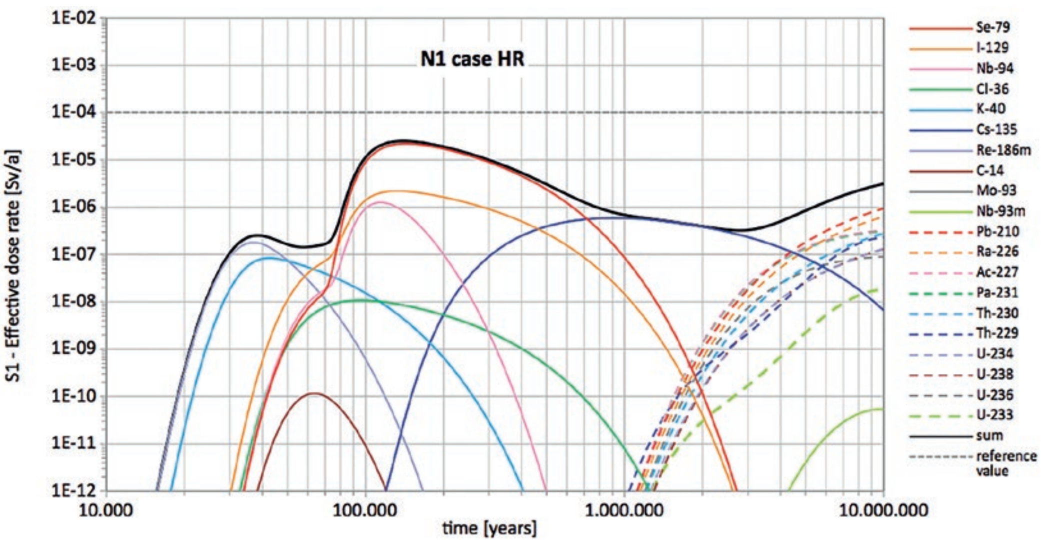


Figure 8-10: Calculated effective dose rate from all the wastes in the disposal system, using the maximum diffusion values obtained in OPERA, release of radionuclides from the EBS to the clay host rock after 35'000 years for HLW and the glass alteration rate obtained from the experiment with a low S/L ratio (Schröder and Rosca-Bocancea, 2017). Figure 8-11 in the OPERA Safety case.

Box 8-1: Uranium

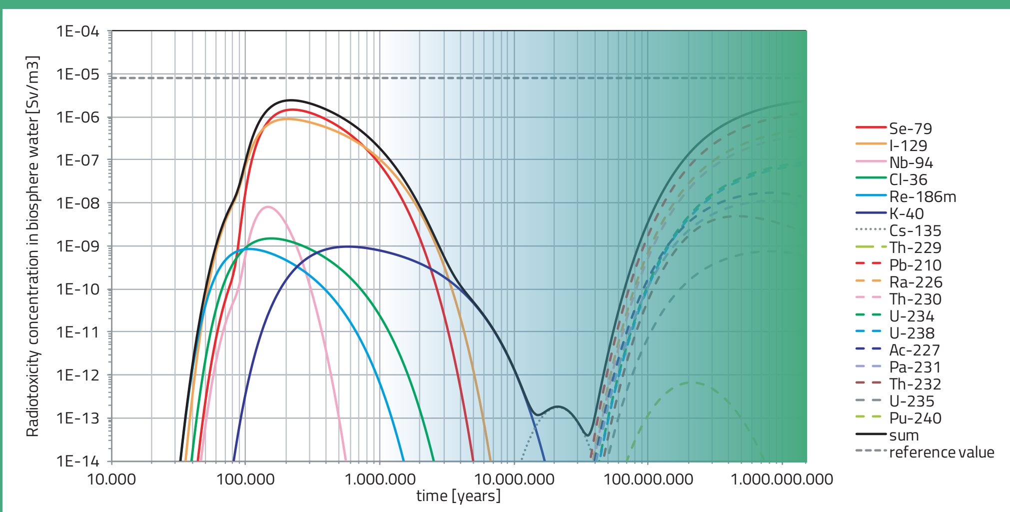
The GDF contains considerable amounts of uranium: about 110,000 tons in total, 99.6% of which consists of depleted uranium, mainly present as U_3O_8 from the uranium enrichment facility, Urenco. Despite the large inventory, within a calculation period of 10 million years, doses from uranium and its daughter radionuclides[17] are not visible in the calculated radiation exposures (Figure 8-6). Uranium is generally assumed to be rather immobile and only with the most conservative parameter values for migration in the Boom clay does the breakthrough of uranium and its daughters become visible but after a period of 10 million years (Figure 8-10).

As discussed in section 8.3.1, radiological dose calculations at long times into the future have limited value and a more appropriate indicator is the radiotoxicity concentration in biosphere water. The figure below[18] depicts this safety indicator for the base case of the NES (the same conditions as for the exposures shown in Figure 8-6), but with the calculations propagated for a much longer period. It can be seen that the uranium series do not begin to contribute significantly to the radiotoxicity released from the EBS until many tens of millions of years into the future. At these times, the calculation basis becomes highly stylised and is largely illustrative, because considerable changes would be expected to occur in both the biosphere and the geosphere. This is indicated by the darkened shading with increasing time. Although the graph extends to more than a billion years, it is recognised that most of Earth's crustal rock are recycled on such timescales, so neither the EBS nor the Paleogene Clay itself might be expected to survive for this time.

The safety assessment assigns a solubility limit to uranium to provide a more realistic evaluation of its behaviour. However, there are uncertainties with respect to the solubility limits of U_3O_8. In a Paleogene clay, uranium is assumed to be present in its more soluble U(VI) form. In the expected redox range in these clays, mixed valence uranium oxides (U_4O_8 and U_3O_8) might control the solubility of uranium, but it is argued that applying the thermodynamic solubility constants for these minerals could lead to underestimation of the real solubility[19]. Consequently, these minerals are not considered in the derivation of the solubility limits and K_d values used in the OPERA safety assessment. Schroder et al. observe (NRG745 p. 25) that the uncertainties in values used for uranium solubility have the largest effect on its calculated radiotoxicity concentration in the host rock and biosphere.

Uranium forms strong complexes with organic matter, which generally leads to high retardation, as is evident from the long-delayed arrival of uranium shown in the figure below. For conditions expected in the Netherlands, the DOC-bound fraction of uranium dominates the soluble amounts in most cases. However, with significant amounts of bicarbonate and high concentrations of uranium in solution, uranium solution chemistry might be dominated by the stable uranyl carbonate ion, $UO_2(CO_3)_3^{4-}$, and under certain specific conditions (high bicarbonate, DOC and uranium contents of Boom clay pore waters, combined with high pH of >13), uranium might migrate with little retardation through a Paleogene Clay. Under such conditions, high concentrations of soluble uranyl carbonate are calculated Schroder (2017). However, this specific combination of conditions over extensive volumes of the potential diffusion pathway in the Boom clay appears unfeasible. Other experiments also imply that reduction of uranyl carbonate might occur in Boom clay, which would strongly decrease the mobility of uranium.

The complex redox behaviour of uranium and its carbonate species results in uncertainties in the solubility and sorption behaviour that can only partially be resolved without further experimental research. One consideration to improve the



Radiotoxicity concentration in biosphere water in the OPERA central assessment case (N1DV), calculated over 1.5 billion years. PA-model 9.3-multiwaste. See text for note on shading.

current understanding of uranium mobility is to integrate specific sorption of uranyl carbonate - based on recent experimental research - in the model used to derive the K_d values for the safety assessment. More detailed study of the speciation and behaviour of naturally occurring uranium in Paleogene clays and the overlying and underlying Paleogene sands would also provide direct evidence as to its fate.

The overall conclusion of the OPERA safety assessment is that, for the NES, uranium will remain within the EBS and the clay host rock for as long as the formation is there. Furthermore, any migration of uranium and its daughters, even after hundreds of million years, is not expected to change the background levels significantly. This is what is observed in the Cigar Lake ore deposit in Saskatchewan, Canada. The deposit is contained in a small clay-rich halo at 450 m depth surrounded by water saturated sandstone. No radiometric signature from the one-billion-year-old deposit has been detected in the biosphere.

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17. Schroder et al. (NRG745: see footnote below) note that, on geological time scales, it is sufficient to understand the solubility and migration behaviour of 238U, rather than assessing the inventory of all radionuclides in the decay chain.

18. T.J. Schröder, E. Rosca-Bocancea, J. Hart (2017). Safety assessment of uranium on very long timescales. OPERA-PU-NRG745

19. Schröder TJ, JCL Meeussen, Final report on radionuclide sorption in Boom Clay, OPERA report OPERA-PU-NRG6123, February 2017.

What is the confidence that disposal of radioactive waste in Paleogene clays is feasible and safe?

9. Synthesis and conclusions

A GDF in clay host rock is one of two options that are being investigated in COPERA; the other being a GDF in rock salt (Verhoef et al., 2020). COPERA builds on earlier work, especially on the previous two national programmes of research into geological disposal of waste in clay host rocks (CORA and OPERA). Although the first national programme, OPLA, and earlier Dutch research, considered only rock salt as a host rock, essential concepts for disposal of waste, in particular the use of a multibarrier system (OPLA, 1989) to contain the radionuclides and isolate the waste, were already developed at that time and are still being used today.

OPERA established a state-of-the-art methodology for producing a safety case using the capabilities of national organisations and international cooperation. This approach is also used in COPERA, which builds on and propagates many of the results of OPERA. The methodology developed in OPERA to show that acceptable levels of post-closure safety are achievable for any type of disposed waste is also used in COPERA. Further work has been carried out to examine in more detail the constraints on some of the parameter values used in a safety assessment. Distinguishing different ranges in which key parameters might lie allows discrimination between calculated results representative of a normal evolution scenario, alternative scenarios or what-if scenarios.

To date, most of the resources available in COPERA have been devoted to clay studies and, more specifically, to the clay formations available in the Netherlands. This knowledge needs to be extended, improved and eventually made site-specific in COVRA's staged work programme, in order to develop more detailed designs of the GDF and multibarrier system and to verify the assumptions made and the parameter ranges used in the post-closure safety assessment.

The waste forms that will be disposed in the GDF must alter and degrade by the action of water before any radionuclides can be mobilised from them, meaning that evaluation of how, and how much water flux can eventually penetrate into the EBS is a central focus of our studies. Alteration of waste consumes water, so the understanding of mechanisms of water movement and interaction in the clay host rock and EBS is key to the determination of waste alteration rates in the multibarrier system in a normal evolution scenario. Understanding the behaviour of trace elements in clay formations provides evidence of how radioactive isotopes of these elements would behave. For cost-effective management of the relatively small research budget, our engineered barrier system studies to address these processes were performed in the framework of the European Joint Programming (EURAD) project, through participation by COVRA staff or by students supported by COVRA.

At the time of the development of COPERA (Verhoef et al., 2020), implementation of an operational disposal facility was not foreseen until 2130 (I&E, 2016). Recent documents preparing the next update of the national programme have not amended this date (van Gemert et al., 2023). Site specific safety cases are therefore assumed not to be needed until after 2050. This COPERA safety case is thus - as was the OPERA safety case - conditional, because it is not site-specific and some key data are lacking. Nevertheless, relatively detailed material on the demonstration of the construction, operation and closure of the GDF is included in this COPERA Safety Case in order to give context for the R&D currently performed and to reduce uncertainties in the cost estimate, which is to be updated periodically. Practical engineering and operational aspects of the GDF design have therefore been treated at more length in the COPERA safety case than was done in the OPERA safety case.

This final chapter provides a synthesis of the current status of development and safety evaluation of a GDF in clay host rocks in the Netherlands, draws conclusions based on this and looks forward to future developments. Dutch policy on the decision-making process for implementation of the GDF in the Netherlands is being developed by Rathenau (Cuppen, 2022) in the framework of the national programme that is required to be updated every 10 years (EC, 2011), i.e., in 2025.

9.1 COPERA's role in strengthening the knowledge infrastructure

COPERA (2020-2025) progresses and supports national radioactive waste management policy, which calls for final disposal of Dutch wastes in a GDF. The main goal of COPERA is to develop the knowledge required for implementing safe and efficient geological disposal of waste in poorly indurated clays and rock salt, taking into account all the steps in the radioactive waste management chain. COPERA contributes to (Verhoef et al., 2020):

- strengthening national nuclear knowledge infrastructure and building an international network for geological disposal of radioactive waste;
- promoting a societal discussion on geological disposal of waste, informed by up-to-date knowledge and focussed on taking societal responsibility for a final solution for radioactive waste;
- consideration of multinational disposal options as a part of the dual-track strategy.

Universities are key to maintaining and expanding knowledge and passing this on to the next generations. Accordingly, COVRA has established long-term relationships centred on research into clay host rocks with Delft University of Technology (TU-Delft). This relationship has, for example, resulted in TU-Delft inviting and allowing COVRA to participate in their research drilling project studying geothermal wells and in TU-Delft acting as an Affiliated Entity of COVRA in EURAD, which allows the university to participate in this European Joint Programming. In the future, relationships such as that established with TU-Delft are also to be developed with other Dutch universities in order to nurture the next generation of experts in disposal of radioactive waste in the Netherlands. It is also planned to work as much as possible with international experts in the European Joint Programming in order to optimise COVRA's use of financial resources for research, and to ensure comprehensive coverage of the relevant scientific and technical issues.

An important linkage is with the Dutch Research Council (NWO) whose SECUUR project (Safe Environment for Clay Underground Repository) led by TU-Delft is studying the geotechnical properties of clays cored at about 400 m depth (Vardon et al., 2022). The end-user group consists of the Dutch nuclear regulatory authority (ANVS), the Dutch and Belgian waste management organisations (COVRA and ONDRAF/NIRAS), the Dutch and British Geological Survey (TNO and BGS) and the Belgian drilling company (SMET). Research projects such as SECUUR strengthen the national knowledge base, contribute to optimising drilling techniques for obtaining clay cores for research into geological disposal of waste and expand COVRA's international network. Further international networking examples contributing to COPERA are the NEA Clay Club, the scientific committee for the three-yearly Clay Conference, EURAD and IAEA. In EURAD, COVRA worked directly on engineered barrier system studies, while TU-Delft and TNO performed other clay related studies. COVRA participates in or follows meetings held by the IAEA network for countries with URFs.

COPERA organises yearly research meetings in the Netherlands. Researchers present their recent work at these meetings and actors with a designated role in the national programme (EC, 2011) are invited to participate, for example Rathenau, ANVS and Dutch Ministries. The benefit of the participation of such actors is that they can be informed by the researchers themselves. PhD, Bachelor and Master students also present results of their activities in these yearly meetings. EURAD also organises activities to help orient junior professional careers in the field of radioactive waste management and COVRA has supported the participation of such juniors at the EURAD meetings, where they can present their work.

The ERDO Association (Association for Multinational Radioactive Waste Solutions) was established in 2020 during the period of the current COPERA programme, and COVRA provides the secretariat. ONDRAF/NIRAS also studies poorly indurated clay as a host rock for a GDF and has become an ERDO member during COPERA (2020-2025).

9.2 Feasibility of constructing a GDF in Paleogene clays, its operation and closure

This COPERA Safety Case has gathered and integrated existing information and carried out studies for two different GDF concepts in clay:

- a single level GDF designed to dispose of HLW and LILW in a Paleogene clay formation at 500 m depth, as in OPERA;
- a multi-level GDF comprising several underground facilities in one or more Paleogene clay formations, with LILW disposed of at a shallower depth than HLW: i.e. the wastes are emplaced at different depths according to their hazard potential.

Both conceptual designs are dimensioned to contain the expected waste inventory that will arise over the next decades, also allowing for the currently envisaged increase in nuclear power in the Netherlands (van Gemert et al., 2023). The feasibility of constructing underground facilities in clay at depths less than 225 m has already been demonstrated by Dutch traffic tunnels and by the Belgian URF at Mol. A depth of 500 m or greater increases the isolation provided by the geological environment but may also present increasing engineering challenges. The tunnel support system envisaged for disposal of ILW at 500 m depth in the planned GDF at Bure in France (which uses data from the nearby URF at this depth), clearly shows that these challenges can be overcome.

Results from the on-going SECUUR project should provide further confidence in the feasibility of constructing a GDF at disposal depths appropriate to the Netherlands clay formations. SECUUR uses Belgian Boom clay cores extracted from Mol (Belgium), Borssele (Netherlands) as well as clay cores extracted from the Paleogene clay formations at Delft. The nature and variability of Paleogene clay properties and in-situ pore water pressures still need to be determined on a regional basis across the Netherlands.

Existing tunnelling techniques using Tunnel Boring Machines (TBMs) can be used to excavate clay host rock and generate stable open spaces for the emplacement of waste packages, following construction of a supporting liner system. The designs presented in this COPERA safety case include layouts and tunnel features that are practicable for construction with the current state of technologies. Constraints on construction, as identified in the Belgian design, have been taken into account to optimise designs, e.g., at intersections between transport tunnels and disposal tunnels.

In COPERA, the same techniques for emplacing waste packages are assumed as those currently used by COVRA: i.e., forklift trucks for packages up to 20 tons and air cushions for heavier loads. Air cushion vehicles are commercially available, which allows us to include cost estimates with a high degree of certainty. A larger tunnel inner diameter than used in OPERA allows stacking of the LILW packages with forklift trucks.

As in OPERA, different types of concrete are used for manufacturing the impermeable segments of the tunnel liner, the more porous backfill that fills the empty void space in the disposal tunnels and the low permeability concrete buffer that comprises part of the HLW supercontainer. There are options for the cement type, water to cement ratio and content of aggregates and cement with which the different types of concrete are made. In EURAD, COVRA, together with Belgian and German research organisations, investigated several types of concrete - especially those that are impermeable in engineering terms: the waste package concrete used for the conditioning of compacted waste in 200 l drums (LILW) and three different series of porous backfill.

Paleogene clays are treated as aquitards in groundwater management. Both the Paleogene clay and the concretes used in the EBS are porous media that have very low permeability and diffusion values which need to be known for the post-closure safety assessment. It is important to understand the methodology with which the most reliable values can be obtained and how these values might change during the long-term evolution of the multibarrier system. These values, as well as their expected evolution, have been obtained in EURAD. The knowledge developed in EURAD, as well as existing knowledge from work on compacted blocks of re-constituted excavated clay, allowed a simpler and less expensive procedure to be proposed to close the disposal facility.

9.3 Feasibility of siting a GDF in Paleogene clays

COPERA and OPERA are not siting studies. Nevertheless, it is important at an early programme stage to have confidence that suitable locations for a GDF can be found in the Netherlands. Paleogene formations at appropriate depths for HLW emplacement (200-1,000m) are present across the Netherlands. There are formations that have appropriate thickness and clay content to provide the necessary containment, although there are currently significant uncertainties in their depths and thicknesses. A palynological analysis of the available Paleogene clay material in existing core stores could significantly reduce uncertainties in depths and thicknesses. In addition, knowledge on the geotechnical properties of these clays is scarce. The NWO project SECUUR aims at increasing knowledge using freshly cored clays from Belgium at 225 m depth and well-conditioned clays from Delft at about 400 m depth. This may provide validation of some of the geoscientific assumptions used in OPERA and COPERA.

There is a significant shortage of fresh or well-conditioned Dutch cored clay available for research into geological disposal of waste. COVRA realises the potential sensitivity of localised geological investigations that provide input for its radioactive waste disposal programme. Consequently, we have taken a consensual and transparent approach to obtain access to boreholes and clay cores taken at relevant disposal depth, making clear that the results are part of generic clay studies and that the work does not represent in any way the commencement of a GDF siting programme.

9.4 The objective and design of the multibarrier system

The multibarrier system will contain all high-activity and long-lived waste that are currently stored at COVRA's premises and likely to be generated over the next decades. The safety concept for this system aims to isolate the waste and contain the radionuclides in the waste so that their radioactivity and radiotoxicity will never pose an unacceptable risk to people or the living environment at the surface. The hazard potential of the wastes, i.e., their capacity to cause harm if people came into direct contact with them, is initially extremely high, but it diminishes rapidly over the first hundreds of years after the waste has been emplaced in the GDF, then more slowly over future thousands of years. The safety concept places emphasis on assuring complete isolation and containment of the wastes over the early period. It also recognizes that small amounts of radionuclides from the waste will eventually move into the rock formations that surround the clay host rock and might be transported to people in the far future as the engineered barriers slowly degrade through natural processes. However, the multiple barriers in this system ensure that any releases will be so small that they can cause no harm to future generations.

