

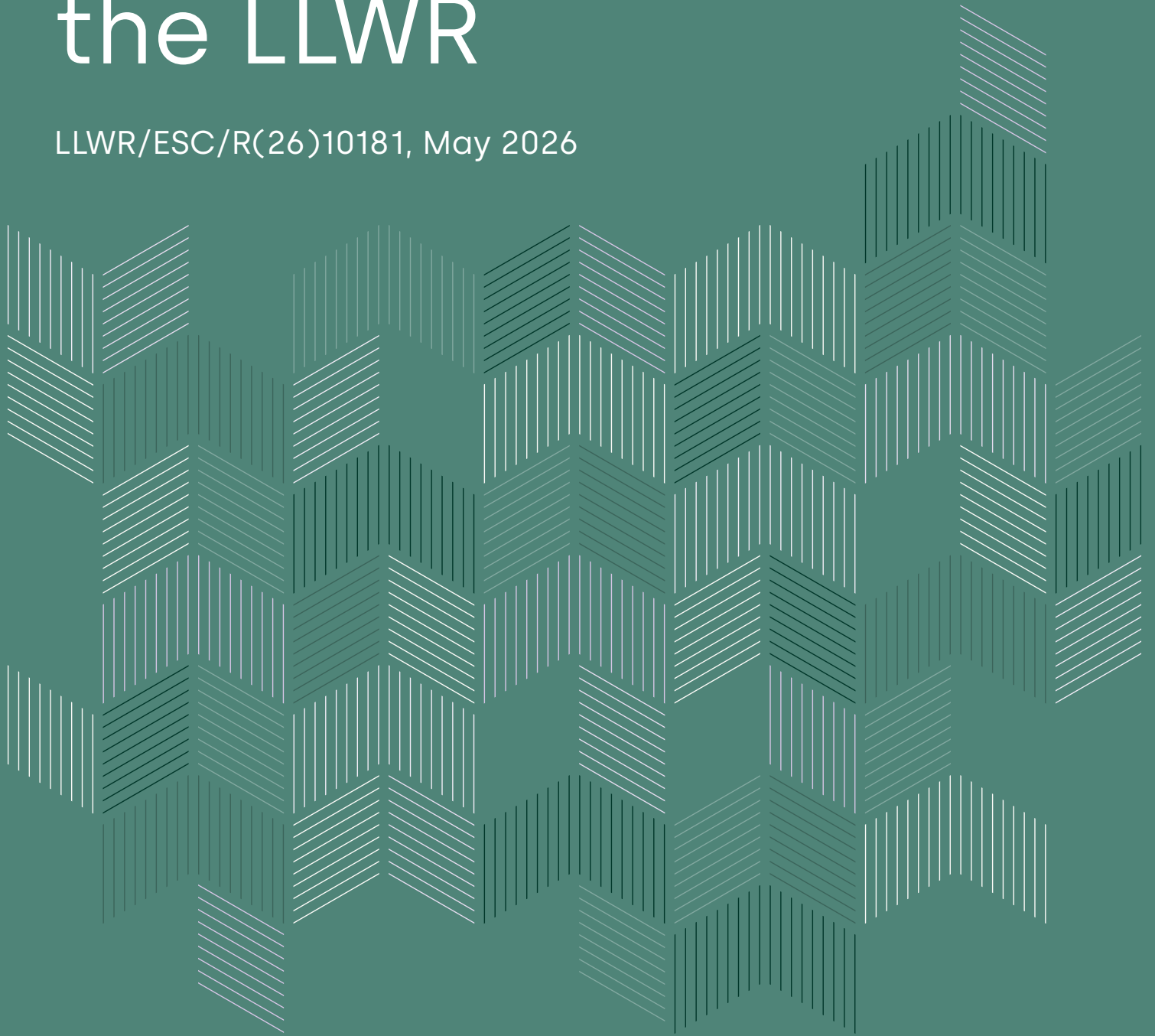


Nuclear Waste
Services

HYDROGEOLOGICAL RISK ASSESSMENT

2026 Environmental Safety Case for the LLWR

LLWR/ESC/R(26)10181, May 2026





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Preface

The Low Level Waste Repository (LLWR) is the United Kingdom's principal facility for the disposal of solid Low Level Waste (LLW). It is a near-surface disposal facility in which waste was disposed in trenches and is now being disposed in vaults excavated into the ground surface. The LLWR is owned by the Nuclear Decommissioning Authority (NDA) and operated on their behalf by a wholly-owned subsidiary division, Nuclear Waste Services Ltd.

We, Nuclear Waste Services, are committed to operating the LLWR as a safe and efficient facility that provides a continuing option for the disposal of LLW in the United Kingdom. This will be achieved consistent with good practice for the near-surface disposal of radioactive waste, in accordance with environmental, health and safety, and security regulation and guidance, and in compliance with the terms of our Nuclear Site Licence and Permit to dispose of radioactive waste. We are also committed to working with the NDA to ensure optimal use is made of the LLWR to support the NDA's mission, in accordance with government policy. This may involve the disposal of a broader range of wastes than just LLW as currently defined in the United Kingdom¹.

One of the means we use to operate the LLWR safely is to maintain and implement an Environmental Safety Case for the site. This is one of the reports presenting the 2026 Environmental Safety Case for the LLWR – the 2026 ESC. The 2026 ESC is a major update based on a comprehensive review of our previous 2011 ESC and subsequent developments. The 2026 ESC addresses both the environmental safety of the disposal facility and the rest of the site. It considers the disposal of both LLW and some less-hazardous Intermediate Level Waste (ILW). Assessing the disposal of some less-hazardous ILW does not imply any decision has been made to dispose of such waste at the LLWR. The work has been undertaken to understand the safety implications if such a decision were made and hence support consideration of the option by the NDA.

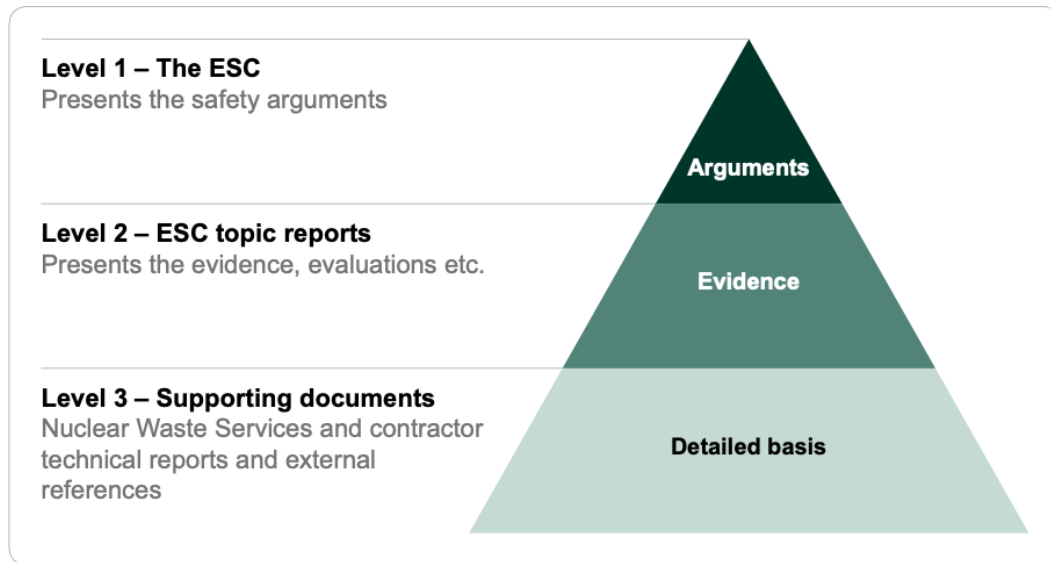
The 2026 ESC is issued under the authority of the Nuclear Waste Services' Executive Director of Sites and Operations.

The 2026 ESC consists of documents at two levels:

- A single 'Level 1' report outlines the plan for the development of the LLWR and the main arguments concerning environmental safety and how it is achieved.
- A series of 'Level 2' reports present the evidence that underpins our safety arguments, including descriptions of our management framework, system understanding, design and management choices, assessments and implementation.

¹ In government policy, LLW is defined as radioactive waste having a radioactive content not exceeding four gigabecquerels per tonne (GBq t⁻¹) of alpha or 12 GBq t⁻¹ of beta/gamma activity.