The behaviour of man-made radionuclides in the waste forms is also impacted by the presence of natural radionuclides in other safety barriers. A start has been made in COPERA to determine the elemental compositions of the clay host rock, its pore waters and the concrete barriers, in order to put the concentrations of those elements that have radioactive isotopes in the waste form into perspective with those in the engineered and natural barriers and to provide analogue information on their chemical behaviour in these barriers. The natural concentrations in clay host rock of non-radioactive isotopes can be higher than the concentrations of some of their equivalent radioactive isotopes in waste forms. The transport of an element could occur as a radionuclide from the waste form in the EBS towards the clay host rock but also as a (non-)radioactive isotope from the clay host rock diffusing into the EBS.

The wastes are surrounded by relatively massive, engineered barriers. The volume of concrete can be 100 times larger than that of the waste in a disposal cell in the GDF, especially for HLW. The properties and behaviour of steel and concrete will dominate the evolution of the disposal cells containing vitrified HLW and Spent Research Reactor Fuel (SRRF) during the first thousands to tens of thousands of years, since the engineered barriers are designed to prevent contact between pore waters and the waste until the waste no longer heats the clay host rock. With the current

design, this period has been estimated to be 1,200 years for vitrified HLW. For SRRF, there is sufficient dissipation of heat in the EBS so that the clay host rock will not be heated. The times for which contact between HLW and pore waters is prevented by the EBS are longer than this thermal period. Currently, the required thicknesses of the steel and concrete in the HLW supercontainer that are needed for shielding during its emplacement are greater than the thicknesses required to achieve complete containment of HLW during the period in which the clay host rock is heated by the waste. That a larger thickness for shielding is needed in the operational phase than required for the containment in the post-closure phase is also the case for Molybdenum waste and spent ion exchange resins (LILW). The 200 litre drums of these two types of waste are encapsulated in 1,000 l concrete reinforced containers.
These reinforced containers inspire the optimization of the concrete buffer in the HLW supercontainer such as the use of aggregates with a larger density to increase shielding and reinforcement to limit the generation of cracks during concrete hardening of the buffer and the thermal induced stress by heating of the buffer by the waste .

The clay host rock limits access to the EBS of water that causes degradation of waste forms. In addition, the high strength and ultralow permeability of COVRA's various waste package concretes limits the access of water to the wastes and can control the release of radionuclides. Accordingly, understanding of the transport of water in concrete is important for estimating representative alteration rates of the waste forms. These representative rates are small, so that the mobilisation of radionuclides into a dissolved state in the multibarrier system is small and only very small fluxes can enter the clay host rock. Whilst dominating the EBS, the concrete engineered barriers comprise a small proportion of the total multibarrier system, which is dominated by the clay host rock. The rock formations that surround the clay host rock have currently been conservatively assumed to contribute only to physical isolation and not to nuclide retention.

9.5 How is the multibarrier system expected to perform?

As noted above, the most critical time over which the performance of the multibarrier system has to be assured is the first few hundreds to a few thousands of years, owing to the initially high hazard potential of the wastes.

However, safety assessments address much longer periods and consider how the multibarrier system performs for tens and hundreds of thousands of years. Eventually, changes in the natural environment, particularly those associated with far future glacial cycles, make quantitative estimates of future performance less useful, as their timing and durations are uncertain. Nevertheless, in common with other international safety cases for geological disposal of waste, the environmental impacts for the next million years are estimated in COPERA. At such long times, it becomes more appropriate to use other indicators of performance, rather than calculated radiation doses to far-future humans, e.g., fluxes of natural radionuclides in the surface environment.

Following the work in EURAD-1 ACED, COPERA has assessed how the multibarrier system with vitrified HLW and other types of waste evolve over these long periods. The approach of aiming to identify, where possible, realistic parameter values allows one to discriminate between the results calculated in OPERA using values that are representative for a normal expected evolution scenario and those that are more appropriate for what-if scenarios, meant to test the robustness of the system.

The expected behaviour is that the engineered barriers for heat-generating HLW will provide total containment of the man-made radioactivity for at least 35,000 years, after which the hazard potential of vitrified HLW has become less than uranium ore. Beyond 35,000 years, our calculations made in OPERA, shows that almost all residual activity that escapes from the engineered barriers will be contained by the clay host rock for hundreds of thousands to millions of years.

However, for some radionuclides, chemical retention is unlikely in the clay host rock and these more mobile radionuclides can move into the rock formations that surround the clay host rock, where their concentrations will be diluted and dispersed. Calculations made in OPERA confirm that no safety concerns result, since the contribution in radiotoxicity of these more mobile man-made radionuclides will be negligible compared to exposures from natural radionuclides in drinking water.

9.6 Open issues in the safety assessment

Future evolution scenarios are grouped into a normal evolution scenario, which is considered to represent the most likely behaviour of the disposal system, alternative scenarios that are less likely, what if scenarios that are unlikely but serve to test the robustness of the multibarrier system and human intrusion scenarios.
This is done in order to quantify and understand how the barriers act in concert to isolate the waste and contain the radionuclides. The distinction between the classes of scenarios is based on the features, events and processes that are considered in each and on the plausible ranges of parameter values that can be assumed to model the processes involved in the evolution of the multibarrier system.

The clay formation is an important barrier in the multibarrier system. Parameter values (e.g. diffusion, porosity, K_d values) for transport of elements have been obtained from modelling in OPERA and these values have been used to model the transport of radioactive isotopes of these elements from the wastes. Any model uses assumptions, and it is important to consider whether parameter values obtained can be verified with experimental results. In COPERA, a start has been made to use the experimental results obtained in OPERA in this verification process.

The determination of plausible ranges of parameter values for the engineered barriers requires understanding which processes are involved and interact in the evolution of the system of multiple barriers. Parameter values are usually obtained from laboratory measurements or experiments. The representativeness of experiments for the post-closure conditions in each barrier in the multi-barrier system needs to be evaluated. Determining parameter values such as diffusion values may require long-term experiments. All the components of the EBS have limited fluid contents in their pore systems and their degradation may consume the small amounts of water present, so that water inflow from the clay is necessary. But the corrosion rates used in the modelling may have been obtained from experiments in which a barrier material is exposed to an unrepresentatively large volume of water.

Such parameter values are used in assessing the safety for a normal evolution scenario but must be treated with caution, as they can be overconservative. In COPERA, a start has been made on means of determining the parameter values for the safety assessment using a more realistic approach to describe the interface reactions involved in dissolution and corrosion of waste forms and other engineered barriers. Natural and archaeological analogues showing preservation of materials in clay and concrete illustrate that degradation processes can be much slower than typically assumed in safety assessments, including those in which a normal evolution scenario is assessed.

Calculation of how the multibarrier system responds to any evolution scenario is necessarily a simplification of many complex and interacting processes. With increasing computational power, the need for simplifications will reduce in the future. The completeness of system understanding is also an important issue in safety assessment. It must be clear which processes or properties of barriers have been omitted from safety assessment models, and with what justification. OPERA included the following simplifications and assumptions, each of which makes the calculated results more conservative (i.e., pessimistic) than the realistically expected evolution:

- All LILW containers fail just after closure of the GDF;
  - This failure implies immediate contact between LILW and water, which was made additionally conservative in a so-called 'instant release' scenario in which all of the radionuclide inventory is assumed to enter solution immediately, uncontrolled by solubility limits except for uranium, thorium and neptunium, and begin to migrate in the porosity of the surrounding EBS materials. For all the radionuclides to enter solution fully immediately the containers fail, would require improbable/ impossible volumes of water to access the waste. It is more likely that the flux of water to the waste form is controlled by the low permeability of the clay and concrete.
- A short lifetime of the steel overpacks in the HLW supercontainer of 35,000 years;
  - The lifetime of the overpacks is expected to be longer than this, but using a chosen time to permit safety assessment release calculation is still current practice (e.g. Samper et al., 2022). It is not currently possible to estimate reliably the distribution in time or the mean value of package failure times. The computational power required for this estimation using chemo-hydro-mechanical modelling is, currently, too large.
- Simultaneous failure of all waste containers within a waste family;
  - Simultaneous failure leads to a higher peak in the calculated dose rate than assuming that failures are distributed over a long period.
- Extensive interaction between the clay pore water and the concrete tunnel backfill, the HLW supercontainer buffer and waste package concrete for LILW, leading to early degradation of their containment properties;
  - In OPERA, the radionuclides are assumed to be instantaneously dissolved for LILW, SRRF and CSD-c. Only, the time of failure differs: at the start of the post-closure phase for LILW and after the failure time of the supercontainer for SRRF and CSD-c. The radionuclides are then uniformly distributed in the concrete materials of the EBS and then to diffuse through the concrete at a higher rate than they subsequently diffuse in the clay: i.e., the concrete provides hardly any containment for the radionuclides released from the waste form. At time of OPERA, the diffusion properties for the different types of concrete were not well-known and assuming a uniform concentration in concrete from the start was a conservative choice. The knowledge obtained in EURAD during COPERA shows that the concrete barriers can have lower diffusion values for water than the clay host rock (see Chapter 6).
- No radionuclide sorption outside the clay formation in the overlying geological formations;
  - This simplification will be continued in generic (i.e., not site-specific) safety assessments, as there would otherwise be a wide variety of possible overburden formations to consider. In OPERA, only the youngest epoch (Oligocene) in the Paleogene was considered, i.e., Boom clay. Across the Netherlands, several other older Paleogene clay formations are also present. The safety assessment of the multilevel GDF proposed in COPERA may include sorption of overlying Paleogene clay formations.
- Relatively rapid dissolution of the vitrified HLW;
  - Chapter 6 and Chapter 7 show that parameter values obtained from experiments in which solid pieces of glass are exposed to a (cementitious) solution are highly dependent on the solid/liquid ratio. Low solid/liquid ratios in which glass specimens are immersed in unrealistically high volumes of aqueous solutions, lead to overconservative dissolution rates. Natural analogues show that glass is transformed into clay minerals and zeolites, which consumes the small amounts of water that are present in the disposal system, given the transport properties of the barriers concrete and clay and their geometry in the multibarrier system. Future work would permit more realistic estimation of ranges of glass alteration rates.

A number of processes and events that might lead to greater predicted releases of radionuclides were not treated in OPERA. These are listed below, together with comments on those issues where progress has been made in COPERA.

- A full assessment still needs to be performed of alternative evolution scenarios that might lead to different behaviour to that of the normal evolution scenario;
- Climate evolution and future glaciation cycles are expected, but not yet included as part of the normal evolution scenario, which assumes continuation of the current climate;
  - Their potential impacts will depend on the time at which they might occur and the geographical location of the GDF in the Netherlands, so the level of detail to be treated in an assessment can be site-specific or region specific.
- The normal evolution scenario has looked at radionuclide movement in water but the 'gas pathway' was not yet analysed in OPERA;
  - Work carried out during COPERA, indicates that, disposal tunnels with packaged vitrified HLW, the gas generation rates appear so low that perturbation of the clay host rock is not expected (Levasseur et al., 2021): i.e., there is no enhanced radionuclide movement in the normal evolution scenario. For the other types of waste, work on the impact of gas generation is on-going and it is possible that accounting for the properties of concrete in the EBS might allow a gas pathway to be omitted from a future

safety assessment (Blanc et al., 2024).

- The potential for criticality to occur in regions of the EBS holding enriched SRRF and the consequent impacts were not assessed in OPERA;
  - Criticality of SRRF has been investigated in COPERA (Koets et al., 2022). The current separation between the fuel elements as stored needs to be retained if the canister might be surrounded by water in the post-closure phase. The mechanical load at storage is much less than at disposal depth. For provision of this separation in the post-closure phase, investigations for manufacturing a mechanical stable solid in mm sized and smaller void volumes between the fuel elements have been done (Koets et al., 2022) but the predicted results need to be experimentally verified. Above all, studies of the Oklo ore body natural analogue (Bentridi et al., 2011) highlight the reduction in the probability of re-criticality occurring due to the low permeability of the clay alteration that was progressively formed around the ore, which reduced the access of water to the ore and hindered transport of fission products out of the system. Criticality in the current multibarrier system with low permeability of the concrete buffer and clay host rock, is therefore considered to be unlikely.
- OPERA and COPERA have only looked at the radiological impact of radionuclides that might move to the biosphere from the multibarrier system. There are also chemically toxic elements in the waste that could have health effects, if those elements migrate into the biosphere, and this requires evaluation.
  - Data on Paleogene clay show that they contain chemo-toxic elements such as lead as trace elements. The concentrations and speciation of such elements in the clay and clay pore waters will control the rate at which they can be mobilised from the waste and migrate into the clay, which will be a topic of a future study.

9.7 Other evidence underpinning confidence in the post-closure safety

Many of the materials used in the engineered barriers have been observed to be preserved in clays and concrete for long periods of time. Understanding the conditions under which they are preserved may improve our understanding of which processes need to be included in a normal evolution scenario.

An example is the microbial degradation of organic waste, which is usually envisaged to be much faster than purely chemical degradation of organic waste. During storage, organic waste degrades little, since the water content is too small for microbial activity, but water will contact organic waste in the post-closure phase. The preservation of examples of Roman wood in the Netherlands and of 2-million-year-old trees in Quaternary clay in Italy show that microbial activity can have negligible impacts on degradation for very long times if there is very little access to water. The limited transport of water through concrete in the EBS may thus control the degradation rate of organic waste. The rate of access of water to the waste may have implications for the required disposal depth of organic waste and the necessary thickness and clay content of the clay host rock.

The material with the largest volume in the engineered barriers is concrete. The existence of Roman concrete in many locations today illustrates that its physical properties and structural stability can be maintained for thousands of years, if well-engineered. The Pantheon in Rome is an extant, load bearing structure whose roof has maintained its integrity for around 2,000 years in mainly dry, surface conditions. In wet underground conditions, leaching is possible if the type of cement used to manufacture the concrete has not been carefully chosen. The existence of Roman concrete submerged below the sea shows that, for some compositions, leaching can be negligible for thousands of years.

At a broader scale, natural radioactivity, present in all rocks, soils, concrete, trees and other living matter and waters around us, provides useful yardsticks against which to compare the impacts of radionuclides from disposed waste. Radionuclide-specific comparisons are most useful, as they provide understanding of how natural radionuclides and radionuclides in the wastes might enter into and behave in our living environment. An evaluation has been made for carbon-14 (Neeft, 2018) and evaluations for other radionuclides in Dutch Paleogene clay have recently started.

In OPERA, trace elements concentrations were measured in Boom clay samples from several locations in the Netherlands (Koenen and Griffioen, 2014). Further analyses of Boom clay performed in COPERA, shows that, with increasing clay content, there is an increase in concentrations of non-radioactive isotopes of radio-nuclides that are present in the waste and of natural radionuclides. In some cases, these increases occur with increasing organic carbon content of Boom clay (e.g., for uranium) or inorganic carbon content (e.g., for strontium). Information on how the chemical properties of clay formations contribute to containment of radionuclides will increase when similar measurements, currently being carried out by the Dutch Geological Survey and Utrecht University on recently extracted clay cores from Delft, are completed. These analyses should provide in-situ distribution values between the solid and aqueous phases of the clay host rock. These measured distribution values are representative as input for assessing the normal evolution scenario.

Natural radioactivity levels in the Netherlands are typical of those across Europe and the unavoidable natural radiation exposures to which we are all subject are much higher than those from even our most pessimistically calculated releases from the multibarrier system. We live in, and human-kind has evolved in, a naturally radioactive environment. In the very far future (many millions or hundreds of millions of years), we expect the degraded waste forms, with their considerably reduced radioactivity, to have similar properties to a uranium ore body, containing mainly the residues of the depleted uranium wastes, thorium and natural radioactive potassium. These residues will either become more deeply buried and isolated in earth's crust by further deposition of sediments, or will be eroded away by natural processes, with their contents being distributed among, and becoming part of, the natural radioactive background.

Confidence in the reliability of the safety assessment calculations performed in OPERA is further enhanced by the fact that the results are broadly similar to those estimated independently for a wide range of wastes and host rocks, in other national programmes.

9.8 Progress in COPERA

There were uncertainties in several areas of OPERA, and, as discussed in section 9.6, assumptions and simplifications were needed to establish the safety assessment models and calculations. Some of these uncertainties have begun to be addressed in COPERA. Possible solutions to obtain more disposal representative performance have been described in section 9.6. The main areas that were identified in the OPERA Safety case for further work and that have been progressed in COPERA are:

- Evaluation of the generation and the behaviour of corrosion gases in the engineered barriers system and their behaviour in a Paleogene clay;
  - The generation of gases has been addressed during COPERA in the EURAD-1 ACED project (Blanc et al., 2024) and their behaviour in clay host rock was studied in the EURAD-1 GAS project: e.g., Levasseur et al. (2021). The work on gas generation stresses the importance of estimating corrosion rates for configurations representing realistic disposal conditions, taking into account the properties of the engineered barriers and clay host rock, rather than using results based on experiments on metal samples exposed to unrealistic quantities of cementitious solutions (Blanc et al., 2024).
- Improving understanding of the nature and rates of interactions between the Paleogene clay and the tunnel liners and other cement-based barriers;
  - Improvement has also been obtained through the EURAD-1 ACED project (Deissmann et al., 2021). The rates of these reactions are highly dependent on the diffusion properties of concrete and clay host rock. The next objective is to study the impact of these interactions on radionuclide distribution values in the solid and aqueous phase in concrete and clay host rock. This impact will be investigated in the EURAD-2 Radionuclide mobility under perturbed conditions (RAMPEC) project.
- Testing alternative cements and concretes for components of the engineered barrier system that would be appropriate in the deep Paleogene clay;
  - During COPERA, in the EURAD ACED and MAGIC projects, concretes made with CEM III/B, CEM III/A have been tested as alternative formulations to concrete made with CEM I. Concrete made with CEM III/B appears less vulnerable to leaching than CEM I, which might enhance its durability. In addition, concrete made with CEM III/B provides a reducing environment in which low and predictable anaerobic corrosion of steel takes place. Although much work has been done, some confirmation of the results is required, as well as further experiments on the retention of natural radionuclides in concrete.
- Definition and evaluation of alternative GDF design concepts that might be suitable for Paleogene clays;
  - Two alternatives have been developed: a single level GDF (as in OPERA but with improvements to the design where tunnels cross each other, see Chapter 4) and a multilevel GDF layout that could be implemented at sites where several Paleogene clay formations are accessible, which is usually the case. This could allow disposal of waste at depths related to their hazard potential. Further work on GDF design is currently foreseen in the EURAD-2 'HLW Repository optimisation including closure' (OPTI) project;
- Developing viable systems for moving and emplacing larger waste containers in the underground part of the disposal facility;
  - The system has been improved by looking more closely at how COVRA currently moves large, heavy objects in restricted storage areas and by increasing the internal diameter of the disposal tunnels, so that a forklift truck can be operated.
- Further studies on how any requirements for retrievability of waste packages can be incorporated in the GDF design operations and safety case development;
  - Chapter 4 and 6 show the definition of requirements for the safe retrievability of waste packages: i.e., providing sufficient manoeuvring space that is stable for a long period, at least in the operational phase, and ensuring sufficient radiation protection.
- Establishing mechanisms for knowledge maintenance and transfer over the decades and generations leading up to eventual disposal;
  - Currently, mechanisms for knowledge maintenance and transfer over a few decades have been achieved by documentation and archiving and by educating and working with Bachelor, Master and PhD students.

Complementary to COVRA's technical disposal studies, the Rathenau Institute is currently looking at a societally based approach to identifying possible siting areas and locations for a GDF.
The inclusion of all Paleogene clays has increased the area in which potential locations for a GDF might be found.