This is the Level 2 report '*Hydrogeological Risk Assessment*'. The ESC Level 1 and 2 reports are listed in the table below, which also shows for the Level 2 reports the set of arguments for which each report mainly provides evidence. A brief description of the contents of each Level 2 report is also given. The ESC is supported by a large number of technical and scientific reports and references that we refer to as 'Level 3' documents. We have also produced a Guide to Key Points of the ESC, to help a wider group of stakeholders understand its nature, conclusions and implications.



Level 1	
Main Report [1]	
Level 2	
Management and dialogue	
Management and Dialogue [2]	Describes our environmental management systems and interactions with regulators and stakeholders
System characterisation and understanding	
Site History and Description [3]	Provides a history and description of the site
Disposal Facility Inventory [4]	Describes the wastes already disposed and wastes that may be disposed at the facility
Engineering Design [5]	Presents the engineering design of the current facility and proposed changes as further disposal vaults are built and the disposal facility is closed

Near Field [6]	Describes our understanding of the chemical and physical evolution of the engineered disposal system
Hydrogeology [7]	Describes our understanding of the geology and hydrogeology of the site
Site Evolution [8]	Describes our understanding of how the site will evolve, with a focus on coastal erosion
Monitoring [9]	Presents our programme of environmental monitoring supporting the ESC
Optimisation and Site Development Plan	
Optimisation and Site Development Plan [10]	Describes our approach to optimising the design and management of the disposal facility and wider site, and sets out our Site Development Plan
Waste Management Plan [11]	Presents our plans for managing the wastes produced by previous uses and operation of the site
Assessments	
Safety Functions [12]	Presents our understanding of how the different aspects of the repository system and its management contribute to the safety of the facility
Engineering Performance Assessment [13]	Presents our analysis of how the various components of the engineered disposal system will perform, which is an input into our impact assessments
Environmental Safety During the Period of Authorisation [14]	Presents evidence that the LLWR is currently being operated safely and will continue to be so during the period that the facility is permitted
Assessment of Long-term Radiological Impacts [15]	Presents evidence that, if the LLWR is managed in accordance with the Site Development Plan, the site will remain safe in the long term
Hydrogeological Risk Assessment (this report)	Presents evidence that the disposal facility protects groundwater from both radiological and non-radiological contaminants in the disposed wastes now and will continue to do so in the future

Assessment of Radiological Impacts on Non-human Biota [16]	Presents evidence that the LLWR does not have adverse consequences for non-human biota populations now and will not in the future
Implementation	
Implementation [17]	Sets out how we use the ESC to manage the site, including setting Waste Acceptance Criteria and other controls on the types and quantities of waste accepted for disposal
Audit	
Addressing Regulatory Requirements and Feedback [18]	Provides a cross-reference between the contents of the ESC and regulatory guidance and feedback

Executive Summary

This Hydrogeological Risk Assessment (HRA) provides evidence that the engineered waste disposal facilities at the Low Level Waste Repository (LLWR) will protect groundwater and surface water from pollutants in the wastes disposed of at the LLWR. The report addresses impacts to groundwater from radiological and non-radiological pollutants during, and after, the Period of Authorisation (PoA)² and forms part of the overall Environmental Safety Case (ESC) for the LLWR.

Guidance on the specific regulatory requirements relevant to near-surface disposal of solid radioactive waste is set out in the environment agencies' *'Near-surface Disposal Facilities on Land for Solid Radioactive Wastes: Guidance on Requirements for Authorisation'* (the GRA).

Requirement R10 states that:

'The developer/operator of a disposal facility for solid radioactive waste should demonstrate that the disposal system provides adequate protection against non-radiological hazards.'

A key part of Requirement R10 is demonstration of a level of protection that is *no less stringent* than that required to achieve *nationally acceptable standards for disposing of hazardous waste*. The relevant standards are the Water Framework Directive and the Groundwater Daughter Directive, as implemented through the Water Environment (Water Framework Directive) (England and Wales) Regulations 2017 and Schedules 22 and 23 of EPR 2016.

As part of this demonstration, we focus on presenting evidence to support the following safety arguments:

- optimisation of the management, design and operation of the LLWR with respect to radiological hazards will also act to limit as far as is practicable the potential impacts of non-radiological hazards from the LLWR;
- results from the current monitoring programmes at the LLWR demonstrate that appropriate standards for protection of people and the environment are being met during the operational phase;
- appropriate environmental monitoring programmes will be implemented during the period of active institutional control;
- assessment calculations demonstrate that releases to groundwater will comply with relevant regulatory standards during, and after, the PoA.

² The PoA is the period during which a Permit is held. For the LLWR, this encompasses disposal operations, the period after disposal operations cease when the site closure engineering is being installed and a subsequent period when the site is being monitored.

Design of the disposal system

Disposals to the LLWR were originally by tumble tipping into a series of seven trenches. This was a widespread and internationally accepted practice when the trenches were first commissioned. Post-placement engineering included the installation of an interim cap and a cut-off wall, which act to limit release of aqueous contaminants to the environment. Since 1995, all disposals at the LLWR have been to engineered vaults. The vault disposal system will ensure that the waste is contained within the facility *'over timescales at least as long as those provided for by the barriers at a site for disposal of non-radiological hazardous wastes.'*³

The following are key components of our engineered design for the vaults that have been identified as part of an ongoing process of design optimisation:

- a single domed final cap consistent with best practice to reduce infiltration;
- a cut-off wall to reduce lateral inflows into, and outflows from, the facility and which will also help to direct discharges to the underlying geology, rather than the near-surface environment;
- disposal of the wastes in steel containers, which reduces releases at early times;
- the use of cementitious grout, which provides a chemical barrier to the release of contaminants and provides a substrate for sorption, as well as reducing voidage in the waste stacks;
- removal of leachate from the vaults during the PoA to prevent the release of contaminants to the geosphere;
- design of a low sidewall in Vault 9 and subsequent vaults to ensure that unsaturated conditions continue to be promoted after the end of the PoA.

We consider that the optimised design provides a better long-term disposal solution for wastes than a conventional landfill design.

Quantitative HRA

We have undertaken a quantitative HRA, to assess the impacts of non-radiological (trench and vault) disposals. We consider this HRA to be no less robust than an equivalent HRA required for a hazardous waste landfill.

We assessed environmental impacts resulting from the whole waste inventory. We used a 'repository' approach, in which the source term was derived from the disposed inventory, approximating the effects of various physical and chemical processes in the near field.

We assess calculated concentrations of contaminants against LLWR Assessment Standards, which correspond to standards set out in regulatory guidance. In our reference assessment calculation, including known inventories for both the trenches and the vaults, we

³ Paragraph 6.4.2 of the GRA.

found no future LLWR Assessment Standard exceedances for non-radiological hazardous substances.

In the reference case assessment, calculated concentrations of non-hazardous pollutants were many orders of magnitude lower than the relevant LLWR Assessment Standards for the whole of the assessment time period.

Our assessment calculations indicate that, even if key barriers perform significantly less well than anticipated, there is no significant risk of hazardous substances or non-hazardous pollutants entering the environment as a result of disposals at the LLWR.

Similarly, our radiological assessment calculations demonstrate that we will prevent the input of radionuclides to groundwater during, and after, the PoA.

We have also investigated the potential for contaminants in the grout used in vault disposals to present an environmental risk. All calculated concentrations are several orders of magnitude lower than the relevant assessment criteria.

Overall, calculated concentrations are significantly lower than those calculated in the 2011 ESC and in the 2017 HRA. This is predominantly because we have a revised understanding of cap performance and we expect a much better level of performance than we did in the previous HRA and in the 2011 ESC.

Monitoring

We present evidence and arguments to demonstrate that our environmental monitoring programme is sufficient to '*confirm understanding of the effects that construction, operation and closure of the facility have on the characteristics of the site*,⁴ to demonstrate that the repository is compliant with relevant environmental standards, and to provide confidence that non-radiological disposals at the LLWR are not having an adverse impact on the quality of groundwater or surface water at or near the site.

Our extensive environmental monitoring programme includes regular monitoring of leachate, groundwater and surface water for potential non-radiological contaminants.

Leachate monitoring provides data on the hydrology of the trenches, enables assessment of interim cap integrity and tracks composition of the source term, identifying any changes and ultimately informing the non-radiological analysis suite across all three water systems.

Monitoring of groundwater levels enables assessment of the performance of site engineering and confirms or facilitates improvement of hydrogeological understanding. Groundwater quality monitoring identifies any potential issues relating to contaminant releases from the LLWR and tracks potential long-term variations in groundwater chemistry.

Surface water monitoring enables assessment of engineered barrier performance and facilitates understanding of site hydrology. Any potential repository-derived contamination of surface waters or other changes in surface water chemistry can also be identified.

⁴ Paragraph 6.4.34 of the GRA.

We review monitoring data to demonstrate that the LLWR is not having an adverse impact either on groundwater or on surface water. Most substances are present in groundwater, surface water and leachate at concentrations below their LLWR Assessment Standards. Where LLWR Assessment Standards are breached, this is generally due to elevated background concentrations relative to the LLWR Assessment Standards due to off-site activities. Groundwater and surface water quality remain consistent with levels quoted in the 2017 HRA, the 2011 ESC and with the findings of a major independent review of the hydrogeochemistry of non-radioactive contaminants reported in 2010. This found that groundwater contamination is not observed to be localised around the LLWR.

Where the concentrations of non-radiological contaminants in leachate exceed LLWR Assessment Standards, these concentrations are broadly consistent with the groundwater baseline. It is therefore considered unlikely that the impacts from these contaminants entering groundwater would be discernible from background concentrations. Overall, there is no evidence of an evolving risk to the environment from the non-radiological component of the waste.

On the basis of the arguments presented in this report, we consider that:

- there is not a significant risk of pollution in the future; and
- there will be no statistically, environmentally significant or sustained upward trend, with no significant increasing frequency in pollutant 'spikes'; and
- all necessary and reasonable measures have been taken to avoid the entry of hazardous substances into groundwater.

We consider, therefore, that we demonstrate that we prevent the input of hazardous substances and limit the input of non-hazardous pollutants to groundwater.

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1 Introduction

1.1 Objectives

This Hydrogeological Risk Assessment (HRA) provides evidence that the engineered waste disposal facilities at the Low Level Waste Repository (LLWR) will protect groundwater and surface water from pollutants in the wastes disposed of at the LLWR. The report addresses impacts to groundwater from radiological and non-radiological pollutants during, and after, the Period of Authorisation (PoA)⁵ and forms part of the overall Environmental Safety Case (ESC) for the LLWR.

The LLWR is regulated by the Environment Agency under Schedule 23 of the Environmental Permitting (England and Wales) Regulations 2016 (EPR 2016). Submission of the ESC is a requirement of the site's Permit. Guidance on the specific regulatory requirements relevant to near-surface disposal of solid radioactive waste is set out in the environment agencies' *'Near-surface Disposal Facilities on Land for Solid Radioactive Wastes: Guidance on Requirements for Authorisation'* (the GRA) [19].

Principle 3 of the GRA states that:

Principle 3: Level of protection against non-radiological hazards at the time of disposal and in the future

Solid radioactive waste shall be disposed of in such a way that the level of protection provided to people and the environment against any non-radiological hazards of the waste both at the time of disposal and in the future is consistent with that provided by the national standard at the time of disposal for wastes that present a non-radiological but not a radiological hazard.

This leads to a specific requirement (Requirement R10) in the GRA that:

Requirement R10: Protection against non-radiological hazards

The developer/operator of a disposal facility for solid radioactive waste should demonstrate that the disposal system provides adequate protection against non-radiological hazards.

The relevant standards to consider when demonstrating that disposals of solid radioactive waste at the LLWR address Principle 3 and Requirement R10 are the European Union (EU) Water Framework Directive 2000/60/EC (WFD) and its daughter directive, the Groundwater Daughter Directive 2006/118/EC (GWDD). This is because these are the standards which

⁵ The PoA is the period during which a Permit is held. For the LLWR, this encompasses disposal operations, the period after disposal operations cease when the site closure engineering is being installed and a subsequent period when the site is being monitored.

must be complied when disposing waste that presents a non-radiological, but not radiological, hazard.

Both directives are concerned with the protection of groundwater against pollution, the prevention and limitation of inputs of pollutants to groundwater and the prevention of deterioration of status of groundwater bodies.

In England, the WFD and the GWDD are implemented by the following pieces of legislation.

- The Water Environment (Water Framework Directive) (England and Wales) Regulations 2017 (WER 2017).
- The groundwater provisions in Schedule 22 of EPR 2016.

Disposal of solid radioactive waste in a near-surface disposal facility (such as the LLWR) constitutes a groundwater activity as defined in Schedule 22. This necessitates consideration of the protection of groundwater from the input of radionuclides present in the wastes.

The objective of this report is therefore to demonstrate that the engineered system will ensure that the impacts from radiological and non-radiological contaminants present in wastes disposed of, or planned to be disposed of, at the LLWR comply with the requirements of WER 2017 and Schedule 22 of EPR 2016.

As part of this demonstration, we focus on presenting evidence to support the following safety arguments:

- optimisation of the management, design and operation of the LLWR with respect to radiological hazards will also act to limit as far as is practicable the potential impacts of non-radiological hazards from the LLWR;
- results from the current monitoring programmes at the LLWR demonstrate that appropriate standards for protection of people and the environment are being met during the operational phase;
- appropriate environmental monitoring programmes will be implemented during the period of active institutional control;
- assessment calculations demonstrate that releases to groundwater will comply with relevant regulatory standards during, and after, the PoA.

The scope and structure of the report are defined and outlined in Subsections 1.2 and 1.3, respectively.

1.2 Scope

This report considers the radiological and non-radiological impacts to groundwater that may be associated with the disposal of solid radioactive waste, including less-hazardous Intermediate Level Waste (ILW), at the LLWR. We have included less-hazardous ILW at the request of the NDA, who have asked us to explore options for disposing of ILW at the LLWR

site, in line with UK government policy [20] to consider alternative options for the management of ILW. The UK government policy advocates engagement with regulators on the potential to use the LLWR to dispose of less-hazardous ILW. Our inclusion of ILW within the scope of the 2026 ESC is a means by which we intend to engage with the regulator to explore the potential to dispose of ILW at the LLWR site. However, it does not indicate that a decision to dispose of ILW has been made.

The following aspects are considered in this report:

- the design of the facility;
- the radiological and non-radiological components of the inventory;
- the hydrogeological conceptual model;
- Environmental Assessment Limits (EALs), Minimum Reporting Values (MRVs) and compliance points;
- groundwater assessment calculations for the complete lifecycle of the LLWR;
- design of the monitoring programme;
- review of past and current monitoring data.

As an HRA, the focus of this report is on environmental impacts arising from release of contaminants to groundwater. The impacts of radionuclide exposures which may occur via other pathways (e.g. gaseous emissions) are addressed in the '*Environmental Safety During the Period of Authorisation*' [14] and the '*Assessment of Long-term Radiological Impacts*' [15] reports. With the exception of asbestos, following advice from the Environment Agency [21] we do not assess non-radiological impacts that may arise from the human intrusion and coastal erosion pathways. Our assessment of impacts from asbestos is presented in reference [22]. We lay out our technical position on chemotoxic and flammable gases in our '*Near Field*' report [6] .

The report does not address waste acceptance, although non-radiological capacities (which describe the maximum amount of each contaminant that can safely be disposed of), and Waste Acceptance Criteria (WAC), are derived using the same models and reference assessment assumptions as presented in this report. The '*Implementation*' report [17] describes how the results of the assessments are used to derive WAC and manage disposals to the repository.

The scope of this report excludes assessment of the impact of any legacy or more recent sources of contamination at the LLWR that could pose a hazard to people or the environment. Such sources include:

- the legacy of the Royal Ordnance Factory (ROF) operations and buildings [3];
- storage of chemicals within the Plutonium Contaminated Material (PCM) magazines and adjacent area [3];
- potential contaminant migration from locations outside the site boundary;

- LLWR operations other than waste disposal, including transport, waste conditioning and herbicide application;
- drainage systems and foul sewers.

The radiological and non-radiological impacts arising from legacy or more recent contamination have been assessed in a separate site characterisation report [23] and in the '*Environmental Safety During the Period of Authorisation*' [14] report. Our plans for the treatment of legacy or more recent contamination are set out in the '*Waste Management Plan*' report [11].