9.9 Areas for further work to improve the design and safety case

Many areas in OPERA that were identified as requiring further work have been dealt during COPERA (see previous section). Those that are left are mainly related to further characterising Dutch clay host rock and to improving safety assessments:

- Improving knowledge of the lithological, geotechnical, hydrogeological and geochemical properties of Paleogene Clays at disposal depth by testing and sampling in boreholes;
- Taking reliable porewater samples in the Paleogene Clays and the under- and overlying formations to gather palaeo-hydrogeological data (e.g., environmental isotopes) to help understand and quantify rates of diffusion and deep flow and transport in and around the Paleogene Clay;
- Measuring in situ pore water pressure and hydraulic pressure gradients in the Paleogene clays at disposal depth and their evolution;
- Performing analysis of additional 'alternative evolution' scenarios, especially those for different climate states;
  - As explained in section 9.6, the impact of climate change on the behaviour of the multibarrier system is region specific in the Netherlands.

An issue emphasized in COPERA is the definition of plausible ranges of parameter values that are used in safety assessments for different type of scenarios. The normal evolution scenario contains a range of cases (or realisations) to encompass the expected range of variability and uncertainty in key parameters that affect system behaviour. The transport of water in concrete and clay and the water consumption rate by the waste form during alteration has been identified as a key process for determination of a plausible radionuclide dissolution rate in a normal evolution scenario. The use of this understanding is to be further explored in order to make the calculated dose rates in the normal evolution scenario more realistic and help to distinguish the calculated results from other scenarios.

The clay host rock is an important barrier to minimize radionuclides entering the biosphere but measurements of non-radioactive isotopes present in trace amounts in the clay host rock do not seem to be explicable using transport models employed in assessing the safety in the normal evolution scenario. First estimates of reasonable values can be made by comparing the concentrations of radionuclides in the waste with the concentration of non-radioactive isotopes in the clay host rock and clay pore water.

9.10 Overall conclusions

Since the publication of the OPERA Safety case six years ago and the initiation of COPERA (2020-2025), advances have been made in developing safety cases at COVRA and also in refining GDF design concepts. The overall conclusions of OPERA on the safety of a GDF in Dutch clays remain unaltered. Nevertheless, with increased knowledge, we are able to make a progressively more refined and detailed analyses and expect to improve this further as more research takes place. Most of the additional inputs for updating the safety case have been obtained through participation in projects in the European Joint Programming framework. The current integration of up-to-date knowledge provides valuable input for supporting Dutch radioactive waste management policy and for reporting by the regulatory body in the framework of the European Waste Directive (EC, 2011).

A focus of the COPERA research has been on demonstrating the feasibility of the GDF design, including its operation and closure, in order to reduce uncertainties in the cost estimate. This has involved studying more closely work done in Belgium, in other national programmes (e.g. through IAEA meetings and documentation by ITA-AITES), as well as past and current practices in the Netherlands. COVRA has high confidence that a GDF for both LILW and HLW can be safely and efficiently constructed and operated in clay host rock in the Netherlands, and that future changes in the waste inventory can easily be accommodated in the designs currently being developed.

Improving knowledge of the characteristics of concretes most suitable for disposal of waste and understanding their chemo-mechanical evolution in the post-closure phase was another focus of the research, allowing improvements in cost-effectiveness since the OPERA study.

COPERA has built on OPERA, which built upon CORA and OPLA, thus maintaining the essential continuity of expertise and knowledge in the Netherlands. A significant development from OPERA to COPERA is that the focus on Boom clay has been widened to take into account other Paleogene clay formations that can be present at different depths. This implies that the GDF design may be optimized by disposing of wastes at different depths, according to their hazard potential.

COPERA(2020-2025) has cooperated with the Delft University of Technology to carry out some deep geological sampling. Geotechnical testing of the extracted clay cores in Delft in 2022 for the DAPWELL project took place through SECUUR, a programme mainly funded by the Dutch Research Council. COVRA contributes to research projects such as SECUUR, in which a knowledgeable end-user group of experts works together with relevant technical organisations to ensure a sound national basis for the Dutch disposal programme..

OPERA already showed that the proposed Dutch multibarrier system is capable of providing high levels of safety that match those estimated for GDFs in other national programmes and that would easily meet national and international standards. In practice, COPERA has shown that the assumed normal evolution scenario for safety assessments in OPERA was highly conservative. Improved understanding of the concrete engineered barriers and more developed modelling of the behaviour of trace elements in clay host rock, allow us to make more realistic estimates of performance and these indicate even lower safety impacts than those calculated in OPERA. Measurements on trace elements in the extracted clay cores and clay pore water will provide data that will improve the safety assessment models. But, even with the conservative assumptions in OPERA, the forecast potential radiation exposures of people were far below exposures to natural background radiotoxicity and would not occur until tens or hundreds of thousands of years into the future.

A parallel COPERA study has been prepared on the safety and feasibility of disposal in a GDF in salt. Both host rocks rely on the depth and stability of the surrounding rock formations for isolation of waste and both provide containment of radionuclides in the waste. The characteristics of the waste families and many of the approaches developed in OPERA are also directly applicable to the evaluation of a multibarrier system with rock salt. The differences in the functioning of the clay and salt multibarrier systems depend on how water comes into contact with the engineered barriers and the waste form and how radionuclides are contained by the multiple barriers if they are released from the waste form. Although the waste inventory in both multibarrier systems is the same, the mechanisms of radionuclide mobilisation from each waste form and its associated migration rates are different, as are the mechanisms and scenarios for how radionuclides might enter our living environment. With our current state of knowledge, a choice between the two host rocks cannot be made with respect to safety. Both host rocks offer viable solutions to waste packaging, management and GDF scheduling, and will have different costs.

9.11 Looking forward

Although the COPERA(2020-2025) research budget for a GDF in clay has been lower than in OPERA, the continuity of the funding as well as the experience established by researchers in OPERA made it more feasible for Dutch organisations (TNO, TU-Delft and COVRA) to participate in EURAD. EURAD is intended to assist in developing a European knowledge base. The research carried out by Dutch participants makes the Netherlands an active contributor to this knowledge base.

COPERA and OPERA make important contributions to satisfying the Dutch obligations under both the EC Waste Directive (EC, 2011) and the IAEA Joint Convention by showing that progress has been made for a GDF in the Netherlands (I&W, 2020). The continuity of COPERA maintains expertise in the Netherlands and allows participation in national and international initiatives. A road map for the next phase of COPERA(2025-2030) has been prepared and is presented in Chapter 10.

Finally, we note that the present report is a scientific/technical document, describing engineering and geological requirements needed to assure that a safe GDF can be implemented in the Netherlands. We are, however, fully aware that a successful GDF programme must address both societal and technical issues. Globally, the greatest obstacles to geological disposal of waste have been those related to achieving sufficient public and political support for the concept itself and, most specifically, for siting work, including exploratory drilling.

COVRA's transparency policy aims at providing full information to the public at all times. The Rathenau Institute is currently looking at a societally based approach to identifying possible siting areas and locations for a GDF. COVRA's personnel working in COPERA have also given guest lectures at universities and in other national fora in which geological disposal of waste can be discussed. The public is also able to become more familiar with geological disposal through guided tours of COVRA's premises. This report is also a contribution to publicising the progress being made in the disposal of waste in the Netherlands and will be presented at a public information meeting after its publication.

How to manage the necessary knowledge to support the long-term decision making about disposal?

10. Roadmap for the future Dutch GDF programme

The findings in safety cases are used to select and prioritise the R&D activities to be carried out in the Dutch disposal programme over the coming years. Conditional safety cases will be made in the next decades to steer the research with each a roadmap as a final chapter. COPERA will run at least until 2050 (Verhoef et al., 2020). Currently, interesting developments are occurring internationally in site selection, construction and operation of GDFs. Changes are also occurring in the Dutch national programme. The rolling agenda of COPERA is updated every 5 years in order adjust to changing national and international landscapes. The publication of this safety case is aligned with the review cycles of the national programme on radioactive waste (NPRA) set by the Ministry of Infrastructure and Water Management every 10 years. In 2023, this Ministry initiated a public consultation about the scope of the NPRA (Van Gemert et al., 2023). Several scenarios addressing the generation of radioactive waste from potential future nuclear plants were identified and a Strategic Environmental Assessment for these plants was defined (Van Gemert et al., 2023). On 28 February 2024, the Dutch parliament voted to include 4 large nuclear power plants in these scenarios instead of 2 (Erkens, 2024). The nuclear landscape will certainly change drastically in 2035 if all proposed nuclear plants are realized, and the volume and the characteristics of waste will change, especially if the spent fuel from the nuclear power plants is not reprocessed. In all these scenarios, the current target date of 2130 for emplacement of all radioactive waste in a GDF has been left unaltered. This roadmap for a future GDF in clay host rock therefore assumes a long-term storage period of slightly more than 100 years. The roadmap focuses on the maintenance and development of scientific and technical knowledge related to the characteristics of the waste and their evolution in a multibarrier system with a clay host rock, and the technologies for the safe implementation of disposal of waste.

For 2130, or any other target implementation date, it is relevant to note that past experience in Europe is that exploratory drillings specific to geological disposal of HLW has begun about 40 years in advance of development of a final site. While this long period has been mainly due to societal and political difficulties, time was also needed to allow technical solutions and safety assessment approaches to develop and mature. The Netherlands is in a good position to build on this knowledge base and take advantage of the advanced nature of its national geoscience database. For example, the Netherlands is one of the few European countries that have implemented successfully the European INSPIRE Directive (EC, 2007). Through the Dutch programme BasisRegistratieOndergrond (BRO), all available underground data are accessible for any citizen and company through DINOLOKET, a website controlled by the Dutch Geological Survey of TNO, hosted by the Ministry of the Interior and Kingdom Relations. This implementation of INSPIRE should allow a more efficient selection of suitable locations for exploratory drillings. At present, COVRA's participation in drilling projects is limited to those that have been undertaken for a purpose other than geological disposal of waste, but which have allowed COVRA to participate.

10.1 Drivers for COPERA

The planned long-term storage of Dutch wastes for over 100 years leaves the Netherlands ample time to learn from experience in other countries, to carry out research and to accumulate the knowledge to make well-founded decisions. The current safety case is a further conditional generic safety case following the OPERA Safety case of 2017. Some assumptions have been verified and, where necessary, updates have been made. In the next decades, more iterations of the conditional safety case will be carried out. The principal drivers for research remain the same as defined in OPERA (Verhoef et al., 2017):

1. Strengthen confidence in the safety of disposal; for a GDF in clay this involves investigating the currently preferred Paleogene clays, considering potential design options and modelling the post-closure safety performance;
2. Assess the disposability (see Box 10-1) of different types of wastes and waste packaging families; investigating waste packaging options and requirements for collection, treatment and conditioning of waste families to facilitate their eventual disposal;
3. Assure adequate funding for disposal of waste, based on regularly updated cost estimates for the GDF; identifying, and, where possible, optimising cost-determining features of a GDF.

Box 10-1: Disposability of waste

To set waste acceptance criteria, the 'disposability' of the waste and its packaging is assessed. The disposability of the waste is determined by collection, treatment and storage conditions and the requirements for disposal, as reflected in the RMS (see Figure 3-4):

1. The available knowledge about the waste. All the necessary details of the physical and chemical properties, radionuclide content etc. of the materials that may act as waste forms are usually obtained during the collection and the treatment of the waste. This knowledge also determines whether the radio-activity of the waste will decay sufficiently during its storage to allow the materials to be re-used.
2. How the conditioned waste is expected to behave in the multibarrier system. Choices for the treatment of waste are currently made primarily to ensure safe storage of the waste. These choices can also affect the durability of the waste packages in the multibarrier system, potential release mechanisms and their associated radionuclide release rates and the behaviour of the host rock in the post-closure phase.
3. How the waste form and waste package may have altered during storage. The extent of the alteration of the waste form and package is determined by the physical and chemical processes taking place, by the storage conditions and by the storage period.

COVRA is responsible for developing iterative safety cases and uses these cases as an instrument to steer research and manage knowledge over many decades. The disposal concept and the costs associated with this concept will be continuously updated by new developments in civil engineering and radiation protection. Figure 10-1 shows the cycle that will be run for the next decades, with the post-closure safety assessments remaining conditional until a site has been selected.

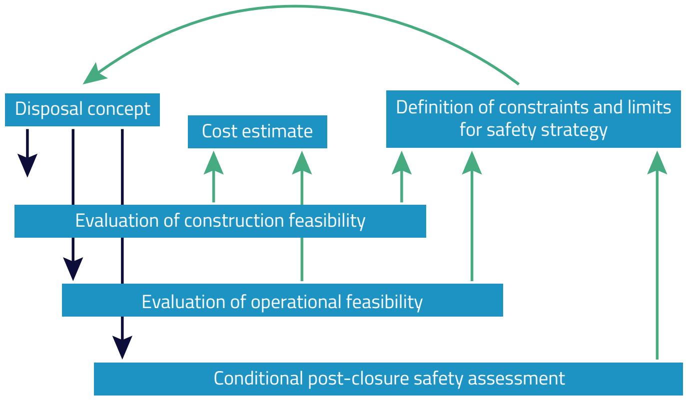

Figure 10-1: Cycle for updating disposal concept and costs.

10.2 Key topics for a GDF in clay host rock

From an international perspective, COPERA is modest in scale and scope. This reflects the long-term policy context in the Netherlands, which allows COPERA to acquire knowledge efficiently through international collaboration and through selective research activities within the Netherlands. The limited size of COPERA's budget for studying both a GDF in clay host rock and a GDF in rock salt, as well as multinational solutions, requires setting priorities for the research activities. The priorities are determined by how much additional research contributes to the three drivers listed in section 10.1. As described below, the same methodology as developed in the OPERA Safety (Verhoef et al., 2017) has been used to assign priorities to each component in the disposal system of engineered barriers and natural barriers.

Based on the future research needs discussed in this report and progress made in the latest EURAD European Joint Programme of R&D, the key topics for future research identified earlier (Verhoef et al., 2020) have been slightly revised. Figure 10-2 shows the key topics for each component in the multibarrier system; these topics are described below in more detail.

10.2.1 Biosphere - Priority 4

The biosphere is not part of the multibarrier system but acts as the receptor for any radioactivity that moves upwards from the geo-sphere. The safety assessment needs to model biosphere processes that determine how people might be exposed to radionuclides that have left the multibarrier system. The Netherlands disposal planning is at a conceptual stage and therefore, the IAEA reference biospheres (IAEA, 2003a) are currently sufficient to assess the safety of the multibarrier system. However, a study will be required on the origin and transport of the naturally occurring radionuclides in drinking water in the Netherlands, to verify the models employed in the safety assessment calculations.

All three Dutch research programmes have looked only at the impacts of radioactive elements that might move to the biosphere. There are also chemically toxic elements in the waste materials that could have health effects if they migrate to the biosphere, and this requires evaluation. The proposed investigations of trace elements in Paleogene clays as well as underlying and overlying Paleogene sands would increase knowledge of the migration of chemically toxic elements in the natural barriers. The evaluation of the potential impacts of chemically toxic materials in the engineered and natural barriers will be performed when the radiological exposure scenarios have been completed.

10.2.2 Surrounding rock formations

The rock formations on top of the clay host rock contribute to isolation. At any site, there will likely be several clay formations that also contribute to containment. Each clay formation is overlain and underlain by a sand formation. The distributions of natural concentrations and chemical speciation of major and trace elements in these Paleogene sand formations can serve as input to determine how dissolved species migrate in Paleogene clays and through the whole Palaeogene sequence.

10.2.2.1 Salinity in Paleogene sands - priority 2

The multilevel design of the GDF introduced in this safety case increases the importance of knowledge about the Paleogene sand formations that surround the clay formations. COVRA's preference is that suitable Paleogene clays are surrounded by confined sandy formations (Griffioen et al., 2016; PCR, 2013) whose pore waters are too saline for potable water extraction. The extent of their confinement needs to be investigated across the Netherlands.

10.2.2.2 Salinity in deeper groundwaters - priority 3

The national hydrogeological model (LHM) was extended in OPERA to calculate the transport of radionuclides from Boom clay at 500 m depth to the biosphere (Valstar and Goorden, 2017; Valstar and Goorden, 2016). Increases in salinity could increase the travel time if salinity was taken into consideration when deriving the migration parameter values used for the post-closure safety assessment. The increased knowledge referred to in section 10.2.2.1 can be used to further extend the LHM by incorporation of the salinity and by considering Paleogene clays other than Boom clay. The available information of the LHM at shallower disposal depth has not yet been included in a safety assessment.

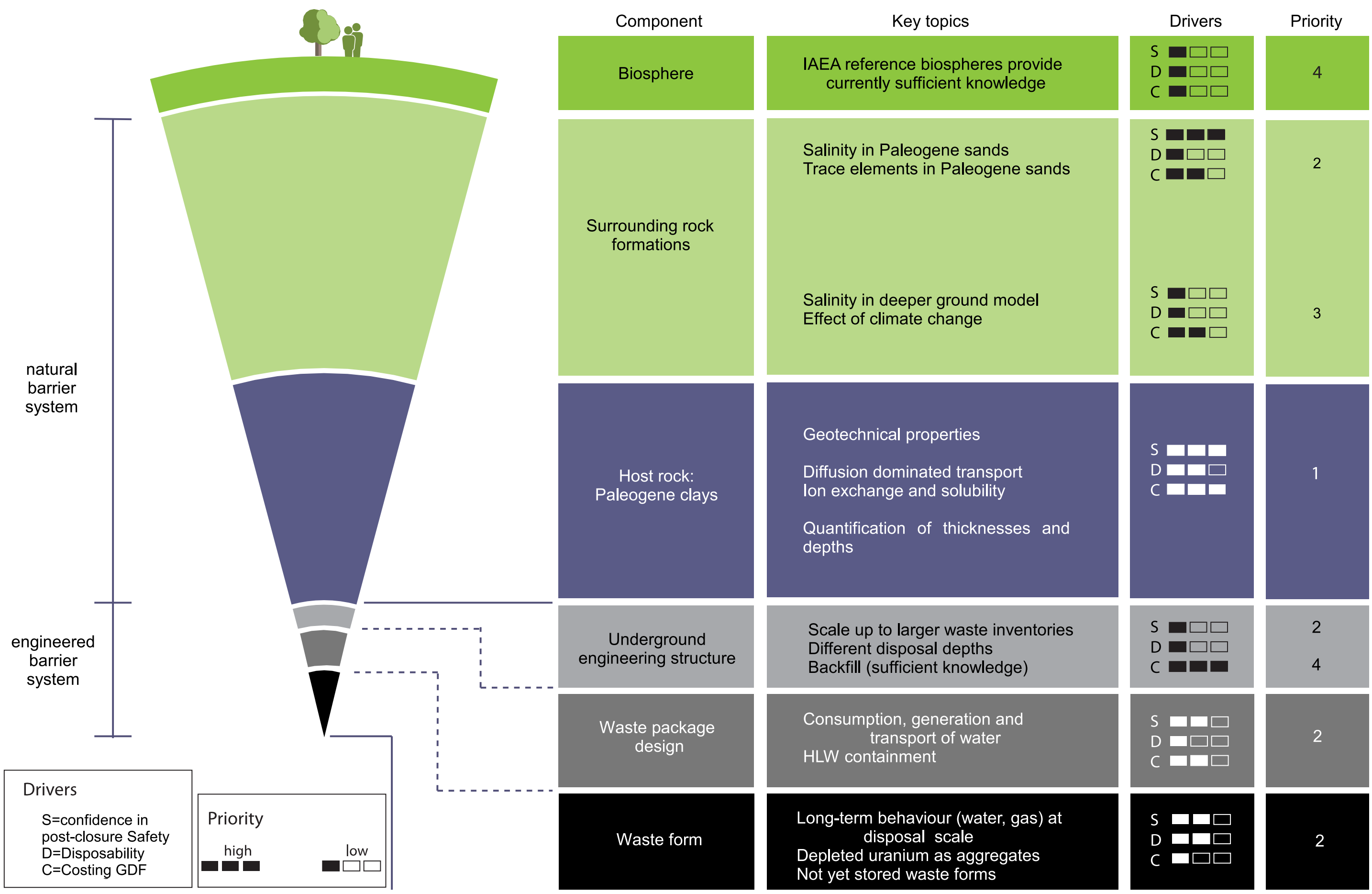

Figure 10-2: Key topics for research into geological disposal of waste in clay host rocks according to the component of the multibarrier system. Research for the construction of the geological disposal facility and biosphere are also included.