We have included as much information in this report as practicable without making the document unduly lengthy. Where relevant, therefore, reference is made to other ESC reports for background and supporting information.

- The '*Site History and Description*' report [3] provides a detailed description of the LLWR site and the historical development of the disposal facility trenches and vaults.
- The waste inventory of the LLWR is summarised in the '*Disposal Facility Inventory*' report [4].
- The development of the geological and hydrogeological conceptual models used to support the HRA is provided in the '*Hydrogeology*' report [7].
- The non-radiological calculations presented in this HRA determine contaminant concentrations in groundwater at appropriate compliance points. Concentrations obtained from our monitoring programme are also relevant and are discussed as part of our assessments. These data are compiled from the '*Monitoring*' report [9] and underlying reports.
- The radiological assessments that demonstrate that we prevent the input of radionuclides to groundwater are summarised in the '*Environmental Safety During the Period of Authorisation*' [14], the '*Assessment of Long-term Radiological Impacts*' [15] and the '*Assessment of Impacts on Non-human Biota*' [16] reports. The mathematical details underpinning our groundwater assessment calculations are presented in two detailed Level 3 reports [24, 25].
- Information on the design of the facility and optimisation of the facility with respect to radiological impacts are described in the '*Engineering Design*' [5] and the '*Optimisation and Site Development Plan*' [10] reports, respectively.

Reference to underpinning reports may be required for certain details. In particular, we set out the following information in reference [25]:

- mathematical and implementation details of the performance assessment model used in the quantitative HRA;
- a detailed consideration of the non-radiological inventory used in the assessment calculations;

- tables of data used in the assessment model;
- detailed results from sensitivity studies performed as part of the assessment;
- the treatment of organic complexation within the assessment.

1.3 Structure

The report is structured as follows.

- Section 2 provides the background to the current work. It first sets out the regulatory context for the main legislative and regulatory requirements and guidance of relevance, as well as giving a brief overview of the current HRA and how it relates to previous related work. It then presents a brief overview of the site, including the setting and surface water bodies. The geology, hydrogeology and hydrogeological conceptual model are presented, before providing a summary of past, present and future operations. Finally, EALs and MRVs are defined, against which measured concentrations and quantitative HRA outputs can be assessed.
- Section 3 summarises the disposal system, including the inventory and the different disposal regimes and various aspects of the site engineering designed to limit environmental impacts and risks to people. This section concludes with a discussion of the key aspects of the disposal system design from the perspective of limiting contaminant release.
- Section 4 presents the quantitative HRA. The format of this section follows as closely as possible the Environment Agency's recommended template for a landfill HRA.
- Section 5 discusses aspects of the monitoring programme, including management of the programme, the methodology used to analyse the data, baseline water quality, the monitoring plan for leachate, groundwater and surface water, and concluding with a review and discussion of the monitored data.
- Section 6 presents the conclusions. The evidence supporting regulatory compliance is presented and discussed. We present remaining significant uncertainties and, where appropriate, our plans to address them.

2 Background

2.1 Report Context

This report, and the work documented herein, should be viewed within the context of the wider operations of the LLWR. The LLWR has been designed as a disposal facility for solid radioactive waste. As noted in Subsection 2.1.1, the regulatory context of the LLWR differs from that applicable to a landfill facility. Specific design guidance for landfills [26, 27] is therefore not directly applicable to the LLWR, although it provides a useful benchmark.

2.1.1 Regulatory Context

As noted in Subsection 1.1, the GRA (Requirement R10) requires that:

Requirement R10: Protection against non-radiological hazards

The developer/operator of a disposal facility for solid radioactive waste should demonstrate that the disposal system provides adequate protection against non-radiological hazards.

Paragraph 6.4.2 of the GRA gives guidance on how Requirement R10 may be met, stating that (emphasis added):

Some waste disposed of at a facility receiving radioactive waste may be potentially harmful wholly or partly because of its non-radioactive properties. There are *nationally acceptable standards for disposing of hazardous waste*. However, these standards may not be suitable to apply directly to waste that presents both radiological and non-radiological hazards. Accordingly, *these standards need not necessarily be applied, but a level of protection should be provided against the non-radiological hazards that is no less stringent than would be provided if the standards were applied*. This could be achieved by ensuring that materials posing non-radiological hazards are contained within the facility over timescales at least as long as those provided for by the barriers at a site for disposal of non-radiological hazardous wastes.

A key part of Requirement R10 is therefore demonstration of a level of protection that is *no less stringent* than that required to achieve *nationally acceptable standards for disposing of hazardous waste*. As noted in Subsection 1.1, the relevant standards are the Water Framework Directive (WFD) and the Groundwater Daughter Directive (GWDD), as implemented through the WER 2017 and Schedule 22 of EPR 2016.

The GRA also includes a requirement (Requirement R14) relating to monitoring:

Requirement R14: Monitoring

In support of the environmental safety case, the developer/operator of a disposal facility for solid radioactive waste should carry out a programme to monitor for changes caused by construction, operation and closure of the facility.

As explained in paragraph 6.4.34, this includes:

- radiological monitoring during the PoA to provide assurance of radiological protection to members of the public;
- monitoring of non-radiological parameters during construction and the PoA to:
 - *confirm understanding of the effects that construction, operation and closure of the facility have on the characteristics of the site;*
 - *to demonstrate that the changes in, and evolution of, the parameters monitored are consistent with the environmental safety case.*

Our monitoring programme is described in reference [9] and we present key results relevant to demonstrating protection of groundwater and surface waters in Section 5. Our monitoring programme considers leachate, surface water and groundwater.

WER 2017

For groundwater bodies, paragraph 13(5) of WER 2017 sets out the following objectives:

- (a) prevent deterioration of the status of each body of groundwater;*
- (b) prevent or limit the input of pollutants into groundwater;*
- (c) protect, enhance and restore each body of groundwater, and ensure a balance between abstraction and recharge of groundwater, with the aim of achieving good groundwater chemical status and good groundwater quantitative status, if not already achieved, by 22nd December 2021;*
- (d) reverse any significant and sustained upward trend in the concentration of any pollutant resulting from the impact of human activity in order to progressively reduce pollution of groundwater.'*

Thus, the legislation requires consideration of protection against:

- pollution of groundwater;
- over-exploitation of groundwater.

Activities at the LLWR do not involve large-scale abstraction of groundwater. The focus of this report, therefore, is on demonstrating that we address the objectives related to groundwater quality.

In order to ensure protection of groundwater quality, WER 2017 places a *prohibition of direct discharges of pollutants into groundwater* [28] and explains that:

“direct discharges of pollutants into groundwater’ means the discharge of pollutants into groundwater without percolation through the soil or subsoil.’

In the context of protecting groundwater when permitting disposals of radioactive waste, the Environment Agency advises that [29]:

‘...we would consider that a direct input⁶ occurs if the pollutant is introduced at any location below the typical maximum upper level of the saturated layer of an unconfined aquifer.’

The Environment Agency explains further that:

“Typical’ in this context would employ a representative winter water table level, based on hydrogeological records and/or expert opinion, and discounting extremes in weather, or artificial suppression by engineering techniques such as pumping.’

Environment Agency guidance [29] states that, with respect to the 'prohibition' requirement:

‘The assessment of whether a direct discharge is likely to occur should consider the anticipated life of the permit, as far into the future as is reasonably practicable.’

All vaults at the LLWR either have been, or will be, constructed from the surface and will not extend below the regional groundwater table [5]. Over the lifetime of the site's Permit, the Regional Groundwater will remain below the level of the vaults, as demonstrated in our hydrogeological studies [30]. The 'prohibition' requirement will therefore be satisfied by the LLWR as any discharges of pollutants must percolate through the underlying soil before entering the Regional Groundwater.⁷

EPR 2016

The requirements of EPR 2016 are distinct from, albeit complementary to, those of WER 2017. Paragraph 6 of Schedule 22 of EPR 2016 explains that, for groundwater activities:

‘For the purposes of implementing the Water Framework Directive and the Groundwater Directive, the regulator must, in exercising its relevant functions, take all necessary measures -

a) to prevent the input of any hazardous substances to groundwater, and

⁶ EPR 2016 refers to 'inputs' rather than 'discharges', which is the term used in WER 2017. The Environment Agency explains that, in the context of EPR 2016, the two terms can be taken to be interchangeable [28].