10.2.2.3 Trace elements in Paleogene sands - priority 2

Knowledge of the major and trace elements in the pore water in the Paleogene sand formations can be used as input to modelling to determine effective diffusion values of elements in Paleogene clays, and also for obtaining radionuclide-specific yardsticks as complementary safety indicators (IAEA, 2003b) for the dose rate calculated with reference biospheres.

This safety case shows that Paleogene clays also contain chemo-toxic elements such as lead, which is also present in the surrounding Paleogene sands. The waste should only contribute negligibly to the flux of chemo-toxic elements from the clay towards the sand formation. The aim is that knowledge of the current distribution and mobility of naturally occurring radioactivity and chemotoxicity in the Paleogene clays and sands should put in perspective the predicted mobilisation and migration of man-made radionuclides and chemically toxic elements from the GDF.

10.2.2.4 Effects of climate change - priority 3

In both OPERA and COPERA, potential hydrogeological behaviour as a function of changing climate and glacial cycling has not been assessed. However, assessing this is important for a GDF in clay host rock, most specifically in the northern Netherlands, since ice caps have covered only this part of the current Dutch territory. Glacial loading compresses the clay but also affects the movement of rock salt beneath the Paleogene clays. Local thinning of Paleogene clay formation can take place due to rise of salt domes. During periods of ice retreat, meltwater flow beneath an ice sheet may locally deeply erode the overburden, this has occurred to a depth of 600 m in the North of the Netherlands (Ten Veen, 2015) and produce large volumes of meltwater that would dilute the concentrations of any radionuclides present locally in the groundwater system. It will be essential to look in more detail at the likelihood and consequences of such a scenario for a GDF in the northern part of the Netherlands.

Glacial loading can also be a driving force for enhanced water flow beneath an ice sheet in other parts of the Netherlands. Permafrost can penetrate to depths of tens, perhaps hundreds of metres in areas that are not covered by ice sheets, leaving unfrozen ground beneath any lakes that are present. These patches of unfrozen ground (taliks) can act as recharge or discharge points for groundwater that might be mobile in formations below the permafrost. The studies for assessing the confinement of sand formations mentioned in section 10.2.2.1 should help to assess how flow might be affected under permafrost conditions. The results obtained in the Cryo-hydrology Network, CatchNet, a collaborative research project funded by SKB, NWMO, BGE and COVRA, should increase our knowledge about transport processes in cold-climate conditions, and these will be transferred into the Dutch situation.

The majority of recent studies suggest that there will be a prolonged warm interglacial period, possibly out to well over 100,000 years (Ganopolski et al., 2016) even if CO_2 emissions are controlled. The hazard potential of vitrified HLW has already reduced to that of uranium ore after 20,000 years (Gruppelaar et al., 1998). Any mobilisation of residual activity from the GDF should be set in the context of large-scale remobilisation of naturally occurring radioactivity in the subsoil surface sediments by the large rivers and subglacial waters that will exist as an ice-sheet melts.

10.2.3 Clay host rock

The clay host rock is the most important barrier in the multibarrier system as it controls the performance of the EBS and provides containment of radionuclides. Improving knowledge on how the clay host rock performs and evolves is critical for understanding and quantifying its contribution to containment in the post-closure phase. The feasibility of constructing a geological disposal facility at depths down to 225 m has already been demonstrated in Belgium in Boom Clay. Uncertainty in geotechnical properties of clays at the greater disposal depths foreseen in the Netherlands is high, owing to lack of data. Preliminary results show that the stiffness of the clay will increase with the increasing salinity of pore waters that will be encountered at these depths (Nguyen et al., 2013). A GDF is easier to construct in a stiffer clay, but the Excavation Damaged Zone around all openings may be larger with a higher fracture density. This might lead to changes in the normal evolution scenario in the post-closure phase.

The geotechnical properties of re-constituted clay were investigated in the second Dutch research programme (Barnichon et al., 2000) due to lack of quality of the available drill cores. Good quality cores became available during COPERA(2020-2025). Geotechnical measurements on these cores will contribute to providing a plausible range of parameter values for construction of the GDF. Plausible ranges of element specific diffusion values in the clay host rock also need to be defined for the post-closure phase using core sample materials.

10.2.3.1 Geotechnical properties - priority 1

The geotechnical properties of the Paleogene clays (Ypresian and Landen) cored at about 400 m depth in Delft (Vardon et al., 2022) are being studied as a function of salinity in the Dutch Research Council (NWO) project, Safe Environment for Clay Underground Repository (SECUUR) that is led by Delft University of Technology. Boom clay cored in Mol at 225 metre depth is used as a reference since the clay at that location has been well characterized.
A modest contribution to the financial resources for the execution of research proposed in SECUUR has been provided by COPERA (2020-2025), which began funding two PhDs in 2023. The remaining clay borehole cores that are left in 2028, at the end of SECUUR, can be further studied and further research on geotechnical properties will be defined near the end of this NWO Project.

10.2.3.2 Diffusion dominated transport - priority 1

Reliable in-situ permeability measurements in the Dutch Paleogene clays have so far not been possible, due to the disturbance made by coring and the necessary period to recover. Geological processes such as glacial loading and burial may give rise to a pore water pressure in the clay that is higher than expected at the sample depth. Such pressure anomalies are present in clays if diffusion values for water and water permeability are very small. A larger anomaly will indicate a smaller permeability and is therefore important knowledge for the post-closure safety (Neuzil, 2015). COVRA will seek opportunities to participate in projects to obtain in-situ pore water pressures in monitoring boreholes in Paleogene clays that are accessible for a sufficiently long time to recover from the disturbance.

Because of the low permeability of clays, water movements are slow, and transport of radionuclides is expected to take place predominantly by diffusion. There is work in Belgium showing evidence for diffusion by analysis of long-lasting experiments e.g. Aertsens et al. (2023). There is, however, as yet, insufficient direct evidence for the rates, and impacts on pore water compositions assuming diffusion dominated transport in Paleogene clays in the Netherlands. There are several techniques to gain experimental results in order to determine diffusion values from non-retarded elements present in the clay host rock, such as chlorine profiles and helium.

One of the most direct techniques to assess diffusion within clay and obtain parameters for diffusion is by measuring the out-diffusion of chlorine as a function of the distance into the clay across sequences where the clay is in contact with a freshwater aquifer and then modelling the chlorine profile (Mazurek et al., 2011). However, the Paleogene clays that we are considering as potential GDF host formations are surrounded by Paleogene sands that also have saline pore waters. This means that the assessment of chlorine diffusion profiles is unlikely to be useful. An alternative could be modelling measured helium profiles. Ion exchange of thorium and uranium in groundwaters with clay minerals and immobile dissolved organic matter contributes to the chemical containment of these elements in clays. These natural radionuclides are present in higher amounts in the Paleogene clay than in the surrounding Paleogene sands. As they undergo radioactive decay, helium is produced and is retained within the clay host rock when the diffusion values for helium are small enough (Mazurek et al., 2011). Measurements of gamma intensity in boreholes e.g. Vardon et al. (2022) can be used to obtain data and determine the necessary numbers and spacings of samples to be analysed from the clay cores and other subsoils (Rufer et al., 2024).
Helium migrates at a faster rate than water. The water permeability would be too high if there is no water pressure anomaly, then such an expensive helium analysis of freshly cored clay is probably less useful.

Other methods to determine in situ diffusivity values at formation scale require long-term experiments in which traces of dissolved species are introduced in the clay host rock at a borehole and measured after some years in another borehole at a distance from the borehole in which the species was introduced or an Underground Research Facility (URF). Such work would look at the behaviour of dissolved species that are not ion-exchanged with solid phases in the clay host rock. These experiments would use dissolved species are not present in Paleogene clays in order to facilitate the measurement. In the Belgian URF in Mol, tritiated water has been used e.g. Aertsens et al. (2023). The British Geological Survey is currently investigating whether non-radioactive dissolved gas can be used as a tracer.

Anaerobic corrosion of metals produces hydrogen gas and, as many metals do not form hydrides in this corrosion process, the hydrogen is expected to enter the clay host rock in these cases. Perturbation of the clay host rock would occur if the generated gas cannot be sufficiently dissipated by diffusion (Verhoef et al., 2020).
Water is consumed in any alteration of the waste form that leads to the formation of gas (Mladenovic et al., 2024). The clay host rock and the engineered barriers, such as concrete limit the access of water to the waste form. Situations in which gas generated by the waste form and package cannot be sufficiently dissipated by diffusion in the clay host rock, need to be identified by modelling the transport of water in the multibarrier system to find the water consumption rates at which dehydration of the clay host rock or concrete does not take place. This will allow identification of evolution scenarios or cases (normal, alternative or what-if) in which gas migration other than diffusion in clay host rock can take place. The results obtained in EURAD-1 GAS are therefore foreseen to be included in the next safety case, before the evaluation of the national programme in 2035.

10.2.3.3 Ion exchange and solubility - priority 1

Radionuclides from the GDF can enter the clay host rock after their release in solution from the engineered barriers and it is important to know the relative concentrations of these radionuclides in the pore water and on the clay minerals. The last two Dutch research programmes (CORA and OPERA) had made assumptions for the behaviour of radionuclides in a Paleogene clay but a verification of these assumptions for the post-closure safety assessment has not been made. The evaluation of the experimental data obtained for trace elements in Paleogene sands, combined with the study of trace elements in Paleogene clays is expected to verify assumptions, contribute to understanding of how radionuclides might migrate in Paleogene clays and provide a justified plausible range of diffusion values in a clay host rock for the post-closure safety assessment.

A start has been made in this safety case by analysing the experimental results obtained in OPERA from a single Paleogene (Boom) clay using samples taken across the Netherlands (Koenen and Griffioen, 2014). Clay pore water characterisation is not possible from Paleogene clay samples available at the Dutch Geological Survey in Zeist, since they are stored dry at room temperature without any encapsulation. The dry conditions enhance the oxidation of pyrite present in these clays, acidifying the clay pore water.

In COPERA (2020-2025), some funding has been allocated for measurements in Paleogene (Landen and Ypresian) clay cores extracted in Delft, at around 400 m depth (Vardon et al., 2022). These include data gathering by the Dutch Geological Survey and Utrecht University on the mineralogy, trace element concentration and clay pore water composition. These clay cores are stored in Delft at 4 °C and are sealed with paraffin wax and resin, and therefore considered the best available clay cores in the Netherlands for this geochemical characterisation.

In OPERA (2011-2017), Paleogene (Boom, Watervliet and Asse) clay cores were extracted in Borsele in 2011 (PCR, 2013).
The mineralogy and main components in Boom clay and Boom clay pore water have been measured in OPERA but no trace elements in the clay pore water were measured. These 'Borsele' cores are also stored dry at room temperature at COVRA's premises but are encapsulated in PVC liners. Their quality for geochemical characterisation is therefore not as good as the 'Delft' cores but better than the samples stored in Zeist. Pore water from neighbouring Paleogene sands is easier to access than clay pore water and can be used as a first approximation for trace elements in clay pore water at disposal depth. This research is funded by the Ministry of Infrastructure Water Management and mainly performed by TNO.

In EURAD-2 Work Package (WP) Radionuclide mobility under perturbed conditions (RAMPEC)(2024-2029), the impact of perturbations on the distribution values between the solid and liquid phase in argillaceous, crystalline and cementitious systems are being studied. From the Dutch side, COVRA and NRG will work on the database for the retention/transport parameters. NRG also develops macroscopic models. The NRG activities in RAMPEC are co-funded through a programme other than COPERA.

10.2.3.14 Quantification of thicknesses and depths - priority 1

In CORA, the distribution, depths and thicknesses of Paleogene clays were determined (Simmelink et al., 1996). These distribution and thicknesses have not yet been included in DGM nor REGIS (models from Dinoloket) for every location in the Netherlands. The suitability of a multi-level GDF for larger regions of the Dutch territory would be easier evaluated if this information would be made publicly available through the Dinoloket website.

10.2.4 Underground engineering structure

10.2.4.1 Scalability to larger waste inventories - priority 2

A multilevel GDF in which waste is disposed of according to their hazard potential is proposed in this safety case. This multi-level GDF is expected to reduce the footprint of the GDF compared to a single level GDF. A multi-level GDF can therefore more easily be adapted for larger waste inventories than a single level GDF. Each waste family may have its own optimal disposal depth, which will help define the required thickness and clay content of potentially useable formations.

10.2.4.2 Different disposal depths - priority 2

The Dutch Paleogene clays are envisaged to be poorly indurated clays, with convergence or creep rates that will require the use of supported tunnels for safe transport and emplacement of waste packages. An engineered tunnel liner with a suitable thickness and strength will be installed as excavation proceeds. The required thickness and strength of the tunnel liner and the spacing between disposal tunnels both depend on the geotechnical properties of the clay host rock. These properties are expected to depend on the depth of the clay formation. These properties are being measured in SECUUR.

Engineering and operational optimisation strategies are being investigated in the EURAD-2 WP HLW Repository optimisation including closure (OPTI). The topics covered include design of buffer, backfill, support structures, emplacement techniques, and overall optimization approaches for waste disposal facilities and their management in radioactive waste management programmes and safety cases. Delft University of Technology will be leading knowledge management in this project and their work is fully funded by the European Commission. COVRA contributes to the development of mutual understanding between research entities, technical support organisations, waste management organisations and civil society organisations.

10.2.4.3 Backfill - priority 4

Cementitious backfill and concrete with the same content of aggregates as the concrete buffer have been studied experimentally in EURAD-1 MAGIC (2021-2024) (Vidal et al., 2024) and some experimental and modelling work has been done in EURAD ACED (2019-2014) (Blanc et al., 2024). The conclusion of MAGIC is that microbial activity is inevitable when working with concrete specimens. Initially, it was thought that this microbial activity is limited to the surfaces of the concrete specimens.

However, microbial analysis performed by the Belgian research entity SCK CEN on COVRA's backfill samples showed that microbial films are also present within the backfill samples, especially those made with CEM III/B. This result implies that all chemo-mechanical evolution results available in the literature on cement paste and grouts may be microbially induced since the chemical alteration rate due to microbial activity is larger than pure chemical alteration rate.

Backfill samples have been made with CEM I and CEM III/B, buffer-like samples were made with CEM III/A and CEM III/B. Experiments to measure the ingress of magnesium and other elements involved in degradation (Mladenovic et al., 2024) are on-going. Nevertheless, understanding of the leaching processes that increase the porosity and decrease the strength of concrete has improved. Leaching can be minimized with a proper type of cement that is used to manufacture concrete. Furthermore, a set of CEM I and CEM III/B backfill and buffer-like specimens remains exposed to saline water at COVRA's premises in order to have some representative concrete specimens that can be examined at a later date. The experimental testing of alternative cement formulations will therefore not be continued during the next 5 years.

10.2.5 Waste package design

10.2.5.1 Consumption, generation and transport of water - priority 2

The engineered barrier system is protected by the host rock and surrounding formations from dynamic natural processes out into the far future, even allowing for impacts of future developments, including climate change. Investigations in the framework of EURAD-1 ACED (Mladenovic et al., 2024) and comparison with Boom Clay in Belgium, showed that the water saturated permeability of waste package concrete can be lower that the permeability of the host clay. Consequently, water movement in waste package concrete can be more restricted than has been assumed in earlier studies. This recent elucidation has important implications for potential radionuclide release rates from the waste forms, since all alteration processes of the engineered barriers require the presence of water, and most of these processes consume water. Estimation of water consumption rates, representative for the geometry of the EBS, as performed for this safety case, will be continued.

10.2.5.2 HLW containment - priority 2

The current safety case uses the Belgian supercontainer concept for HLW, in which a carbon steel overpack (encapsulating the HLW canister) is surrounded by a concrete buffer.

A stainless steel envelope surrounding the concrete buffer was considered as an option in OPERA. However, this stainless-steel envelope would interface a porous cementitious backfill, in which microbial activity cannot be excluded. Assuming a stainless steel corrosion rate representative for disposal would then lead to such high gas generation rates that the clay host rock would be perturbed in the post-closure phase of the GDF (Levasseur et al., 2021). The stainless-steel envelope was therefore omitted from the design of the supercontainer for heat generating for HLW in COPERA.

A preliminary thermo-mechanical calculation shows that the heat induced stresses for encapsulated vitrified HLW might be too high to assure prevention of cracks in the concrete buffer in the operational phase of the GDF (Neeft et al., 2021). The tensile strength of steel is much larger than concrete. If a stainless-steel envelope is ruled out for corrosion and gas production reasons, HLW containment with a reinforced concrete buffer can be investigated. Reinforced steel can be positioned near the outer diameter of the concrete buffer in order to prevent cracking of concrete.

No credit is taken for the strength of concrete in the post-closure evolution, the concrete buffer only provides a high pH to limit corrosion of the overpack for tens of thousands of years.
The strength of concrete is highly determined by the distribution in size of pores and the presence of cracks. So far, no measurable reduction in strength has been observed in COVRA's concrete samples after almost 8 years exposure to a solution as saline as seawater (Vidal et al., 2024).

10.2.6 Waste form

10.2.6.1 Long-term behaviour at disposal scale - priority 2

In contact with pore water, vitrified HLW slowly alters into clay minerals and zeolites, consuming some of the water. The vitrified waste form is not the only waste form that consumes water as it is altered. All metals consume water in the anaerobic corrosion process, which also generates hydrogen. Some metals, such as Zircaloy, pick up most of the generated hydrogen (Sakuragi, 2017) due to the formation of metal-hydrides. Many other metals, such as steel and aluminium, release hydrogen during the corrosion process. As in the case of glass leaching experiments, alteration rates measured on metals immersed in solutions should be replaced by alteration rates representative for the chosen geometry of the engineered barrier system and its ability to transport water. Any solid phases present in these barriers that might ion exchange with dissolved constituents of pore water and thus enhance the alteration rate should be identified. Water is also consumed by the degradation processes of organic matter that lead to the formation of CO_2 e.g. Mladenovic et al. (2019).

Another characteristic of an engineered barrier that can enhance the alteration rate is the distribution in pore sizes, since this determines whether microbial activity might enhance the alteration rate. This is particularly important for organic wastes, which provide sufficient nutrient sources for microbes.

The potential generation rates of gases by the alteration of all waste forms are not yet estimated with sufficient accuracy, but they can have a high impact on the performance of the multibarrier system. These rates should, therefore, be determined for each waste family, based on the relevant disposal configuration.

10.2.6.2 Depleted uranium as aggregates - priority 2

The volume of depleted uranium waste is large, but its contribution to the overall radioactivity in the GDF is very small. If it were to be used as an aggregate in the HLW supercontainer buffer, a smaller thickness of the buffer would be required to provide the same gamma radiation protection as that provided by the use of siliceous or calcareous aggregates. Neutron shielding calculations need to be performed in order to determine whether the reduced thickness of the concrete buffer then provides sufficient shielding in the operational phase of the GDF.

Considering depleted uranium wastes as aggregates to manufacture floors in the GDF is another alternative use of depleted uranium. These floors can be in the transport tunnel and in the disposal tunnels since no gases are expected to be produced when water comes into contact with depleted uranium.

These alternatives for disposing of depleted uranium greatly reduce the footprint and costs of the GDF, since the number of disposal tunnels and length of transport tunnels can be reduced, and Konrad containers for depleted uranium are no longer needed.
Implementation of these alternatives would require knowledge on the strength and water permeability of the depleted uranium granules currently stored at COVRA's premises. External funding for COVRA is foreseen to characterise these wastes over the next decade.

10.2.6.3 Not yet stored waste forms - priority 2

COVRA currently processes solid waste through compaction and manufacturing concrete. The containment of the radionuclides in 200 l drums is through COVRA's waste package concrete.
This waste form may change if COVRA would built a plasma furnace. The expected waste forms arising from this type of processing are glass and spent ion exchange resins. Such types of waste forms are already stored at COVRA's premises and these waste forms can be disposed of in a GDF in clay host rocks. The volumes of waste will be larger if the processing of solid waste remains compaction and containment of the radionuclides by concrete.

The waste package concrete in the 200 litre LILW drums has a sufficiently high strength and low permeability to allow their use for the floor to be made in the transport tunnels. Because a deep backfill is required under the floor of the transport tunnels in order to provide floors of sufficient width, the floor structures will occupy large volumes of the GDF. Using the LILW drums as a component of the transport tunnel floor structures gives considerable spatial economy and, at the same time provides sufficient separation from the HLW in adjacent disposal tunnels. The waste package concrete in the drums encapsulates various compacted waste forms and the possibility of using these drums in the transport tunnels will depend on the potential generation rates of gases by the alteration of these waste forms.

10.3 Planning of activities for a GDF in the Netherlands during the next decade

In the previous section, scientific, technical research activities have been identified. The production of safety cases is coordinated by COVRA with the updating of the national programme every 10 years, but site-specific safety cases are foreseen only after 2050 (Verhoef et al., 2020). The next safety case will be produced in 2034 (see Figure 10-3) and therefore is also foreseen to be a conditional, non-site-specific safety case.

Some prioritization has been made in the list of key topics in section 10.2. The execution of the proposed research in these key topics should allow better estimates of the ranges of parameter values to be used in the post-closure safety assessment and the design of the geological disposal facility.

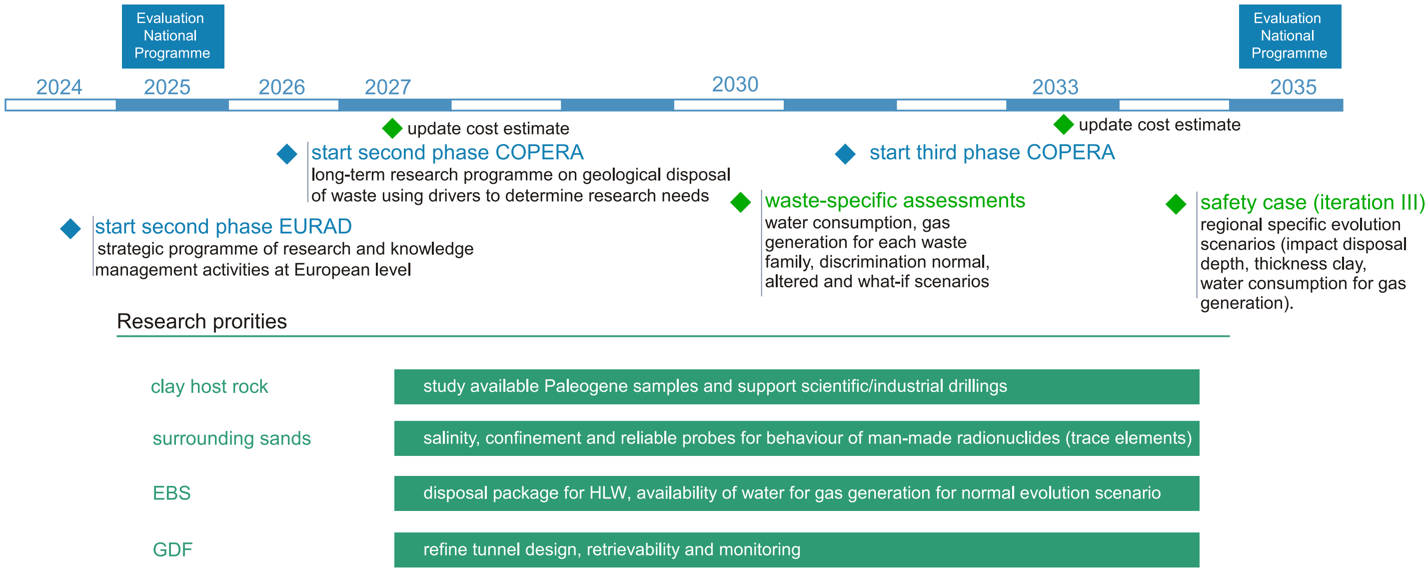

Figure 10-3: Planning of milestones (in green) and research programmes (in blue) for the next decade for a GDF in clay host rock.

The geotechnical studies on available Paleogene clay samples are being performed in the framework of SECUUR(2023-2028) leading to two PhD theses in 2027. These studies are used to refine the design of the GDF. This refinement is expected to have an impact on the milestone: update of the cost estimate, since the disposal concept is expected to be updated. The geochemical studies on available Paleogene samples and surrounding sands are expected to have an impact on the safety assessment. A full safety assessment is foreseen for the next update in 2034, but the contribution of the health-related effects of each waste family is studied in assessments that are made continuously and are foreseen to be published in 2030.

10.3.1 Update cost estimate

The disposal concept and cost estimate are planned to be updated in 2027. The current cost estimate for a GDF in clay host rock is based on price levels of 2022; it was performed with SSK and has been reviewed externally (Tempels et al., 2023). Some optimization has been performed in COPERA, especially by the consideration of different disposal depths for different types of waste.
The uncertainty in the volumes of the different types of waste is currently not taken into consideration in the cost estimate and this needs to be included in the next update.

Work in EURAD-2 OPTI (2024-2026) is expected to provide further details allowing refinement of the tunnel design for the next update in 2027. Further valuable sources of information that may lead to design optimization include EURIDICE, the Belgian Waste Management Organisation (ONDRAF/NIRAS), the International Tunnelling and Underground Space Association (ITA-AITES) and the IAEA network of Underground Research Facilities, in which COVRA is currently allowed to participate.

10.3.2 Waste-specific assessments

The present study looks only at selected issues that may lead to changes in the earlier OPERA safety case for a single level GDF at 500 m depth in Boom Clay. The post-closure safety assessment of the multilevel design of the GDF introduced in this safety case needs to be fully assessed. Each waste family may have its own optimal disposal depth, which will help define the required thickness and clay content of potentially useable formations. Waste-specific assessments may give the constraints in the required geological settings for each waste family.

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March 2022 deep drilling for extraction of Paleogene Clay cores courtesy TU Delft

APPENDIX 1: DEVELOPMENT OF DUTCH POLICY ON RADIOACTIVE WASTE DISPOSAL

COVRA plans and executes the national radioactive waste management policy set by the Dutch government. This policy can set requirements for the safety of geological disposal of waste. COVRA needs to take into account these requirements in its concept for construction, operation and closure of a disposal facility. The figure below shows the key dates in the Dutch policy and the projects that have been undertaken to implement the resulting strategy.

LLW Disposal Pre-1982

A decision to join the international programme for disposal of radioactive waste in the Atlantic Ocean was taken in 1965 and, from 1967 until 1982, the Netherlands deposited packages with solidified Low Level Waste at specific locations in the north-east Atlantic Ocean under strict requirements (NEA, 1985; Rastogi and Sjoeblom, 1999). However, a Committee to re-evaluate the acceptability of this operation was appointed in 1981 (Lambers-Haquebard, 1981) and in 1983, the Dutch government decided to stop off shore deposition of waste packages (Alders et al., 1983; Winsemius, 1983). This led to the decision to look for land-based options for managing all of the radioactive wastes in the Netherlands. As an initial step, an interim storage facility for radioactive waste was foreseen and COVRA was established to manage this interim facility (Winsemius, 1982). The first research programme for disposal of High Level Waste (HLW) from reprocessing of spent nuclear power fuel in rock salt formations was coordinated by the government (ICK, 1975). This HLW was envisaged to be generated with a larger nuclear programme than currently exists i.e. 3500 MWe.

Key Developments from 1984

The Dutch policy for waste management was established in 1984 (Berkers et al., 2023). The policy requires long-term (100 years) interim storage of all the country's radioactive waste, together with development of a research strategy for ultimate disposal. The following three requirements were defined for the storage of both chemotoxic and radiotoxic wastes: Isolation, Control and Surveillance/Monitor (In Dutch Isoleren, Beheersen en Controleren so called IBC-principle). The objective of the IBC-principle is to prevent unacceptable amounts of toxic materials from entering our living environment. Disposal of radioactive waste in the Netherlands was proposed to be in rock salt formations. Both LILW and HLW could be disposed of in a geological disposal facility (GDF) with boreholes from galleries or boreholes directly dug from the Earth's surface. Control and surveillance would remain feasible if facilities were to be constructed in a salt formation (Winsemius and Kappeyne van de Coppello, 1984). The first national research programme OPLA (Dutch acronym for Opberging te LAnd) considered only rock salt formations since the construction method for tunnels in clay was not yet industrialized (OPLA, 1984).

Key Developments from 1993

One of the disposal concepts studied in OPLA (OPLA, 1989, 1993) consisted of boreholes dug from Earth's surface. Emplacement of waste in boreholes was envisaged by free fall and closure of the borehole by creep of salt. This conflicted with the IBC-principle into which a requirement for retrievability of waste had been introduced in the Dutch policy (Alders, 1993). Accordingly, the second national research programme CORA (Dutch acronym for Commissie Opberging Radioactief Afval) was focussed on disposal concepts that could allow for retrieval of waste packages and this was judged to be feasible (van Geel, 2002). In addition to rock salt, it was decided that clay formations were to be studied. The widely spread Neogene and Paleogene clay formations in the Netherlands were considered, but the initial focus was laid on Boom Clay since extensive information was available for this potential clay host rock because of work in Belgium (CORA, 2001).

Key Developments from 2002

COVRA was from its inception responsible for managing the collection of the waste, the processing of the waste and the storage of the waste. For long term storage of the HLW returning from reprocessing in France, COVRA in 2003 commissioned the state-of-the-art HABOG facility which was commissioned by Queen Beatrix. All of the pre-disposal activities (collection, treatment and storage) as well as the construction, operation and closure of the GDF are funded through the waste fees, that COVRA collects (Pronk, 2002). OPLA and CORA had been coordinated by the Dutch geological Survey with COVRA being responsible for the research into disposal of waste (van Geel, 2002). Accordingly, COVRA coordinated the third national programme OPERA (Dutch acronym for OnderzoeksProgramma Eindberging Radioactief Afval). This programme focused on Boom Clay, the specific poorly indurated clay studied in Belgium. Experience in the Netherlands concerning this clay formation was relatively sparse in comparison to the situation with the (Zechstein) rock salt formation that was extensively studied in OPLA. Accordingly, the focus was on Boom clay in OPERA and this allowed significant transfer of knowledge from Belgium to the Netherlands (Verhoef and Schröder, 2011).

Key Developments from 2011

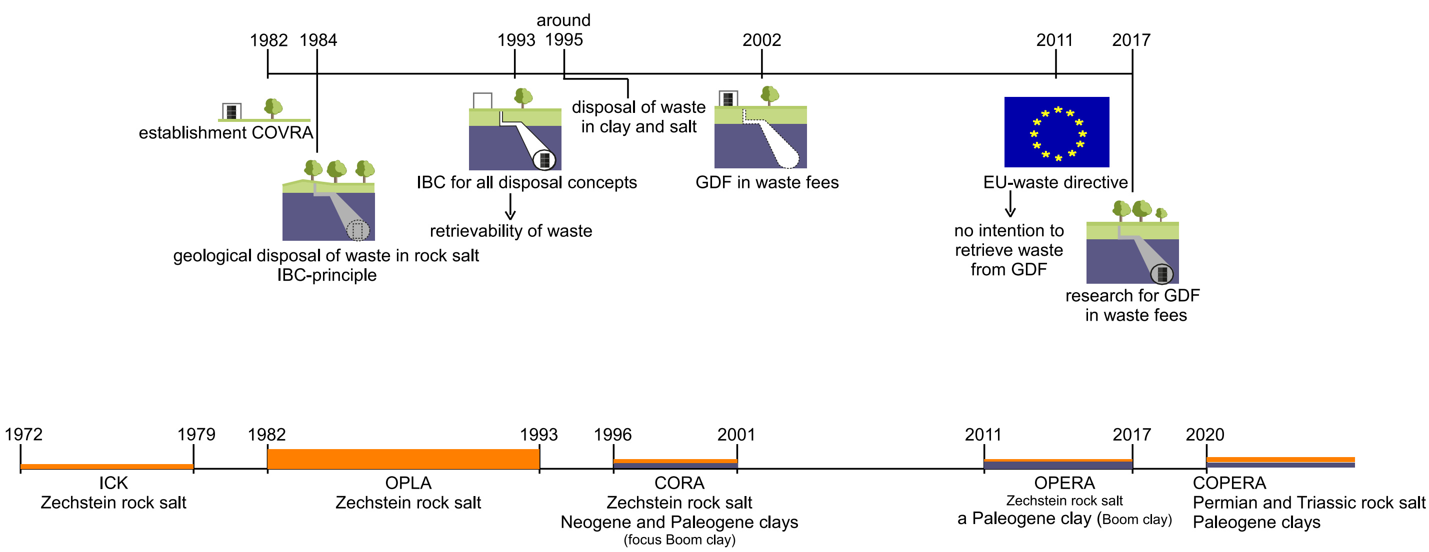
The European Council directive for the responsible and safe management of spent fuel and radioactive waste in the European Union came into force in 2011 (EC, 2011). This directive obliges all Member States to define a national programme for the management of radioactive waste and to document this at given intervals. In order to harmonize Dutch law with the European requirements concerning disposal of waste, The Netherlands needed to revise its definition of disposal of waste to confirm there is no intention to retrieve the waste (Kamp and Teeven, 2013). OPERA produced the initial Dutch safety case for disposal of radioactive waste (Verhoef et al., 2017) and this outcome was presented to the Dutch parliament. Unlike after the programmes, OPLA and CORA, the presentation of OPERA was not followed by a scheduled discussion in the Parliament. Instead, the Netherlands have chosen to structure the discussion about disposal of waste through the national programme in the framework of the council directive for safe management of spent fuel and radioactive waste in the EU (van Veldhoven-van der Meer, 2018).

Key Developments from 2017

Since the publication of the OPERA Safety case in 2017, disposal planning at COVRA has proceeded in parallel on two national geological options - one in clay, as documented in the present report and another in salt, where a dedicated report is also being produced. COVRA also keeps open a further option, namely disposal in a multinational facility, as is studied by the ERDO Association which is domiciled in the Netherlands. Since 2018, a continuous financial resource base for research and development for the management of radioactive waste has become available through the waste fees. The following three drivers are used to prioritize the research for a geological disposal facility in the Netherlands (Verhoef et al., 2020; Verhoef et al., 2017):

- Confidence in the post-closure safety provided by the disposal system of engineered and natural barriers;
- Disposability of the different types of radioactive waste;
- Better understand and optimise costs for geological disposal facility.

A Dutch acronym has been established for COVRA's on-going research programme into geological disposal of waste: COVRA's OnderzoeksProgramma Voor Eindberging van Radioactief Afval (COPERA).



Key dates in policy relevant for geological disposal of waste (top): A more detailed image of the handling of radioactive waste in the Netherlands can be found in Berkers et al. (2023). Dutch research programmes (ICK, OPLA, CORA, OEPRA and COPERA (bottom).

APPENDIX 2: Clay-related publications to which COPERA (2020-2025) contributed

Examples of inputs for COVRA's long-term research programme is the available funding research capacity and infrastructure. Examples of outputs of this programme are safety cases, cost estimates of GDFs, scientific publications and reports (Verhoef et al., 2020). As described in section 1.5.7, COVRA's long-term research programme consists of 7 Work Packages (WPs). Several reports have been published for the tasks described in COVRA's long-term research programme for the period 2020-2025. This Appendix provides a short description of these tasks and WPs related to a GDF in clay host rocks and the publications categorized in these WPs. These reports are published on COVRA's website but there are also reports made with COVRA's (financial) contribution in the Horizon 2020 project EURAD. The publications related to a geological disposal facility in clay are described in this Appendix.

Work package 0: Programme management and monitoring

This work package includes international collaboration. COVRA participates in the NEA Clay Club that examines argillaceous rocks considered for geological disposal of waste. These rocks range from soft, poorly indurated clays to harder, indurated clays. Preliminary site characterisation activities to dispose of waste in Opalinus Clay finished in 2022. Clay cores from this indurated clay have been investigated. The experimental results from these cores and the associated interpretation has been published in a special issue in the scientific journal: Applied Geochemistry. COVRA has contributed to the contextualisation of the performed research and performed the editorial handling of some papers in this special issue.

A. Bath, M. De Craen, E.A.C. Neeft, Preface: Special Issue of Applied Geochemistry on 'Transport parameters, natural tracer profiles and porewater chemistry derived from the Swiss deep drilling programme in a clay-rich sedimentary sequence, Applied Geochemistry (2023) 105851.

Work package 1: Programme strategy

This work package includes cost estimates. A cost estimate for a GDF in Paleogene clays was been made in 2023 by COVRA and reviewed by BouWQ. The reviewer comments are addressed in an updated version of the cost estimate.

E. Tempels, S.E. de Nijs, J. Rooijakkers, J., (2023). *Review kosten-begroting - Nucleaire eindberging COVRA, review of the cost estimate,* only available in Dutch.

Neeft et al., *Costs for a disposal facility in clay host rock - price level 2022,* to be publishedat the same time as this safety case, the updated cost estimate.

Work Package 2: Safety case and integration

The integration of the knowledge obtained through COPERA (2020-2025) for disposal of waste in clay host rock is presented in the present Safety Case and Feasibility report.

Work Package 3: Engineered Barrier System

The understanding of the transport of water and gas in the materials used for waste packaging and backfill have been experimentally verified. The waste forms aluminium and glass have also been investigated.

- Task 3.1: Spent research reactor fuel

The criticality of SRRF in COPERA has been calculated with the detailed geometry of the waste as stored in a canister.

SRRF has a high surface area of aluminium. Filling the void volume with a fluid that hardens after pouring into a material with strength could be a solution. The exact cementitious recipe to be poured in the canisters still needs to be to investigated but the execution of this task by Koets et al., provides a high suitable answer to the question how the SRRF needs to be conditioned for safe conditioning of the waste and remaining subcritical in the post-closure phase.

K. Koets, D. Erkan, A. Meiksane, A. Mantzanas, J. Bregman, *Safe Spent Research Reactor Fuel Disposal,* Report from Conceptual Design Project at Delft University of Technology, www.covra.nl.

Later in 2022, a shortage in lithium nitrate ($LiNO_3$) seemed in an international exchange meeting between European WMOs not present. $LiNO_3$ was found to be sold at a good price and that availability facilitates the definition in the cementitious recipe to be poured between the spent fuel assemblies to condition SRRF for safe disposal.

- Task 3.2: EBS for poorly indurated clay
    - Task 3.2.1: Waste package for HLW

In OPERA, the waste package design for HLW was adopted from the Belgian programme. The safety concept is that there can only be contact between pore water and the waste form when the clay host rock is no longer heated by the waste. This waste package consists of a steel overpack that is surrounded by a concrete buffer. The buffer provides the beneficial conditions to limit corrosion of the overpack for a sufficient long period in the post-closure phase.

    - Task 3.2.2: Vitrified HLW

The European Project Assessment of Chemical Evolution of ILW and HLW Disposal Cells (ACED) aims to clarify which geochemical processes need to be included for representative assessments of the chemical evolution of HLW and ILW disposal cells. COVRA participates in ACED in order to identify which geochemical processes need to be included for the chemical evolution of disposal cell containing vitrified HLW. This identification is published in three ACED reports. This chemical evolution has been used to identify the degradation mechanism and potential radionuclide release mechanism from the vitrified waste form and substantiation of the chosen degradation rate. These activities contribute to the understanding of the expected behaviour of a geological disposal system with poorly indurated clay host rock and assessment basis for the post-closure safety assessment. The range in flux of water from the poorly indurated clay host rock into the EBS is between granitic and indurated clay rock. The information in ACED has also been published by the natural analogue working group.

E. Neeft, E. Weetjens, A. Vokal, M. Leivo, B. Cochepin, C. Martin, I. Munier, G. Deissmann, V. Montoya, P. Poskas, D. Grigaliuniene, A. Narkuniene, E. García, J. Samper, L. Montenegro, A. Mon (2019). *Treatment of chemical evolution in National Programmes, Deliverable 2.4 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593,* www.ejp-eurad.eu.

E. Neeft, D. Jacques, G. Deissmann (2022). *Initial State of the Art on the assessment of the chemical evolution of ILW and HLW disposal cells. Deliverable 2.1 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593,* www.ejp-eurad.eu.

D. Jacques, E. Neeft, G. Deissmann (2024). *Update of the State of the Art on the assessment of the chemical evolution of ILW and HLW disposal cells. Deliverable 2.2 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593,* www.ejp-eurad.eu.

E. Neeft, G. Deissmann; D. Jacques, *The use of analogue information in assessing the chemical evolution of HLW disposal cells.* Proceedings of the NAWG-17 Workshop 8-11th May, 2023, Zadar, Croatia.