⁷ This is consistent with the Environment Agency's guidance on compliance points, which advises that these should be 'just below the water table' [29, 86].

b) to limit the input of non-hazardous pollutants to groundwater so as to ensure that such inputs do not cause pollution of groundwater.'

Schedule 22 of EPR 2016 provides the following definitions:

- a hazardous substance is *any substance or group of substances that are toxic, persistent and liable to bio-accumulate, or give rise to an equivalent level of concern;*
- a non-hazardous pollutant *is any pollutant other than a hazardous substance.*

The UK environment agencies have published a methodology [31] to determine whether, for the purposes of groundwater protection, potential pollutants should be considered to be hazardous substances or non-hazardous pollutants.

Reference [31] states that all radioactive substances are hazardous substances, where radioactive substances *are taken to be any substance that contains one or more radionuclides, the activity of which cannot be disregarded as far as radiation protection is concerned.* It follows that only substances which are within scope ('in scope') of radioactive substances regulations should, based on radioactive properties, be treated as hazardous substances with regard to groundwater protection. Instead, substances that are 'out of scope'⁸ would be hazardous substances or non-hazardous pollutants by virtue of their chemotoxic properties.

Schedule 23 of EPR 2016 provides 'out of scope'⁹ values for radionuclides in solids. We treat all radionuclides present in 'in scope' disposed solid waste as hazardous substances.

Preventing the input of hazardous substances

An input of hazardous substances to groundwater would be prevented if [32]:

'1) there is no discernible concentration of a hazardous substance in the discharge; or

2) there are no discernible concentrations of hazardous substances attributable to the discharge in groundwater immediately down-gradient of the discharge zone, subject to adequate monitoring (or in the case of new discharges a detailed predictive hydro-geological impact assessment); or

3) there are (or are predicted to be) discernible concentrations of hazardous substances in the groundwater down-gradient of the discharge zone attributable to the discharge but all of the following conditions apply:

⁸ Radioactive substances that are 'out of scope' are not subject to radioactive substances regulations. That is, from a legislative and regulatory perspective, they are 'not radioactive'.

⁹ 'Out of scope' values define the minimum activity concentration which may be present in a substance for it to be subject to radioactive substances regulations: if the concentrations were to be above these values the substance would be 'in scope' of radioactive substances regulations.

(a) concentrations will not result in any actual pollution or a significant risk of pollution in the future; and

(b) there is no progressive increase in the concentration of hazardous substances outside the immediate discharge zone, i.e. there will be no statistically and environmentally significant and sustained upward trend or significant increase in pollutant spikes, and

(c) all necessary and reasonable measures to avoid the entry of hazardous substances into groundwater have been taken.'

It is important, therefore, to consider the discernibility of a hazardous substance immediately down-gradient of the source. Groundwater protection guidance produced by the Environment Agency [33] advises that a hazardous pollutant would be discernible if the concentration in groundwater is greater than either:

- that found naturally in groundwater;
- a MRV, usually the limit of quantification or other value set out in legislation.

Whichever of these has the highest concentration would be the discernible concentration.

Reference [32] explains that a 'reasonable measure' is one which is *technically feasible, not disproportionately costly and ... within the control of the operator*. These are similar considerations to those applied when identifying Best Available Techniques (BAT) to prevent or minimise the release of pollutants to the environment.

Thus, even if a hazardous substance is discernible, it may be possible to demonstrate compliance with the 'prevent' requirement of EPR 2016, provided that [29]:

- all necessary and reasonable measures have been taken;
- the discernible input does not compromise the achievement of the of the other requirements of the WFD and GWDD set out in the WER 2017.

For non-radiological contaminants, our approach to demonstrating that we prevent the input of hazardous substances to groundwater is as follows.

- During the PoA we use a combination of monitoring data and calculations using our groundwater assessment to demonstrate that there are no discernible concentrations of hazardous substances derived from the disposed wastes and grout at appropriate compliance points. Consistent with Environment Agency guidance [34], we consider the potential for surface water bodies to be receptors that receive flow from groundwater and we therefore provide monitoring data for surface water as well as for groundwater.
- After the PoA, it is not possible to rely on monitoring data and therefore, for this period, we use our groundwater assessment model to demonstrate that there are no discernible concentrations of hazardous substances derived from the disposed wastes and grout at appropriate compliance points.

We set out the relevant results and arguments in Sections 4 and 5.

Demonstration of 'Prevent' for radionuclides

Environment Agency guidance [29] explains that, during the PoA, an input of a radioactive substance to groundwater would be prevented if:

- the annual radiation dose to members of the public incurred via the groundwater pathway is consistent with, or lower than, 0.01 mSv; or,
- if the annual radiation dose exceeds 0.01 mSv, to demonstrate that:
 - the annual dose to a representative person through all pathways during the PoA is less than 0.3 mSv; and,
 - doses are as low as reasonably achievable, taking into account economic and social factors (i.e. the repository and the waste management arrangements are optimised); or,
- the concentration of radionuclides (in Bq l⁻¹) attributable to the discharge in groundwater immediately down-gradient of the discharge zone is consistent with background concentrations in this or a similar geological formation.

In addition to these criteria, it is necessary to demonstrate protection of non-human biota inhabiting groundwater-fed ecosystems. Dose rates via the groundwater to biota in such ecosystems should be *well below levels considered to result in adverse effects for the type of organism concerned* [29].

As set out in our 'Assessment of Impacts on Non-human Biota' [16], we use an ESC screening dose rate of 10 µGy h⁻¹ when assessing radiological impacts to non-human biota. Assessed dose rates lower than this value would not result in harm to populations.

Where the ESC screening dose rate of 10 µGy h⁻¹ is exceeded, we consider the radiosensitivity of organisms, via the International Commission on Radiological Protection (ICRP) Derived Consideration Reference Levels (DCRLs), the time dependency of the environmental concentrations (and, thus, the dose rates), and the area required to support a population of those organisms. DCRLs are order of magnitude bands of dose rate for a selection of organisms, within which there is some chance of deleterious effects occurring to individuals of that type of organism but where harm to populations is not expected.

Reference [29] states that, after the PoA, an input of radionuclides to groundwater would be considered to be prevented if:

- the radiological risk to members of the public through the groundwater pathway after the end of the PoA is consistent with, or lower than, an assessed risk of 10⁻⁶ y⁻¹; or,
- if the assessed risk through the groundwater pathway exceeds 10⁻⁶ y⁻¹, risks are as low as reasonably achievable, taking into account economic and social factors (i.e. the repository and the waste management arrangements are optimised); or,

- the concentration of radionuclides (in Bq l⁻¹) attributable to the discharge in groundwater immediately down-gradient of the discharge zone is consistent with background concentrations in this or a similar geological formation.

Requirement R6 of the GRA states that:

Requirement R6: Risk guidance level after the period of authorisation

After the period of authorisation, the assessed radiological risk from a disposal facility to a person representative of those at greatest risk should be consistent with a risk guidance level of 10⁻⁶ per year (i.e. 1 in a million per year).

It follows, therefore, that the input of radionuclides to groundwater is considered to be prevented if compliance with the risk guidance level in Requirement R6 can be demonstrated.

The GRA also includes a requirement (Requirement R8) relating to optimisation:

Requirement R8: Optimisation

The choice of waste acceptance criteria, how the selected site is used and the design, construction, operation, closure and post-closure management of the disposal facility should ensure that radiological risks to members of the public, both during the period of authorisation and afterwards, are as low as reasonably achievable (ALARA), taking into account economic and social factors.

Addressing Requirement R8 therefore plays an important part in establishing that the input of radionuclides to groundwater has been prevented.

Taking all of this into account, our approach to preventing the input of radionuclides into groundwater is as follows.

- During the PoA, we use a combination of monitoring data and assessment calculations to demonstrate that:
 - assessed annual doses to a representative person are less than 0.01 mSv, which we report in our '*Assessment of Environmental Safety During the Period of Authorisation*' [14];
 - assessed dose rates to non-human biota in groundwater-fed ecosystems will not cause harm to populations, which we report in our '*Assessment of Impacts on Non-human Biota*' [16].
- After the PoA, we use assessment calculations to demonstrate that assessed risks to the public via the groundwater pathway comply with the risk guidance level in Requirement R6, which we report in our '*Assessment of Long-term Radiological Impacts*' [15].

- Demonstrating that our repository design and waste management arrangements are optimised. We report these in our '*Optimisation and Site Development Plan*' [10] and '*Implementation*' [17] reports, respectively.

Limiting the input of non-hazardous pollutants to groundwater

As noted earlier, EPR 2016 requires that the input of non-hazardous pollutants to groundwater is limited so as to ensure that such inputs do not cause pollution of groundwater.

To understand how this requirement is met, it is necessary to consider the definition of pollution in EPR 2016 and WER 2017. This is defined as:

'the direct or indirect introduction, as a result of human activity, of substances or heat into the air, water or land which may:

- a) be harmful to human health or the quality of aquatic ecosystems or terrestrial ecosystems directly depending on aquatic ecosystems,*
- b) result in damage to material property, or*
- c) impair or interfere with amenities or other legitimate uses of the environment.'*

It follows, therefore, that pollution will occur if an input causes any of these consequences to occur. This implies that limiting the input requires that the input is sufficiently low so as to avoid these detriments being incurred.

The environmental objectives for groundwater bodies set out in WER 2017 imply further that 'limit the input' requires the following:

- no deterioration in the status of a groundwater body;
- no significant and sustained rising trends in the concentrations of pollutants in groundwater.

The United Kingdom Technical Advisory Group provides advice to the environment agencies on the standards necessary to limit point source inputs of non-hazardous pollutants so as to avoid 'pollution' [35]:

'The standards will vary depending on the relevant receptor:

Where the receptor is a river, groundwater dependent wetland or a particular abstraction from groundwater for drinking water, standards are dictated by the relevant Environmental Quality Standard (EQS) or Drinking Water Standard. Where an abstraction is used for purposes other than drinking water, the relevant standards will apply.

Where there is no abstraction but where there is potential for future development because a groundwater body forms part of a Drinking Water

Protected Area, standards are based on current available standards for protecting human health in drinking water supplies. In this case, the standards apply directly to the groundwater body. For each pollutant, the standard is selected by going through the following hierarchy, in sequence:

a) a national drinking water standard for the pollutant established under domestic legislation, including legislation implementing the Drinking Water Directive;

b) if no standard is available via (a), a standard specified in World Health Organisation (WHO) Guidelines for Drinking Water Quality;

c) if no standard is available via (a) or (b), a standard established following peer review by a national authority in another country;

d) if no suitable standard is available via any of the above, an operational value adopted by the agencies based on the best available scientific information on the pollutant concerned.

Where no abstraction or ecosystem receptor requires more stringent protection, the standards derived using this hierarchy would normally apply at a distance of 50 metres from the source of the inputs in the direction of groundwater flow. This distance may be extended up to a maximum of 250 metres where there are other constraints on the future development of the local groundwater resource.'

For non-radiological contaminants, our approach to demonstrating that we limit the input of non-hazardous pollutants to groundwater is therefore as follows.

- During the PoA we use a combination of monitoring data and calculations using our groundwater assessment to demonstrate that pollutant concentrations in groundwater are lower than appropriate regulatory standards and that there are no significant and sustained rising trends in pollutant concentrations in groundwater. Consistent with Environment Agency guidance [34], we consider the potential for surface water bodies to be receptors that receive flow from groundwater and we therefore provide monitoring data for surface water as well as for groundwater.
- After the PoA, it is not possible to rely on monitoring data and therefore, for this period, we use our groundwater assessment model to demonstrate that pollutant concentrations in groundwater are lower than appropriate regulatory standards and that there are no significant and sustained rising trends in pollutant concentrations in groundwater.

We set out the relevant results and arguments in Sections 4 and 5.

2.1.2 Environmental Assessment Limits and Minimum Reporting Values

For consistency with radiological aspects of the ESC, we define EALs for non-radiological contaminants in water in terms of LLWR Assessment Standards (LLWRASs). For hazardous substances, the LLWRAS is taken to be the most restrictive of:

- the Environmental Quality Standard (EQS);
- EU Drinking Water Directive (DWD) standards [36];
- the MRV, which are taken, where possible, from the official Environment Agency list [37];
- the limit of detection (LoD) achievable by our contracted laboratory.

For non-hazardous pollutants, LLWRASs for non-radiological contaminants are currently based on the most restrictive EQSs for fresh water or salt water [36]. Where no EQS is available, the EU DWD standards [36] have been used. LLWRASs are reviewed on an annual basis to ensure they align with any changes to the respective standards.

Where an EQS is quoted as a range of values, the LLWRAS is calculated based on the EQS designated limiting value, usually water hardness, during each monitoring campaign. The quantitative HRA utilises the minimum EQS value against which to assess calculated concentrations in groundwater.

Table 2.1 shows the current LLWRAS for non-radiological contaminants. These LLWRAS are used to assess the outputs of the assessment calculations reported in Section 4. The selection of determinands is presented in Subsection 4.4. They are also used to assess measured contaminant concentrations in surface water and leachate. We review the LLWRAS on an annual basis and update periodically.

Comparison of groundwater monitoring results with LLWRASs does not consider background levels of contamination and, as such, exceedance of the LLWRAS is not necessarily an indication of pollution as background levels may already exceed the LLWRAS. Instead, we have defined Monitoring Assessment Levels (MALs), which are used within the monitoring programme (Section 5) to assess the measured contaminant concentrations in groundwater. The MALs are set to be the highest of the LLWRAS and the level discernible above background. The discernible level and the MALs are presented in Table 2.1. The use of MALs is discussed further in Subsection 5.2.3.

Table 2.1: LLWRAS and MALs for non-radiological contaminants in groundwater

Determinand	Hazardous substance or non-hazardous pollutant	Units	EQS ¹⁰ 2023	DWD ¹¹	MRV ¹²	LOD	LLWRAS	GW Discernible Level	GW MAL
Aluminium	Non-hazardous	µg l ⁻¹		200		10	200	-	200
Ammoniacal nitrogen	Non-hazardous	µg l ⁻¹	N/A			4	15	-	15
Ammonium as NH ₄	Non-hazardous	µg l ⁻¹	N/A			4	1,000	-	1,000
Antimony	Non-hazardous	µg l ⁻¹		20		1	5	-	5
Arsenic	Hazardous	µg l ⁻¹	50	10	1	1	1	19	19
Benzo-a-pyrene (BAP)	Hazardous	µg l ⁻¹	0.05	0.01		0.01	0.01		

¹⁰ All EQS levels are annual average levels, rather than maximum allowable levels (unless otherwise stated).

¹¹ DWD levels are taken from the Water Supply (Water Quality) Regulations 2016.

¹² MRVs are presented only for hazardous substances.

Determinand	Hazardous substance or non-hazardous pollutant	Units	EQS ¹⁰ 2023	DWD ¹¹	MRV ¹²	LOD	LLWRAS	GW Discernible Level	GW MAL
Barium	Non-hazardous	µg l ⁻¹		1,300		1	700	498	700
Benzene	Hazardous	µg l ⁻¹	10	1	1	1	1	-	1
Beryllium	Non-hazardous	µg l ⁻¹				1	4	-	4
Boron	Non-hazardous	µg l ⁻¹	2,000	2,400		1	2,000	203	2,000

Determinand	Hazardous substance or non-hazardous pollutant	Units	EQS ¹⁰ 2023	DWD ¹¹	MRV ¹²	LOD	LLWRAS	GW Discernible Level	GW MAL
Cadmium	Non-hazardous	µg l ⁻¹	≤0.08-0.25 (≤0.08 for CaCO ₃ per litre <40 mg, 0.08 for 40-50 mg, 0.09 for 50-100 mg, 0.15 for 100-200 mg and 0.25 for >200 mg CaCO ₃)	3		1	5	3.83	3.83
Chromium (Total)	Non-hazardous	µg l ⁻¹		50		1	50	-	50
Chromium (III)	Non-hazardous	µg l ⁻¹	4.7			10	4.7	6.77	10

Determinand	Hazardous substance or non-hazardous pollutant	Units	EQS ¹⁰ 2023	DWD ¹¹	MRV ¹²	LOD	LLWRAS	GW Discernible Level	GW MAL
Chromium (VI)	Hazardous	µg l ⁻¹	3.4			10	3.4	-	10
Cobalt	Non-hazardous	µg l ⁻¹	3			3	3	-	3
Copper	Non-hazardous	µg l ⁻¹	1 (bioavailable)	2000		1	1.0	112	112
Cyanide	Non-hazardous	µg l ⁻¹	1	50		9	1	29.3	29.3
Fluoride	Non-hazardous	µg l ⁻¹	1,000-5,000 (more than 50 mg l ⁻¹ CaCO ₃)	1500		200	1,000	459	459
Iron	Non-hazardous	µg l ⁻¹	1,000	200		10	1000	3,420	3,420