- Task 3.2.3: Waste package for LILW

Disposal cells containing cemented ILW are also considered in ACED. COVRA had its waste package concrete for the 200 litre drums investigated in ACED and to seek which geochemical processes need to be included for the expected behaviour of a disposal cell containing cemented ILW. Experimental work to extract parameters for cemented ILW and modelling work has been published in four reports:

Experimental work:
A. Mladenovic, E. Neeft, G. Deissmann, R. Dähn, G. Geng, G. Koskowski, and L. Markku (2019). *ILW: Report describing the selected experiments and the existing/expected experimental results. Final version as of 24.12.2019 of Deliverable 2.11 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593,* www.ejp-eurad.eu.

Mladenovic, A., Markku, L., Neeft, E., Dähn, R., and Kosakowski, G., (2024). *ILW: Report describing the results of characterisation performed during the project. Final version as of 18.03.2024 of Deliverable 2.13 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593.*

Oxidation and carbonation profiles and permeabilities for COVRA's waste package for the 200 litre drums and foamed concrete considered as a backfill have been published in Deliverable 2.13. Only the oxidation profiles for COVRA's waste package concrete are shown in Chapter 6.

Modelling work
J. Govaerts, D. Jacques, J. Samper, E. Neeft, V. Montaya (2022). *Model abstraction methods for upscaling and integration of process knowledge in reactive transport models for geological disposal of radioactive waste. Final version as of 10.01.2022 of deliverable D2.18 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593,* www.ejp-eurad.eu.

Blanc, P., Goaverts, J., Gu, Y., Jacques, D., Kosakowski, G., Leivo, M., Marty, N.C.M., Neeft, E., Shao, H., and Vehling, F., (2024). *Description of ILW modelling results and recommendations for future experiments and numerical work:* Deliverable 2.15.

Modelled oxidation and carbonation profiles and diffusion values for COVRA's waste package and foamed concrete (a type of grout) have been published in Deliverable 2.15 and are to be used for the assessment basis for the post-closure safety assessment.

- Task 3.2.4: Closure of GDF

A stepwise closure is foreseen in which disposal tunnels are filled with a grout after emplacement of waste packages in order to avoid circulation of water around the waste packages in the unlikely case of flooding during the operational phase. The European Project Chemo-Mechanical AGIng of Cementitious materials (MAGIC) was granted funding after finalisation of COVRA's programme of work 2020-2025. This project allowed an unique opportunity to look at grouts. Experimental work to investigate the strength of grouts as a function of the exposure time to clay pore water representative for the Dutch case, has been published in two reports. So far, after more than 6 to 8 years exposure, only larger strengths have been measured than the strength as measured after 28 days hardening. As an increase in strength of concrete is highly correlated to a decrease in parameter values (diffusivity, permeability) for the transport of water, the parameter values as found in ACED can be considered upper bound values.

Some sizes of the pores in the grouts allow microbial activity. Abundant microbial activity was measured in the exposing solutions and surfaces of grout specimens. A part of the samples were gamma radiated to distinguish the pure chemo-mechanical evolution from the microbial induced chemo-mechanical evolution. These samples were sterilized with a dose rate of 6.4 kGy per hour for 9 hours i.e. a dose of 57.6 kGy. This dose rate is about 1000 times larger than the dose rate of the concrete buffer interfacing the carbon steel overpack. The impact of such high radiation doses on the strength of grout as well buffer-representative concrete was measured to be negligible. The expected behaviour of concrete can therefore assumed to be independent of the radiation of the waste.

E. Neeft, T. de Bruin, R. van Kleef, Q T. Phung, J. Perko, S. Seetharam, K. Mijnendonckx, X. Li, L. Hausmannová, R. Vašíček, P. Večerník, V. Hlaváčková, K. Černá, T. Černoušek, O. Helson, X. Bourbon, J. Zghondi, T. Vidal, A. Sellier, J. Shao, T. Rougelot, K. Jantschik, J. Kulenkampff, G. Deissmann, M. Griffa, S. Churakov, T. Gimmi, B. Ma (2021): *Selected experiments for assessing the evolution of concrete, their mechanical safety function and performance targets. Final version as of 28.01.2022 of Deliverable D 16.3 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593,* www.ejp-eurad.eu.

T. Vidal, T. Rougelot, K. Jantschik, M. Middelhoff, J. Kulenkampff, L. Hausmannová, R. Vasicek, M. Griffa, B. Ma, S. Churakov, T. Phung, E. Neeft, G. Deissmann, K. Mijnendonckx, O. Helson, P. Večerník, T. Černoušek, J. Němeček (2024). *Technical report on the results of the selected experiments for assessing the evolution of the studied types of concrete. Final version as of 30.05.2024 of Deliverable 16.4 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593,* www.ejp-eurad.eu.

COVRA also contributed to the state of the art of the chemo-mechanical evolution of cementitious materials in MAGIC:

A. Dauzères, O. Helson, S. Churakov, V. Montoya, J. Zghondi, E. Neeft, J. Shao, A. Cherkouk, K. Mijnendonckx, A. Sellier, G. Deissmann, T. Arnold , L. Lacarrière, M. Griffa, T. Vidal, M. Neji, X. Bourbon, L. Ibrahim, N. Seigneur, S. Poyet, B. Bary, Y. Linard, T. Le Duc, V. Hlavackova, A. Pasteau, K. Jantschik, M. Middelhoff, J. Perko, Q. Tri Phung, S. Seetharam, W. Shan, A. Singh, J. Lloyd, V. Vilarrasa (2022). *Initial State of the Art on the chemo-mechanical evolution of cementitious materials in disposal conditions. Final version as of 09/11/2022 of deliverable D16.1 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593,* www.ejp-eurad.eu

A. Dauzères, O. Helson, S. Churakov, A. Sellier, E. Neeft, K. Mijnendonckx, A. Cherkouk, G. Deissmann (2024). *Deliverable 16.2: Updated State-of-the-art report & Summary of major conclusions from the MAGIC work package.*

Work package 4A: Poorly indurated clays

The clay host rock forms the main barrier in the disposal concept. Improving knowledge on how it performs and evolves is critical to understand and quantify its ability to contain radionuclides over long times. The clay host rock has been given priority 1 in COVRA's programme of work (2020-2025) to confirm the main assumptions underpinning the safety concepts and feasibility of a GDF in poorly indurated clays.

- Task 4A.1 : Geotechnical properties

COVRA funded part of the drilling costs for the 'Delft Aardwarmte Project' (DAP) since suitable fresh cores of poorly indurated clay can be taken during the making of geothermal wells according to the two reports by Abels and Vardon (2020) and Munsterman (2020). The drilling took place in March 2022 and the drilling report has also been published on COVRA's website.

D.K. Munsterman (2020). *The palynological results of the Paleogene and Neogene successions in wells PNA-GT-01 and PNA-GT-04, TNO R11381 report,* www.covra.nl

H.A. Abels, P.J. Vardon, (2020). *Early Cenozoic stratigraphy in the Vrijenban syncline - compilation of current information,* www.covra.nl.

P.J. Vardon, H.A. Abels, A. Barnhoorn, S. Beernink, A.-C. Dieudonné, G. Drijkoningen, S. Beernink, S. Laumann, T. Schmiedel, J. van den Berg, L. Vargas Meleza, (2022). *Drilling report Delftse Hout multipurpose research borehole DAPGEO-02, TU-Delft report,* www.covra.nl.

- Task 4A.2 Diffusion dominated transport

Because of the low permeability of clays, water movements are slow, and transport of radionuclides is expected to take place predominantly by diffusion. There is however yet insufficient evidence for assuming diffusion dominated transport for poorly indurated clay in the Netherlands. One of the easiest elements to assess diffusion within clay is chlorine (Mazurek et al., 2011). With the available knowledge, poorly indurated clays at disposal depth such as Boom Clay are expected to be present in confined saline sandy formations ((Griffioen, 2015; Griffioen et al., 2016) which may make the assessment of diffusion by chlorine difficult.

A poorly indurated (fluvial) clay layer separates fresh water from brackish water in the envisaged DAPWELL project. Predictions of chemical measured profiles at Delftse Hout but also in Rotterdam and Rijswijk has not provided evidence that solely diffusion in poorly indurated clays in the Netherlands is allowed to be assumed for geological disposal. A very small advection rate needed to be included in order to match the measured profile with the modelled one. The poorly (fluvial) clay layers to which access of samples was achieved could be too thin and not continuous enough.
The performed work has also clearly shown that the sampling method is a key issue in the measured chlorine profile.

B.T.M. van Esser (2022). *Transport of Ions Through Clays of the Peize and Waalre Formation, Master of Science thesis at Delft University of Technology,* www.covra.nl

- Task 4A.2.1: Gas

In some cases, diffusion for the transport of radionuclides in a clay host rock may no longer be assumed when the gas generated by corrosion of metals in the waste and waste package cannot be sufficiently dissipated by diffusion. Transport of gas in clays is assessed in the currently running EURAD Work Package GAS.
The conceptualisation of the transport of gas in Dutch disposal cells has been drafted by COVRA and included in the State of the Art. COVRA co-funds TU-Delft's contribution to model the stage when diffusion can no longer be assumed.

S. Levasseur, F. Collin, K. Daniels, M. Dymitrowska, J. Harrington, E. Jacops, O. Kolditz, P. Marschall, S. Norris, X. Sillen, J. Talandier, L. Truche and J. Wendling (2021). *Initial State of the Art on Gas Transport in Clayey Materials. Deliverable D6.1 of the HORIZON 2020 project EURAD, Work Package Gas. EC Grant agreement no: 847593,* www.ejp-eurad.eu.

J. Liaudat, A.-C. Dieudonné and P.J. Vardon (2023). *Modelling gas fracturing in saturated clay samples using triple-node-zero-thickness interface elements,* Computer and Geotechnics **154** (2023) 105128.

- Task 4A.3: Retardation

Retardation of radionuclides is expected to take place by sorption on clay minerals and by precipitation of solubility limited elements. Retardation is, among others, dependent on the elemental speci-ation of radionuclides. The Dutch Geological Survey TNO participates in the currently running EURAD Work Package Fundamental on understanding of radionuclide retention (FUTuRE) in order to increase the understanding of the retention mechanisms of redox sensitive elements e.g. U, Pu, Tc, Np and Se in iron bearing minerals. COVRA co-funds TNO's contribution.

N. Maes, M. Glaus, B. Baeyens, M. Marques Fernandes, S. Churakov, R. Dähn, S. Grangeon, C. Tournassat, H. Geckeis, L. Charlet, F. Brandt, J. Poonoosamy, A. Hoving, V. Havlova, C. Fischer, A. Scheinost, U. Noseck, S. Britz, M. Siitari-Kauppi, T. Missana (2021). *State-of-the-Art report on the understanding of radionuclide retention and transport in clay and crystalline rocks. Final version as of 30.04.2021 of deliverable D5.1 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593,* www.ejp-eurad.eu.

Y. Qian, A.C. Scheinost, S. Grangeon, J.M. Greneche, A. Hoving, E. Bourhis, N. Maubec, S.V. Churakov, M.M. Fernandes (2023). *Oxidation State and Structure of Fe in Nontronite: From Oxidizing to Reducing Conditions.* ACS Earth and Space Chemistry, 7(10), 1868-1881.

S. Grangeon, M. Marques, U. Alonso de los Ríos, L. Charlet, S. Churakov, M. C. Bucur, R. Dagnelie, A. María Fernandez, H. Geckeis, J. Griffioen, M. García Gutiérrez, A. Hoving, F. Javier Leon, T. Missana, A. Oliveira, M. Olteanu, A. Poulain, Y. Qian, J.-C. Robinet, S. Savoye, B. Schacherl, A. Scheinost, C. Tournassat, T. Vitova (2024). *Final technical report on redox reactivity of radionuclides on mineral surfaces. Final version as of 08.02.2024 of deliverable D5.7 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593.*

N. Maes, M. Glaus, B. Baeyens, M. Marques Fernandes, S. Churakov, R. Dähn, S. Grangeon, C. Tournassat, H. Geckeis, L. Charlet, F. Brandt, J. Poonoosamy, A. Hoving, V. Havlova, A. Scheinost, C. Fischer, U. Noseck, S. Britz, M. Siitari-Kauppi, X. Li, O. Fabritius, T. Missana (2024). *State-of-the-Art report on the understanding of radionuclide retention and transport in clay and crystalline rocks. Final version as of 23.02.2024 of deliverable D5.2 of the HORIZON 2020 project EURAD. EC Grant agreement no: 847593.*

- Task 7.2: Knowledge transfer to students

The knowledge transfer to students is an integral part of the research programme. For Task 4A.2, an internship was funded by COVRA and also lead to a Master's thesis (van Esser, 2022). For Task 3.2.1, 5 bachelor students worked on a research project for spent research reactor fuel (Koets et al., 2022); this project was sponsored by COVRA. For Task 4.A.1., COVRA is co-funding 2 PhD students that work in the SECUUR project. For Task 4.A.3, COVRA is financing a post-doc to investigated trace elements in clay pore water from clay cores extracted at about 400 meters depth in Delft.

The book Geology of the Netherlands is frequently used by students. Inclusion of geological disposal of radioactive waste in this book would be an effective long-term method to make the next generation familiar with this topic. COVRA wrote together with the Dutch Geological Survey a chapter for the update of this book:

E.A.C. Neeft, J. Bartol. M.R. Vuorio, G.-J. Vis (to be published in 2025). Chapter 21: Geological disposal of radioactive waste.

APPENDIX 3: Requirements in the RMS

Level 1 requirements

Level 1 requirements are applicable to all the steps in the management of waste. These requirements have been selected from documents prepared by international and national organisations that must be taken account of in COVRA's activities. Preferably, reference is made to documents from national organisations.

L1-DCRE-01: The permitted additional radiation dose for radiological workers in the Netherlands is 20 mSv per year.

Any doses shall be as low as reasonably achievable (ALARA). Some quantification is necessary for the definitions of design requirements. The Netherlands adopts the recommendations provided by the International Commission of Radiation Protection (ICRP) in the Dutch Decree on radiation protection. ICRP has recommended the dose limit of 20 mSv per year. Article 7.34 in this Decree requires that the dose rate shall not exceed 20 mSv per year.

L1-NPRA-01: The disposal facility shall be operational in 2130.

The first national programme states that: *The definitive decision on the disposal method will be taken around 2100. At that time, society may opt for a different (end-point) management option, depending on the state of understanding at that time, and assuming that other alternatives are available by that time. The relatively long period of above ground storage will provide time to learn from experiences in other countries, to carry out research and to accumulate knowledge. In this way, sufficient money can also be set aside to make eventual disposal possible. As a consequence, in the future, a well-argued decision on the management of radioactive waste can be taken, without unreasonable burdens being placed on future generations* (I&E, 2016).

In Article 10.10 in the Dutch radiation protection decree of 2023, it is required that COVRA's waste fees need to be set in a transparent, objective and non-discriminatory manner. Any contribution to the waste fees includes the provision by COVRA of collection, transport, processing, storage and disposal services. The financial scheme determines the contributions in the waste fees for the long-term storage of the waste in the surface facilities and the emplacement of waste packages in an underground facility in 2130.

L1-NPRA-02: Waste shall be isolated from people and the accessible biosphere.

The Dutch policy for waste was established in 1984 (Berkers et al., 2023; I&E, 2016; Winsemius and Kappeyne van de Coppello, 1984) with the following three objectives for the management of chemically toxic waste and radiologically toxic waste: Isolation, Control and Surveillance/Monitor (In Dutch Isoleren, Beheersen en Controleren so called IBC-principle). The implementation of the IBC-principle reduces the possibility of having unacceptable amounts of toxic materials in our living environment. This principle is, for example, also applied to soil remediation (I&E, 2016).

L1-NPRA-03: Any handling of the waste shall be controlled.

Control is also an objective for the management of waste in the Dutch IBC-principle (I&E, 2016; Winsemius and Kappeyne van de Coppello, 1984)..

L1-NPRA-04: Waste shall be enclosed by a series of engineered barriers.

People will be protected by placing a series of barriers between the radioactive waste and the human environment. The packaging of the waste is an engineered barrier that ensures that the waste is enclosed (I&E, 2016). The description for enclosure of waste has only been written for disposal of waste in the national programme. For other steps in the management of radioactive waste, we have looked at the meaning of the IBC principle for chemically toxic waste and radiologically toxic waste as published upon parliamentary questions submitted by Boois (VROM, 1985). The term Isolation is used for two meanings: 1) to prevent direct contact with the waste as well as 2) to prevent contaminants from the waste spreading into the soil by the use of an impermeable layer.
The second use meant by COVRA as containment.

Level 2 requirements

Level 2 requirements are extracted from the national and international requirements, but have been specifically developed into a form that expresses by COVRA's policy. These level 2 requirements are also applicable to all the steps in the management of waste.

L2-COV-01: The incremental radiation dose for radiological workers shall be less than 6 mSv per year.

A lower dose constraint than the Dutch radiation protection decree is used by the organisation (COVRA, 2022).

L2-COV-02: Waste shall be stored in dedicated surface facilities until an end-point management technique is available.

The surface facilities should provide the optimal conditions to keep the waste form and packing in a suitable condition for an end-point management technique. Three examples of end-point management techniques are:

1. Disposal of waste;
2. Recycling of materials used as a waste form or for packaging of the waste;
3. Treating as a conventional waste stream for sufficiently decayed (exempt) waste whose materials cannot be recycled.

L2-COV-03: Simple, robust, reliable and proven techniques shall be used.

From packaging to disposal, wastes might require handling over a period of more than 100 years. The procedures for handling the waste need to be demonstrated and effective. The use of technologies that have been proven will therefore be used for all the steps in the management of waste. These demonstrated techniques also minimize the uncertainty in cost estimates for the operation and closure of the disposal facility.

L2-COV-04: Materials for which broad experience and knowledge already exists, shall be used.

From storage to disposal, engineered barriers need to be stable over a period of more than 100 years There are many processes that may affect the behaviour of barriers used for waste packaging during storage and disposal. Uncertainty in long-term prediction of the behaviour of the engineered barrier system is smaller if demonstrated materials are used. During storage, the behaviour of the waste package can be monitored, and if needed, any deleterious behaviour that is not predicted can be mitigated by active measures. After disposal of waste, safety cannot rely on monitoring, owing to the long-time scales foreseen. The identification of the relevant processes for the prediction of the long-term behaviour of engineered barriers can be performed by studying natural analogues and archaeological analogues.

L2-COV-05: Only solidified waste shall be stored and disposed of.

All waste is solidified in advance of storage. The properties of the solid waste form minimize any release of radionuclides under all circumstances considered for storage (e.g., flooding) and disposal of waste.

L2-COV-06: In the case of fissile material, the containment shall exclude criticality.

Fissile material contained in a package lacks water to moderate a possible chain reaction. Following Posiva (2021), COVRA assumes that the possibility of criticality can be excluded by containment.

Level 3 requirements

The level 3 requirements described here are only applicable to disposal of waste.

L3-NPRA-01 A disposal facility shall be designed to contain all the different types of radioactive waste expected to arise up to 2130.

The policy in the Netherlands is that most of the radioactive waste produced in the Netherlands is managed by a single organisation: COVRA. The exceptions are radioactive waste with a half-life less than 100 days that is allowed to decay at the sites where it is generated and large amounts of NORM waste that are disposed of (or re-used) at two designated landfills. COVRA therefore collects radioactive waste from different waste generators, for example, nuclear power plants, nuclear medicine production plants, research organisations, universities and hospitals. The waste generators transfer different types of waste to COVRA according to their classification based on activity, radionuclide content and chemical and physical characteristics. Low and Intermediate Level Waste is generated by all organisations that consign wastes to COVRA. High Level Waste is only generated by nuclear plants.

L3-IAEA-01: Isolation shall be provided for at least several thousands of years for HLW.

This requirement is a short abbreviation of requirement 9 in SSR-5 (IAEA, 2011a): *The disposal facility shall be sited, designed and operated to provide features that are aimed at isolation of the radioactive waste from people and from the accessible biosphere.*
The features shall aim to provide isolation for several hundreds of years for short lived waste and at least several thousand years for intermediate and high level waste. In so doing, consideration shall be given to both the natural evolution of the multibarrier system and events causing disturbance of the facility.

COVRA makes a distinction between heat-generating HLW and non-heat generating HLW. Non-heat generating HLW is classed as intermediate level waste by the IAEA. Dutch HLW has a contact dose rate larger than 10 mSv per year. LILW as currently stored by COVRA has a contact dose rate smaller than 10 mSv per year. This type of waste is called low level waste in many countries. COVRA does not have a category of short-lived waste for disposal, as this type of waste may be recycled.

L3-NPRA-02: Waste shall be retrievable during the operational phase of the GDF through until its closure.

The requirement for the retrievability of waste has been introduced in Dutch policy in order to have active control over the emplacement of waste packages and closure of the disposal facility (Alders, 1993). This requirement is to prevent the investigation of disposal concepts that do not comply with the IBC-principle.

L3-IAEA-02: The radionuclides in the waste shall be contained by the engineered barriers and natural barriers until radioactive decay has significantly reduced the hazard posed by the waste.

This requirement is split in two requirements (L3-IAEA-02 and L3-IAEA-04). These requirements are short descriptions for requirement 8 in SSR-5 (IAEA, 2011a): The engineered barriers, including the waste form and packaging, shall be designed, and the host environment shall be selected, so as to provide containment of the radionuclides associated with the waste. Containment shall be provided until radioactive decay has significantly reduced the hazard posed by the waste. In addition, in the case of heat generating waste, containment shall be provided while the waste is still producing heat energy in amounts that could adversely affect the performance of the disposal system.

L3-IAEA-03: Passive safety shall be provided by multiple safety functions for containment and isolation.

We combined the following three requirements in SSR-5 related to passive safety and multiple safety functions. L3-IAIA-03 is a short abbreviation of:

- requirement 5: *Passive means for the safety of the disposal facility: The implementer shall evaluate the site and shall design, construct, operate and close the disposal facility in such a way that safety is ensured by passive means to the fullest extent possible and the need for actions to be taken after closure of the facility is minimized.*
- requirement 6: *Understanding of a disposal facility and confidence in safety: The implementer of a disposal facility shall develop an adequate understanding of the features of the facility and its host environment and of the factors that influence its*

safety after closure over suitably long time periods, so that a sufficient level of confidence in safety can be achieved.

- requirement 7: *Multiple safety functions: The host environment shall be selected, the engineered barriers of the disposal facility shall be designed and the facility shall be operated to ensure that safety is provided by means of multiple safety functions. Containment and isolation of the waste shall be provided by means of a number of physical barriers of the disposal system. The performance of these physical barriers shall be achieved by means of diverse physical and chemical processes together with various operational controls. The capability of the individual barriers and controls together with that of the overall disposal system to perform as assumed in the safety case shall be demonstrated. The overall performance of the disposal system shall not be unduly dependent on a single safety function.*

Requirement 6 in SSR-5 has overlaps to a great extent with L2-COV-04.

As explained in Chapter 2, the multi-barrier system addresses two principal objectives providing safety: isolation of the waste and containment of the radionuclides associated with them. These objectives should be provided by nature i.e. passive safety. Each of the barriers (components) in the multi-barrier system has one or multiple safety functions. The safety concept in the conceptualisation stage is the description of how the barriers in the disposal concept are integrated to provide safety after closure. Safety functions with assigned time frames are used for this description. A safety function is the action or role that a natural and/or engineered barrier performs after closure of the GDF to prevent radionuclides in the waste from ever posing an unacceptable hazard to people or the environment. The necessary engineered barrier system (EBS) can be host rock specific.

L3-D-IAEA-04: In the case of heat-generating waste: the engineered containment shall retain its integrity until the produced heat will no longer adversely affect the performance of the multibarrier system.

See description for L3-IAEA-02.

APPENDIX 4: Waste scenarios

Three waste scenarios as described in section 4.1 are:

1. Operation of Borssele Nuclear Power Plant (NPP) until 2033 and replacement of the High Flux Reactor (HFR) by Pallas for the production of medical isotopes;
2. Borssele NPP operation until 2043,i.e. the operational time is extended by 10 years;
3. Waste scenario 2 with additional wastes from two new nuclear power plants with each a capacity three times higher than the Borssele NPP

The figure below the calculated volume for HLW for all nuclear plans at three different years 2030, 2050 and 2130 (Burggraaff et al., 2022).

The characteristics of the waste arising from the replacement of High Flux Reactor (Pallas, in orange) are assumed to be the same as the current SRRF. Medical isotopes may also be produced differently than neutron irradiation of from uranium targets. The waste characteristics are however unknown to COVRA and therefore the volume of waste envisaged by Shine (in dark blue) is excluded from the considered the three waste scenarios.

The characteristics of the waste arising from 10 years extension of the Borssele Nuclear Power Plant (NPP) (+10, in light blue) (waste scenario 2) and two additional NPPs (in green, waste scenario 3) are also assumed to be similar to the Borssele NPP i.e. recycling of spent nuclear power fuel in France. The waste products are vitrified HLW (CSD-v), compacted hulls and end (CSD-c) and spent ion exchange resins (1000 litre concrete containers).

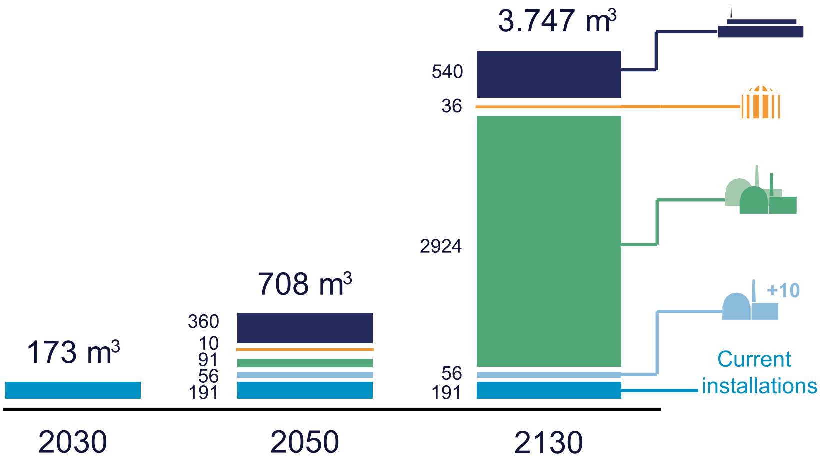
Waste scenario 2 results into an increase of HLW (vitrified HLW and CSD-c) but only a minor increase of processed LILW (1000 litre concrete containers). Only a minor adjustment is needed in order to incorporate the increase in waste volume: increasing the number of disposal tunnels for CSD-v or increasing the length of the disposal tunnels.

Waste scenario 3 results into a major increase of HLW (vitrified HLW and CSD-c) since the assumed electric power per nuclear power plant of 1600 MW with an operational time of 80 years is much more than the Borssele plant of 485 MW with an operational time of 60 in waste scenario 1 (70 years in case of 10 years extension in waste scenario 2). The dismantling waste is also assumed to increase with the power of the plant by which its volume is more than 4 times larger than in waste scenario 1 and 2.

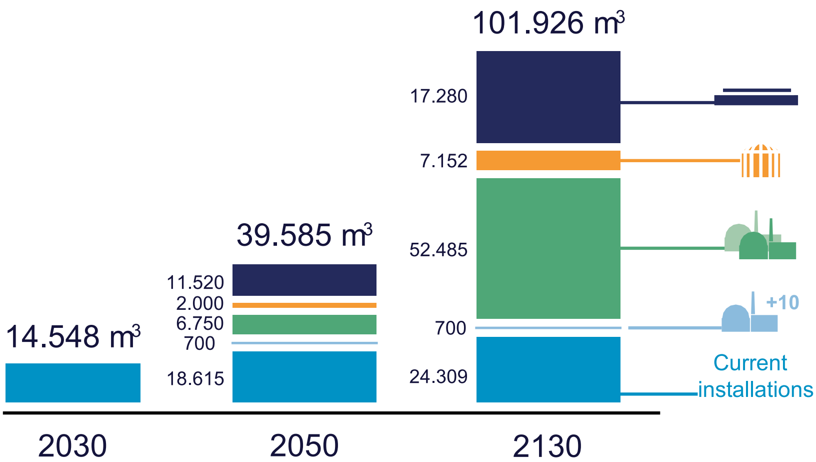
The volume of packaged vitrified HLW becomes larger than the volume of packaged depleted uranium in waste scenario 3. HLW disposal packages are currently foreseen to have a concrete buffer. The use of depleted uranium as aggregates in cementitious materials as hypothesized in COVRA's research programme for disposal packages for HLW (Verhoef et al., 2020) would eliminate a large proportion of the volume of waste to be disposed of. What needs to be known are:

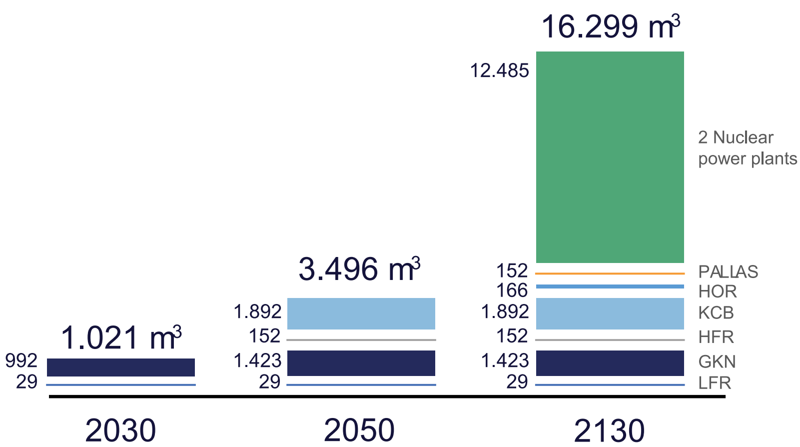
- Technical details such as the strength and permeability of the depleted uranium granules;
- Political acceptance: the use of depleted uranium granules as aggregates in concrete makes the retrievability of depleted uranium as granules more difficult.

The volume of the floor in the GDF already approaches the volume of depleted uranium to disposed of in waste scenario 1. The cost estimate would reduce significantly if it is politically accepted to use waste for the manufacturing of components in the GDF and technical details are known. The same accounts for the 200 litre drums except that its use in the GDF may be limited in the transport tunnels and shafts due to the poor knowledge in waste characteristics. It is already known that the waste package concrete has a minimum in compressive strength of 45 MPa after 28 days hardening but its strength after more than 100 years storage is yet unknown.



Expected volume of HLW for all nuclear plans (Burggraaff et al., 2022)





Expected volume of processed LILW (top) and dismantling waste (bottom) for all nuclear plans (Burggraaff et al., 2022)

Waste Category	In storage volume as defined in Burggraaff et al. (2022)		Packaged for disposal		
	Volume [m³]	Number of canisters / containers	Number of packages	Volume [m³]	Weight per package [tonne]
Spent research reactor fuel	49	244	244	1840	20
Vitrified HLW (vHLW)	111	618	618	4854	22
Compacted hulls & ends (Non heat generating HLW)	121	670	96	4210	20
Dismantling waste (LILW)	3814	-	826	3814	Max 20
TE-NORM (LILW)	49360	-	12600	58070	Max 20
Processed LILW	32161	108500	108500	For three 200 litre drums: max 2.25 tonne For one 1000 litre concrete container: max per 3 tonne	

Expected inventory of wastes for disposal in 2130 for waste scenario 2 showing their mass and volume in storage and their mass and volume when packaged for disposal. The dimensions of packages as described in section 4.1

Waste Category		Packages for disposal			Disposal tunnels		
		$\text{Length}^{L}/\text{Height}^{H}$ (m)	Diameter (m) or Width (m) × Length (m)	N packages	N	$Ø_{out}/Ø_{in}$ (m)	Length (m)
Spent research reactor fuel (HLW)		2.4	2.0	244	6	5.0 / 4.0	100
Vitrified HLW		2.5	2.0	478	16 or 12	5.0 / 4.0	100 or 133
Compacted hulls & ends (HLW)		2.0	2.0	96	1	5.0 / 4.0	200
Dismantling waste (LILW)		1.7	1.6 x 1.7	826	4	5.0 / 4.0 *(5.0 / 4.6)*	185 *(175)*
TE-NORM (LILW)		1.7	1.6 x 1.7	12600	27	5.0 / 4.0	400
Processed LILW	200 l	0.88	0.59	100000	*(20)*	*(5.0 / 4.6)*	*(200)*
	1000 l	1.25	1.00	8500	*(8)*	*(5.0 / 4.6)*	*(205)*

Number and dimensions for disposal at one disposal depth and in italics and brackets different from a variety of disposal depths for waste scenario 2

Waste Category	In storage volume as defined in Burggraaff et al. (2022)		Packaged for disposal		
	Volume [m³]	Number of canisters / containers	Number of packages	Volume [m³]	Weight per package [tonne]
Spent research reactor fuel	49	244	244	1840	20
Vitrified HLW	1425	7918	7918	62188	22
Compacted hulls & ends (Non heat generating HLW)	1689	9382	1340	8419	20
Dismantling waste (LILW)	16299	-	3528	16299	Max 20
TE-NORM (LILW)	49360	-	12600	58070	Max 20
Processed LILW	84646	314100	314100	For three 200 litre drums: max 2.25 tonne For one 1000 litre concrete container: max per 3 tonne	

Expected inventory of wastes for disposal in 2130 for waste scenario 3 showing their mass and volume in storage and their mass and volume when packaged for disposal. The dimensions of the packages as described in section 4.1.

APPENDIX 5: Distribution values between the clay host rock and clay pore water

Cations and cation-complexes can by contained by the slightly negatively charged clay mineral surfaces and immobile dissolved organic matter. This containment can be defined by ratio between the amount of an element that is uptake by the solid (e.g. clay minerals) and the amount that is left in the water. This ratio is called a K_d value e.g. Baeyens et al. (1982). During OPERA, coefficients have become available that are supported by experiments in Boom Clay in Mol (Bruggeman and Maes, 2017). These K_d values are usually larger than the values obtained with the model used in OPERA (Schröder et al., 2017a) to calculate K_d values.

The diffusion accessible porosity for each cation or cation-complex is treated separately from the K_d value in OPERA. Table 5-2 shows these porosities (ε) for all species. The retardation factor (R) is then determined by:

$$R = \frac{\rho}{\varepsilon} K_d + 1$$

where ρ is the density of the solid e.g. clay host rock.

The diffusion value is divided by the retardation factor in order to obtain the retarded diffusion value by sorption. Examples of diffusion values can be obtained from the pore diffusion coefficient and diffusion accessible porosity in Table 5-2.

K_d-values highly depend on the ionic strength and competing cations (Baeyens et al., 1982; Helfferich, 1962). In COPERA, the clay host rock is considered a system with established K_d values for millions of years. These K_d values can be obtained from a proper characterisation of the behaviour of traces of elements such as radioactive potassium, uranium and thorium. Other trace elements are non-radioactive radioisotopes of radionuclides in the waste form. Elemental concentrations in seawater have been used due to lack in these concentrations in clay pore water for the K_d values derived with the elemental concentrations in a Paleogene clay (Boom Clay) with an average clay content of 41.5 wt%.

	(Schröder et al., 2017a)		(Bruggeman and Maes, 2017)		COPERA
	K_d [L/kg] calculated for Mol		K_d [L/kg]		K_d [L/kg]
Element	Median	Range	Best estimate	Range	Average (41.5 wt% clay minerals)
Am	134	52-414	6’500	3’200-32’000	-
Cs	605	183-1’038	9’600	600-18’600	21’000
Cm	100	40-253	6’500	3’200-32’000	-
Eu	109	44-282	6’500	3’200-32’000	> 50’000
Np	100	40-253	6’500	3’200-32’000	
Pu	100	40-253	6’500	3’200-32’000	-
Sn	100	40-253	6’500	3’200-32’000	
Sr	97	40-236	320	180-800	Correlated with inorganic carbon content, not clay
Tc	100	40-253	6’500	3’200-32’000	
Th	100	40-253	6’500	3’200-32’000	> 50’000
U	93	27-243	6’500	3’200-32’000	900 but better correlated with organic carbon content

APPENDIX 6: Activity per waste container 130 years after collecting waste

Activity per ECN canister with 33 elements of spent Highly Enriched Uranium (HEU) research reactor fuel and Low Enriched Uranium (LEU) research reactor fuel (comments/source values as explained in (Verhoef et al., 2016)

Radionuclide	Spent High Enriched Uranium Fuel		Spent Low Enriched Uranium Fuel	
	Activity [Bq]	comments/source value	Activity [Bq]	comments/source value
Ac-226				
Ac-227				
Ag-108 m				
Am-241	1.03E+12	[Dodd,2000]	1.38E+13	[NRG,2012]
Am-242m				
Am-243	2.18E+09	[Dodd,2000]	3.54E+10	EOI [NRG,2012]
Ba-133				
Be-10				
Bi-207				
Bi-214				
C-14	6.57E+07	[Dodd,2000]	6.57E+07	same as HEU
Ca-41				
Cd-113 m				not calculated
Cf-249				
Cf-251				
Cf-252				
Cl-36	0.00E+00	[Dodd,2000]		same as HEU
Cm-241				
Cm-243	1.27E+09	100 × compacted waste	1.27E+10	1,000 × compacted waste
Cm-244	1.94E+09	one -tenth LEU	1.94E+10	[NRG,2012]
Cm-245	1.48E+07	[Dodd,2000]	1.48E+08	ten times HEU
Cm-246	1.85E+06	[Dodd,2000]	1.85E+07	ten times HEU
Cm-247	2.63E+03	100 × compacted waste	2.63E+04	1,000 × compacted waste
Cm-248	1.62E+04	100 × compacted waste	1.62E+05	1,000 compacted waste
Co-60				
Cs-135	1.35E+09	[Dodd,2000]	2.10E+10	Cs-137 (one month), 6.62%
Cs-137	4.39E+13	[Dodd,2000]	6.53E+13	[NRG,2012]
Eu-152				

Radionuclide	Spent High Enriched Uranium Fuel		Spent Low Enriched Uranium Fuel	
	Activity [Bq]	comments/source value	Activity [Bq]	comments/source value
Eu-152 m				
H-3				
Ho-166m				
I-129	2.04E+08	[Dodd,2000]	2.78E+08	Cs-137 (one month), 0.706%
K-40				
Kr-81				
Kr-85	1.19E+11	Cs-137 (one month), 1.310%	3.63E+10	[NRG,2012]
Mo-93				
Mo-99				
Nb-93 m				
Nb-94	1.34E+06	[Dodd,2000]	1.34E+06	same as HEU
Ni-59				
Ni-63	7.06E+03	[Dodd,2000]	7.06E+03	same as HEU
Np-237	2.76E+09	[Dodd,2000]	4.24E+09	EOI [NRG,2012]
Pa-231	1.41E+06	[Dodd,2000]	1.41E+06	same as HEU
Pa-233				
Pa-234				
Pb-202				
Pb-210				
Pb-214				
Pd-107	1.04E+08	[Dodd,2000]	1.36E+08	Cs-137 (one month), 0.1393%
Pm-145		not calculated		not calculated
Po-209				
Pu-238	4.95E+12	[Dodd,2000]	8.25E+12	[NRG,2012]
Pu-239	9.11E+10	[Dodd,2000]	1.56E+12	[NRG,2012]
Pu-240	6.83E+10	[Dodd,2000]	1.47E+12	[NRG,2012]
Pu-241	7.00E+10	[Dodd,2000]	9.44E+11	[NRG,2012]
Pu-242	3.37E+08	[Dodd,2000]	5.29E+09	EOI [NRG,2012]
Pu-244	4.91E+06	100,000 × compacted waste	4.91E+07	1,000,000 × compacted waste
Ra-226	8.78E+05	[Dodd,2000]	8.78E+05	same as HEU
Re-186m				
Se-79	2.97E+09	[Dodd,2000]	9.46E+08	Cs-137 (one month), 0.0487%
Si-32				not calculated
Sm-146				