Determinand	Hazardous substance or non-hazardous pollutant	Units	EQS ¹⁰ 2023	DWD ¹¹	MRV ¹²	LOD	LLWRAS	GW Discernible Level	GW MAL
Lead	Hazardous	µg l ⁻¹	1.2 (bioavailable)	10	1	1	1	20.4	20.4
Manganese	Non-hazardous	µg l ⁻¹	123 (bioavailable)	50		0.1	30	4,700	4,700
Mercury	Hazardous	µg l ⁻¹	N/A average (0.07 max allowable)	1	0.01	0.1	0.01	1.99	1.99
Molybdenum	Non-hazardous	µg l ⁻¹				1	70	-	70
Nickel	Non-hazardous	µg l ⁻¹	4 (bioavailable)	70		1	20	36.4	36.4
Nitrate as NO ₃	Non-hazardous	µg l ⁻¹		50,000		1,000	50,000	11,300	50,000
Nitrite	Non-hazardous	µg l ⁻¹		3,000		200	500	309	309

Determinand	Hazardous substance or non-hazardous pollutant	Units	EQS ¹⁰ 2023	DWD ¹¹	MRV ¹²	LOD	LLWRAS	GW Discernible Level	GW MAL
Phenol	Non-hazardous	µg l ⁻¹	7.7			0.5	7.7	168	168
Selenium	Non-hazardous	µg l ⁻¹		40		1	10	13.3	13.3
Sodium	Non-hazardous	µg l ⁻¹		200,000		-	-	-	-
Sulphate as SO ₄	Non-hazardous	µg l ⁻¹	400,000	250,000		1,000	400,000	165,000	165,000
Tin (inorganic)	Non-hazardous	µg l ⁻¹	25			10	25	-	25
Tributyl phosphate (TBP)	Hazardous	µg l ⁻¹	50			10	10	-	-

Determinand	Hazardous substance or non-hazardous pollutant	Units	EQS ¹⁰ 2023	DWD ¹¹	MRV ¹²	LOD	LLWRAS	GW Discernible Level	GW MAL
Vanadium	Non-hazardous	µg l ⁻¹	20-60 (more than 200 mg l ⁻¹ CaCO ₃)			1	20	-	20
Vinyl chloride	Hazardous	µg l ⁻¹			0.5	5	0.5	-	5
Uranium	Non-hazardous	µg l ⁻¹		30		0.01	15	-	15
Zinc	Non-hazardous	µg l ⁻¹	10.9 (bioavailable)			1	8	315	315

2.1.3 Previous Submissions

A risk assessment relating to the non-radiological impacts of disposals to the LLWR was included in the 2011 ESC [38]. The assessment was conducted in parallel with a radiological assessment and, as far as possible, adopted a consistent approach. In particular, the near field was modelled using a 'repository' approach in which contaminant release from the near field into groundwater was modelled mechanistically, starting from the declared inventory and accounting for physical and chemical processes relevant to the near field. This contrasts with the approach normally adopted for landfill risk assessment, where measured concentrations in leachate are used as the source term for contaminants entering groundwater. The arguments as to why we use the 'repository' approach are presented in Subsection 4.1.1.

The 2011 non-radiological risk assessment considered environmental impacts arising from the following pathways:

- groundwater;
- coastal erosion;
- gas;
- human intrusion.

The groundwater pathway was considered in most detail and included separate assessments for the PoA and the period thereafter.

For the PoA, peak concentrations of non-radiological hazardous substances and non-hazardous pollutants were calculated in regional groundwater directly beneath the trenches.¹³ Average concentrations were also calculated in the regional groundwater between the LLWR and the coast. For the period after the PoA, peak concentrations were calculated directly beneath both trenches and vaults, while average concentrations were again calculated between the LLWR and the coast.

Subsequent to the 2011 ESC, we further developed our approach to the assessment of non-radiological contaminants (see Subsection 2.9 of reference [39]). At the Environment Agency's request, we also undertook calculations for compliance points that would be applicable under landfill guidance [40]. As part of this programme of work we calculated capacities for use in limiting disposals of those substances that have potential to give rise to unacceptable pollution of groundwater. These capacities were set out in reference [39].

In its review of the 2011 ESC, the Environment Agency noted that [41]:

'Overall, we consider that the requirements of the GRA for the non-radiological assessment pathway have been adequately met, although we see scope for future improvement.'

¹³ Contaminant release from the vaults was assumed not to occur because the containers and the removal of any leachate that might be generated by pumping would limit releases to groundwater from the vaults.

The Environment Agency also noted that while Requirement R10 necessitates that a *level of protection against non-radiological hazards should be provided that is no less stringent than would be provided if the hazardous waste standards were applied*, a hazardous landfill operator would not be required to assess non-radiological impacts arising from the coastal erosion and human intrusion pathways [41]. Following this guidance, we have not conducted assessments of non-radiological impacts for these two pathways in the 2026 ESC.¹⁴ We do assess the radiological consequences for these two pathways, which we report in our 'Assessment of Long-term Radiological Impacts' report [15].

Our technical position with respect to non-radiological bulk gases is presented in the 'Near Field' report [6].

A specific recommendation in the Environment Agency's review of the 2011 ESC was for an updated HRA to be produced. This recommendation was formally set out as a Forward Issue (ESC-FI-006) [42]. Specifically, it was noted in ESC-FI-006 that:

'The final report should seek to:

outline and describe the approach taken to the assessment of non-radiological groundwater assessment;

identify and demonstrate that necessary and reasonable measures to avoid the entry of hazardous substances into groundwater have been taken;

demonstrate the current compliance status of historical disposals;

assess the full life impacts of the authorised activity on the groundwater directive compliance points;

provide a reporting format which can easily be updated.'

The requirement for an updated HRA was also a formal Improvement Condition (IC6) of the site Permit [43].

The updated HRA is documented in reference [44]. We made all reasonable efforts to address specific requirements arising from the Environment Agency's review [41], as formally addressed in ESC-FI-006 [42]. In particular, the updated HRA included:

- updated compliance points, more in line with a conventional landfill HRA;

¹⁴ In its review of the 2011 ESC, the Environment Agency recommended that we continue to develop our approach to assessing impacts from asbestos following disruption of the repository by human intrusion or coastal erosion. As a responsible operator, we commissioned work to develop a science-based understanding of asbestos to support our ESC and the development of WAC [22]. As discussed in our 'Implementation' report [17], we have reviewed the assessment in reference [22] and we have concluded that it remains valid and that no update to our asbestos WAC is required.

- recent reclassification of certain pollutants as hazardous substances by the Joint Agencies Groundwater Directive Advisory Group (JAGDAG);
- use of a better characterised trench source term;
- consideration of non-radiological hazardous substances and non-hazardous pollutants in grout;
- use of a single model to represent both the PoA and the post-PoA period;
- use of the 2013 UK Radioactive Waste Inventory (UKRWI), as opposed to the 2007 inventory.

Other updates, not directly arising from the Environment Agency's review of the 2011 ESC, included:

- updated hydrogeological understanding and modelling;
- revised chemical properties for some substances;
- consideration of the effects of organic complexation on contaminant mobility.

When producing that report, we were mindful of Recommendation ASS40 of the Environment Agency's review [41] of the 2011 ESC, that future HRAs should be improved by:

'documenting the assessment in a stand-alone report containing all relevant monitoring, conceptualisation and assessment information and avoiding the need to refer to numerous supporting documents.'

The report, therefore, presented all such information at a level of detail considered to be appropriate and sufficient to enable overall review of our HRA. However, given the large amount of supporting information and the objective to keep the document to a reasonable length, we were not able to include all detailed underpinning information.

In particular, we did not include mathematical details of the quantitative assessment methodology, which were reported in an underlying report [45]. In identifying the appropriate content and structure of this HRA, our principal frame of reference was provided by recent HRAs undertaken for landfill sites. We have continued with this approach for this HRA, noting that the underlying references have changed and are all part of the suite of documents which comprise the 2026 ESC.