Radionuclide	Spent High Enriched Uranium Fuel		Spent Low Enriched Uranium Fuel	
	Activity [Bq]	comments/source value	Activity [Bq]	comments/source value
Sm-151	6.73E+11	[Dodd,2000]	1.09E+13	Cs-137 (one month), 0.4204%
Sn-121m				
Sn-126	2.57E+09	[Dodd,2000]	1.90E+11	Cs-137 (one month), 0.0594%
Sr-90	3.80E+13	[Dodd,2000]	5.31E+13	[NRG,2012]
Tc-99	1.11E+11	[Dodd,2000]	1.11E+11	same as HEU
Tc-99 m				
Th-229	1.31E+04	[Dodd,2000]	1.31E+04	same as HEU
Th-230	3.22E+07	[Dodd,2000]	3.22E+07	same as HEU
Th-231				
Th-234				
Ti-44				
U-232	2.95E+10	100,000 × compacted waste	2.95E+11	1,000,000 × compacted waste
U-233	1.83E+06	[Dodd,2000]	1.83E+06	same as HEU
U-234	2.81E+10	[Dodd,2000]	4.29E+10	[NRG,2012]
U-235	4.72E+08	[Dodd,2000]	5.32E+08	EOI [NRG,2012]
U-236	3.80E+09	[Dodd,2000]	6.79E+09	EOI [NRG,2012]
U-238	1.01E+07	[Dodd,2000]	9.39E+08	EOI [NRG,2012]
U-239				
Zr-93	1.67E+10	[Dodd,2000]	1.67E+10	same as HEU

Activity per ECN canister with uranium collection filters after 130 years cooling (comments/source values as explained in (Verhoef et al., 2016)

Radionuclide	Uranium collection filters	
	Activity [Bq]	comments/source value
Ac-226		
Ac-227		
Ag-108 m		
Am-241	8.52E+08	[NRG,2009]+Pu-241
Am-242m		
Am-243	2.18E+09	same as HEU
Ba-133		
Be-10		
Bi-207		
Bi-214		
C-14		secondary waste stream
Ca-41		
Cd-113 m		
Cf-249		
Cf-251		
Cf-252		
Cl-36		secondary waste stream
Cm-241		
Cm-243	1.27E+09	same as HEU
Cm-244	1.94E+09	same as HEU
Cm-245	1.48E+07	same as HEU
Cm-246	1.85E+06	same as HEU
Cm-247	2.63E+03	same as HEU
Cm-248	1.62E+04	same as HEU
Co-60		
Cs-135		LILW: molybdenum waste I
Cs-137		LILW: molybdenum waste I
Eu-152	8.72E+04	[NRG,2009]
Eu-152 m		
H-3		secondary waste stream
Ho-166m	3.56E+04	[NRG,2009]
I-129	2.06E+00	from Te-129m filter and LILW: molybdenum waste I

Radionuclide	Uranium collection filters	
	Activity [Bq]	comments/source value
K-40		
Kr-85		secondary waste stream
Mo-93		
Mo-99		
Nb-93 m		
Nb-94		not calculated
Ni-59		
Ni-63	1.05E+06	[NRG,2009]
Np-237	1.47E+07	[NRG,2009]
Pa-231	5.54E+04	[NRG,2009]
Pa-233		
Pa-234		
Pb-202		
Pb-210		
Pb-214		
Pd-107	3.10E+07	[NRG,2009]
Pm-145	4.59E+01	[NRG,2009]
Po-209		
Pu-238	2.43E+08	[NRG,2009]
Pu-239	2.08E+10	[NRG,2009]
Pu-240	1.12E+09	[NRG,2009]
Pu-241	4.90E+07	[NRG,2009]
Pu-242	1.03E+04	[NRG,2009]
Pu-244		
Ra-226		
Re-186m		
Se-79	4.65E+07	[NRG,2009]
Si-32		
Sm-146	1.11E+00	[NRG,2009] from Pm-146
Sm-151	7.49E+11	[NRG,2009]
Sn-121m		
Sn-126	3.03E+08	[NRG,2009]
Sr-90	4.27E+12	[NRG,2009]
Tc-99	1.48E+10	[NRG,2009]

Radionuclide	Uranium collection filters	
	Activity [Bq]	comments/source value
Tc-99 m		
Th-229	1.31E+04	same as HEU
Th-230	3.22E+07	same as HEU
Th-231		
Th-234		
Ti-44		
U-232	2.84E+03	[NRG,2009] + Pu-236
U-233	1.87E+04	[NRG,2009]
U-234	9.62E+06	[NRG,2009]
U-235	1.27E+09	[NRG,2009]
U-236	4.59E+08	[NRG,2009]
U-238	2.65E+07	[NRG,2009]
U-239		
Zr-93	1.95E+09	[NRG,2009]

Activity per COGEMA canister from reprocessing spent nuclear power fuel (comments/source values as explained in (Verhoef et al., 2016)

Radionuclide	CSD-V		CSD-C		
	Activity [Bq]	comments/source value	Activity [Bq]	source value	comments
Ac-226					
Ac-227	6.07E+02	10,000 × compacted waste	< 1 Bq	max batch 24 containers	surface
Ag-108 m			1.63E+03	max batch 24 containers	neutron cap
Am-241	1.06E+14	typical [AREVA, 2007]	5.33E+10	max batch 24 containers	surface
Am-242m	1.56E+12	10,000 × compacted waste	1.56E+08	max batch 24 containers	surface
Am-243	2.57E+12	typical [AREVA, 2007]	5.93E+08	max batch 24 containers	surface
Ba-133					
Be-10					
Bi-207					
Bi-214					
C-14		secondary waste stream	1.38E+10	typical [AREVA, 2001]	neutron cap
Ca-41			2.95E+06	max batch 24 containers	neutron cap
Cd-113 m					
Cf-249			3.27E+03	max batch 24 containers	surface
Cf-251			< 1 Bq	max batch 24 containers	surface
Cf-252					
Cl-36		secondary waste stream	6.31E-04	max batch 24 containers	neutron cap
Cm-241					
Cm-243	1.27E+11	10,000 × compacted waste	1.27E+07	max batch 24 containers	surface
Cm-244	2.21E+12	typical [AREVA, 2007]	1.38E+10	max guaranteed [AREVA, 2001]	surface
Cm-245	2.90E+09	max batch 28 containers	1.09E+07	max batch 24 containers	surface
Cm-246	4.77E+10	10,000 × compacted waste	4.77E+06	max batch 24 containers	surface
Cm-247	2.63E+05	10,000 × compacted waste	2.63E+01	max batch 24 containers	surface
Cm-248	1.62E+06	10,000 × compacted waste	1.62E+02	max batch 24 containers	surface
Co-60					
Cs-135	3.01E+10	max batch 28 containers	1.04E+09	max Cs-137, 6.62%	surface
Cs-137	3.30E+14	max guaranteed [AREVA, 2007]	3.25E+12	max guaranteed [AREVA, 2001]	surface
Eu-152		not reported	3.87E+06	max batch 24 containers	neutron cap
Eu-152 m					
H-3		secondary waste stream	9.96E+09	max batch 24 containers	surface
Ho-166m					
I-129		secondary waste stream	5.30E+07	max batch 24 containers	surface
K-40					
Kr-81		secondary waste stream			

Radionuclide	CSD-V		CSD-C		
	Activity [Bq]	comments/source value	Activity [Bq]	source value	comments
Kr-85		secondary waste stream	< 1 Bq	max batch 24 containers	surface
Mo-93			5.79E+09	max batch 24 containers	neutron cap
Mo-99					
Nb-93 m					
Nb-94			5.55E+10	max batch 24 containers	neutron cap
Ni-59			3.59E+11	max batch 24 containers	neutron cap
Ni-63			1.76E+13	max batch 24 containers	neutron cap
Np-237	4.80E+10	typical [AREVA, 2007]	7.80E+06	max batch 24 containers	surface
Pa-231		not reported			
Pa-233					
Pa-234					
Pb-202					
Pb-210					
Pb-214					
Pd-107	6.78E+09	max batch 28 containers	6.74E+06	max batch 24 containers	surface
Pm-145		not reported		not reported	
Po-209					
Pu-238	4.78E+11	max weight and isotopic	1.20E+12	max act Pu-241 and isotopic	surface
Pu-239	1.44E+11	max weight and isotopic	2.14E+11	max act Pu-241 and isotopic	surface
Pu-240	2.31E+11	max weight and isotopic	3.68E+11	max act Pu-241 and isotopic	surface
Pu-241	7.35E+10	max weight and isotopic	1.41E+11	max guaranteed [AREVA,2001]	surface
Pu-242	1.01E+09	max weight and isotopic	2.09E+09	max act Pu-241 and isotopic	surface
Pu-244	4.91E+05	10,000 × compacted waste	4.91E+01	max batch 24 containers	surface
Ra-226	1.03E+02	10,000 × compacted waste	1.03E-02	max batch 24 containers	surface
Re-186m					
Se-79	2.01E+10	max batch 28 containers	5.50E+07	typical [AREVA,2001]	surface
Si-32					
Sm-146					
Sm-151	5.47E+13	max Cs-137, 0.4204%	5.38E+11	max Cs-137, 0.4204%	surface
Sn-121m					
Sn-126	3.80E+10	max batch 28 containers	8.83E+07	max batch 24 containers	surface
Sr-90	2.05E+14	max guaranteed	2.76E+12	max Cs-137,5.73%	surface
Tc-99	1.25E+12	max batch 28 containers	9.17E+09	max Cs-137,6.132%	surface
Tc-99 m					

Radionuclide	CSD-V		CSD-C		
	Activity [Bq]	comments/source value	Activity [Bq]	source value	comments
Th-229	1.17E+04	10,000 × compacted waste	1.17E+00	max batch 24 containers	surface
Th-230	1.68E+05	10,000 × compacted waste	1.68E+01	max batch 24 containers	surface
Th-231					
Th-234					
Ti-44					
U-232	2.95E+09	10,000 × compacted waste	2.95E+05	max batch 24 containers	surface
U-233	3.20E+06	10,000 × compacted waste	3.20E+02	max batch 24 containers	surface
U-234	4.77E+08	max weight and isotopic	3.06E+06	max batch 24 containers	surface
U-235	2.88E+06	max weight and isotopic	1.25E+06	max batch 24 containers	surface
U-236	4.21E+07	max weight and isotopic	1.21E+07	max batch 24 containers	surface
U-238	5.53E+07	max weight and isotopic	1.88E+07	max batch 24 containers	surface
U-239					
Zr-93	1.05E+11	max batch 28 containers	8.91E+09	max batch 24 containers	mainly neutron cap

Activity per 200 litre drum processed molybdenum waste 130 years after collecting the waste (comments/source values as explained in (Verhoef et al., 2016)

Radionuclide	Molybdenum waste stream I		Molybdenum waste stream II	
	Activity [Bq]	comments/source value	Activity [Bq]	comments/source value
Ac-226				
Ac-227				
Ag-108 m				
Am-241	1.24E+05	fraction uranium collection filters	8.59E+05	fraction uranium collection filters
Am-242m				
Am-243	3.17E+05	fraction uranium collection filters	2.20E+06	fraction uranium collection filters
Ba-133				
Be-10				
Bi-207				
Bi-214				
C-14				
Ca-41				
Cd-113 m				
Cf-249				
Cf-251				
Cf-252				
Cl-36				
Cm-241				
Cm-243	1.84E+05	fraction uranium collection filters	1.28E+06	fraction uranium collection filters
Cm-244	2.82E+05	fraction uranium collection filters	1.96E+06	fraction uranium collection filters
Cm-245	2.14E+03	fraction uranium collection filters	1.49E+04	fraction uranium collection filters
Cm-246	2.69E+02	fraction uranium collection filters	1.87E+03	fraction uranium collection filters
Cm-247	3.82E-01	fraction uranium collection filters	2.65E+00	fraction uranium collection filters
Cm-248	2.36E+00	fraction uranium collection filters	1.64E+01	fraction uranium collection filters
Co-60				
Cs-135	1.16E+07	Cs-137 (upon collection), 6.62%		
Cs-137	3.62E+10	upon collection max A2		
Eu-152	1.27E+01	fraction uranium collection filters	8.79E+01	fraction uranium collection filters
Eu-152 m				
H-3				
Ho-166m				
I-129	1.53E+05	Cs-137 (upon collection), 0.706%		
K-40				
Kr-81				

Radionuclide	Molybdenum waste stream I		Molybdenum waste stream II	
	Activity [Bq]	comments/source value	Activity [Bq]	comments/source value
Kr-85				
Mo-93				
Mo-99				
Nb-93 m				
Nb-94		not calculated		not calculated
Ni-59				
Ni-63			1.06E+03	fraction uranium collection filters
Np-237	2.13E+03	fraction uranium collection filters	1.48E+04	fraction uranium collection filters
Pa-231	8.04E+00	fraction uranium collection filters	5.58E+01	fraction uranium collection filters
Pa-233				
Pa-234				
Pb-202				
Pb-210				
Pb-214				
Pd-107	4.51E+03	fraction uranium collection filters	3.13E+04	fraction uranium collection filters
Pm-145			4.63E-02	fraction uranium collection filters
Po-209				
Pu-238	3.53E+04	fraction uranium collection filters	2.45E+05	fraction uranium collection filters
Pu-239	3.01E+06	fraction uranium collection filters	2.09E+07	fraction uranium collection filters
Pu-240	1.62E+05	fraction uranium collection filters	1.13E+06	fraction uranium collection filters
Pu-241	7.11E+03	fraction uranium collection filters	4.94E+04	fraction uranium collection filters
Pu-242	1.49E+00	fraction uranium collection filters	1.04E+01	fraction uranium collection filters
Pu-244				
Ra-226				
Re-186m				
Se-79	5.20E+05	Cs-137 (upon collection), 0.0487%	4.69E+04	fraction uranium collection filters
Si-32				
Sm-146				
Sm-151	1.09E+08	fraction uranium collection filters	7.55E+08	fraction uranium collection filters
Sn-121m				
Sn-126	1.05E+06	Cs-137 (upon collection), 0.0594	3.05E+05	fraction uranium collection filters
Sr-90	3.07E+10	Cs-137 (upon delivery), 5.73%	4.30E+09	fraction uranium collection filters
Tc-99		filtered		
Tc-99 m				
Th-229				
Th-230				

Radionuclide	Molybdenum waste stream I		Molybdenum waste stream II	
	Activity [Bq]	comments/source value	Activity [Bq]	comments/source value
Th-231				
Th-234				
Ti-44				
U-232				
U-233	2.71E+00	fraction uranium collection filters	1.88E+01	fraction uranium collection filters
U-234	1.40E+03	fraction uranium collection filters	9.70E+03	fraction uranium collection filters
U-235	1.84E+05	fraction uranium collection filters	1.28E+06	fraction uranium collection filters
U-236	6.67E+04	fraction uranium collection filters	4.63E+05	fraction uranium collection filters
U-238	3.84E+03	fraction uranium collection filters	2.67E+04	fraction uranium collection filters
U-239				
Zr-93	2.83E+05	fraction uranium collection filters	1.97E+06	fraction uranium collection filters

Activity per waste container for depleted uranium, spent ion exchange resins and compacted waste 130 years after collecting the waste (comments/ source values as explained in (Verhoef et al., 2016)

Radionuclide	Depleted uranium	Spent ion exchanger		Compacted 90 litre drums with waste	
	Activity [Bq]	Activity [Bq]	comments value	Activity [Bq]	comments / source value
Ac-226					
Ac-227				1.31E+02	[COVRA,2012]
Ag-108 m		3.26E+08	[IAEA,2002]	1.50E+03	[COVRA,2012]
Am-241				1.50E+08	[COVRA,2012]
Am-242m					
Am-243				3.65E+02	[COVRA,2012]
Ba-133				1.82E+01	[COVRA,2012]
Be-10		8.00E+04	[IAEA,2002]	1.79E+04	[COVRA,2012]
Bi-207				3.41E+02	[COVRA,2012]
Bi-214					
C-14		7.09E+09	[IAEA,2002]	1.98E+07	[COVRA,2012]
Ca-41		2.00E+06	[IAEA,2002]		not reported
Cd-113 m					reported activity upon collection after 30 years < 1 MBq
Cf-249				1.32E+02	[COVRA,2012]
Cf-251					
Cf-252					
Cl-36		4.00E+06	[IAEA,2002]	4.53E+04	[COVRA,2012]
Cm-241					not reported
Cm-243					not reported
Cm-244				1.41E+04	[COVRA,2012]
Cm-245					not reported
Cm-246					not reported
Cm-247					not reported
Cm-248				1.83E+00	not reported, deduced from Cf-252
Co-60		1.51E+04	A2 value upon collection		
Cs-135		3.00E+06	[IAEA,2002]		not reported
Cs-137		3.00E+10	A2 value upon collection	9.16E+08	[COVRA,2012]
Eu-152				2.48E+02	[COVRA,2012]
Eu-152 m					
H-3		2.04E+04	from conditioning and waste characteristics	1.08E+07	[COVRA,2012]
Ho-166m					not reported
I-129		6.00E+05	[IAEA,2002]	2.22E+03	[COVRA,2012]
K-40				4.22E+04	[COVRA,2012]

Radionuclide	Depleted uranium	Spent ion exchanger		Compacted 90 litre drums with waste	
	Activity [Bq]	Activity [Bq]	comments value	Activity [Bq]	comments / source value
Kr-81				2.84E+01	[COVRA,2012]
Kr-85				1.36E+04	[COVRA,2012]
Mo-93		3.91E+05	[IAEA,2002]		reported activity upon collection after 30 years < 1 MBq
Mo-99					
Nb-93 m				1.06E-01	[COVRA,2012]
Nb-94		4.78E+07	[IAEA,2002]	1.53E+03	[COVRA,2012]
Ni-59		4.39E+08	[IAEA,2002]	1.18E+04	[COVRA,2012]
Ni-63		2.27E+11	[IAEA,2002]	3.70E+08	[COVRA,2012]
Np-237				2.84E+02	[COVRA,2012]
Pa-231				3.97E+02	[COVRA,2012]
Pa-233					
Pa-234					
Pb-202					reported activity upon collection after 30 years < 1 MBq
Pb-210				7.71E+03	[COVRA,2012]
Pb-214					
Pd-107		6.00E+04	[IAEA,2002]		not reported
Pm-145					not reported
Po-209					reported activity upon collection after 30 years < 1 MBq
Pu-238				6.91E+07	[COVRA,2012]
Pu-239				1.26E+07	[COVRA,2012]
Pu-240				8.06E+05	[COVRA,2012]
Pu-241				3.68E+03	[COVRA,2012]
Pu-242				1.99E+06	[COVRA,2012]
Pu-244					reported activity upon collection after 30 years < 1 MBq
Ra-226				6.69E+06	[COVRA,2012]
Re-186m				2.37E+04	[COVRA,2012]
Se-79		2.40E+06	[IAEA,2002]		not reported
Si-32					reported activity upon collection after 30 years < 1 MBq
Sm-146					reported activity upon collection after 30 years < 1 MBq
Sm-151				1.65E+04	[COVRA,2012]
Sn-121m		1.98E+06	[IAEA,2002]	5.30E+02	[COVRA,2012]
Sn-126		5.40E+06	[IAEA,2002]		not reported

Radionuclide	Depleted uranium	Spent ion exchanger		Compacted 90 litre drums with waste	
	Activity [Bq]	Activity [Bq]	comments value	Activity [Bq]	comments / source value
Sr-90		6.11E+07	[IAEA,2002]	1.94E+07	[COVRA,2012]
Tc-99		6.00E+06	[IAEA,2002]	8.88E+05	[COVRA,2012]
Tc-99 m					
Th-229				2.25E+02	[COVRA,2012]
Th-230				2.84E+01	[COVRA,2012]
Th-231					
Th-234					
Ti-44					reported activity upon collection after 30 years < 1 MBq
U-232	4.68E+08			1.69E+02	[COVRA,2012]
U-233	0.00E+00			2.04E+03	[COVRA,2012]
U-234	1.73E+11			5.15E+05	[COVRA,2012]
U-235	3.46E+09			1.35E+06	[COVRA,2012]
U-236	4.10E+10			8.52E+01	[COVRA,2012]
U-238	1.50E+11			3.83E+07	[COVRA,2012]
U-239					reported activity upon collection after 30 years < 1 MBq
Zr-93		2.00E+05	[IAEA,2002]		not reported

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This report presents an overview of the results and conclusions of the Safety Case for a geological disposal facility in the Paleogene Claysof the Netherlands.
The report is a scientific/technical document that describes engineering and geological requirements needed to assure that a safe GDF can be implemented in the Netherlands.

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