In its review of the 2017 HRA [46] the Environment Agency stated that:

'The 2017 HRA satisfactorily demonstrates that all necessary and reasonable measures have been taken to ensure compliance criteria are likely to be met, or if compliance criteria are not likely to be met, able to demonstrate that: (a) that there is not a significant risk of pollution in the future; and (b) that there will be no statistically, environmentally significant or sustained upward trend, with no significant increasing frequency in

pollutant 'spikes', and (c) that all necessary and reasonable measures have been taken to avoid the entry of hazardous substances into groundwater.'

The Environment Agency concluded that:

'...we are satisfied that the LLWR does offer a level of protection in excess of that provided by an equivalent directive waste landfill.'

On this basis, the Environment Agency closed IC6 and further advised that:

'...as the 2017 HRA is a sufficiently robust assessment and the LLWR site has sufficient monitoring and assessment procedures in place, then future updates to the HRA can be aligned with the 10 yearly "major review" frequencies of the ESC, although this will be kept under review.'

We have therefore updated our HRA as part of our major review of the ESC. This report documents that update.

2.1.4 Key Updates for 2026

Our technical position has evolved since the 2017 HRA was produced, although the fundamental assessment approach remains largely the same. The most significant changes include:

- addressing radionuclides in our HRA as well as non-radiological contaminants, in contrast to the previous HRA, where we only considered non-radiological contaminants;
- use of an inventory derived from the 2022 UKRWI, as opposed to the 2013 UKRWI used in the previous HRA;
- the inclusion of ILW within the inventory;
- an updated view of key contaminants, which reflects our evolving technical position since 2011 and the inclusion of ILW within the inventory;
- a revised repository design, which has been identified on the basis of extensive optimisation studies [10];
- a more formal assessment approach, as outlined in our 'Assessments Manual' [47];
- the use of a formal Engineering Performance Assessment (EPA) [13] to conceptualise and parameterise the evolution of barrier performance, and which provides a framework to aid the definition of assessment cases;
- a revised understanding of cap performance [13]: we expect a much better level of performance than we did in the previous HRA based on work carried out in our EPA;
- a revised understanding of near-field evolution: we expect drier and higher pH conditions than we did in the previous HRA;

- two different representations of the vault near field: one which treats the vault as well-mixed compartments with advective transport of contaminants through the grouted waste mass, and a second representation which explicitly represents stacks of waste containers and models diffusive transport of contaminants through grout;
- revisions to our hydrogeological understanding, although the high-level conceptual description of the hydrogeology remains unaltered.

2.2 Overview of Site Details

2.2.1 Site Setting

The LLWR is located on the coastal plain of West Cumbria near the village of Drigg and approximately 0.35 km from the Irish Sea coast at Drigg Beach (see Figure 2.1). It is approximately 3 km north of the Ravenglass Estuary where the Rivers Irt, Mite and Esk converge.

The site occupies an area of approximately 100 ha, of which approximately 40 ha form the existing waste repository. The topography surrounding the site varies from 25 m above Ordnance Datum (m OD) to the north-east and west of the site (Barn Scar Hill) to less than 5 m OD at the south-eastern site boundary. To the west of the site the topography gently undulates towards a small cliff line marking the edge of the Drigg Beach. Inland of the LLWR site the topography rises, initially gradually but then steeply, to be dominated by the Lakeland fells.

The broader coastline is characterised by a beach and inter-tidal zone with sand dunes and low cliffs inland. Most of the coastal plain consists of grassy fields with only a small amount of land dedicated to woodland and the growing of crops. Roads and minor tracks allow access to the coastal plain and to settlements that are located there.

Currently, the closest point of the LLWR vaults lies approximately 350 m inland from the present-day coastline and studies [8] have indicated that the site may be disrupted by coastal erosion or inundated as a result of sea-level rise in the future. The most likely outcome is considered to be disruption of the LLWR site by undercutting of the engineered structures within a timeframe of several hundreds to a few thousand years. The '*Site Evolution*' report [8] provides details of the expected evolution of the site and the underpinning studies that have been conducted.

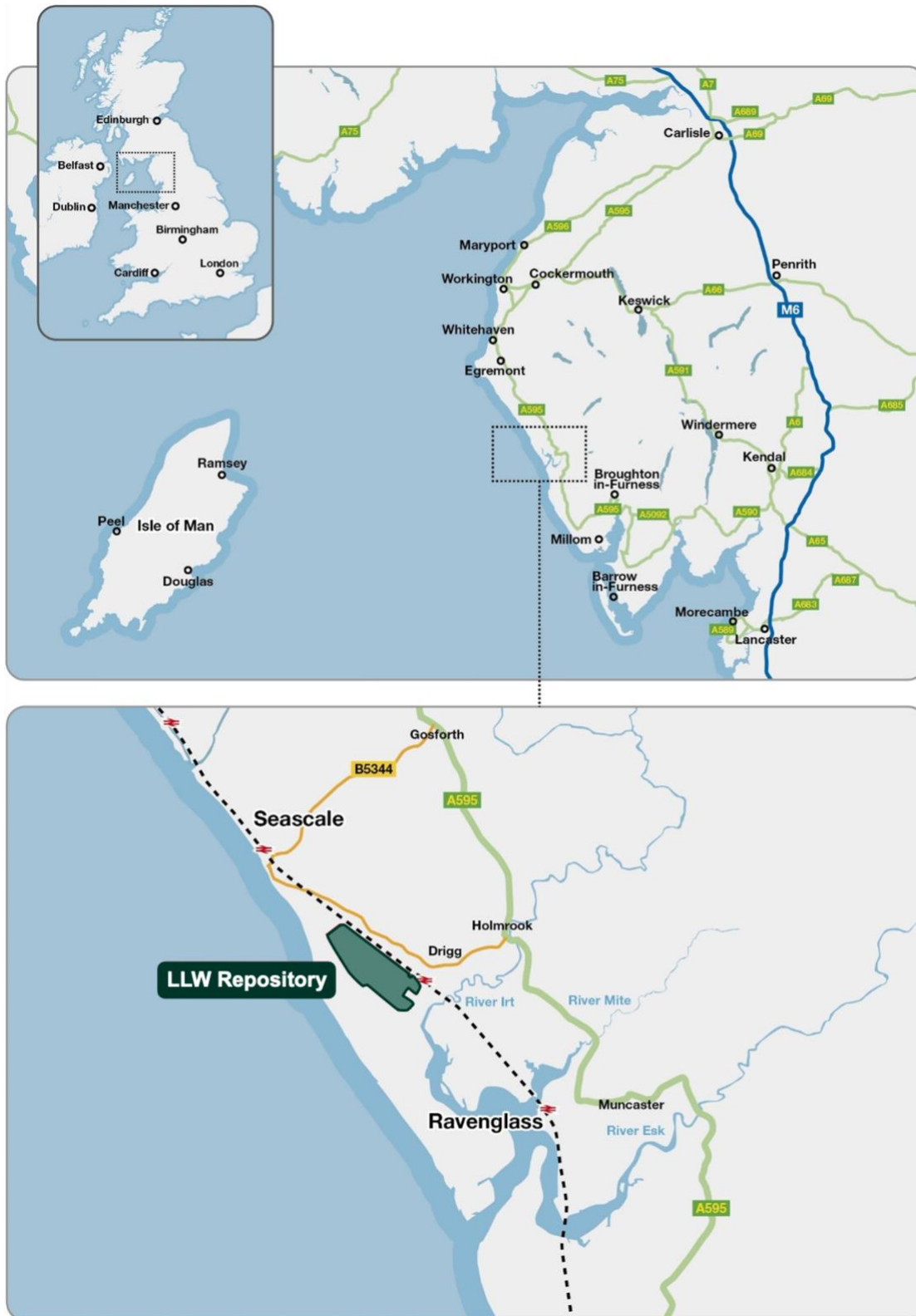


Figure 2.1: Location of the LLWR site

A railway line runs along the entire length of the north-eastern boundary of the site. Along the western boundary the site borders a Site of Special Scientific Interest (SSSI), known as the Drigg Coast SSSI. The site is principally surrounded by grazing land. Key features of the LLWR site are shown in Figure 2.2 and Figure 2.3.



Figure 2.2: Current layout of the LLWR site

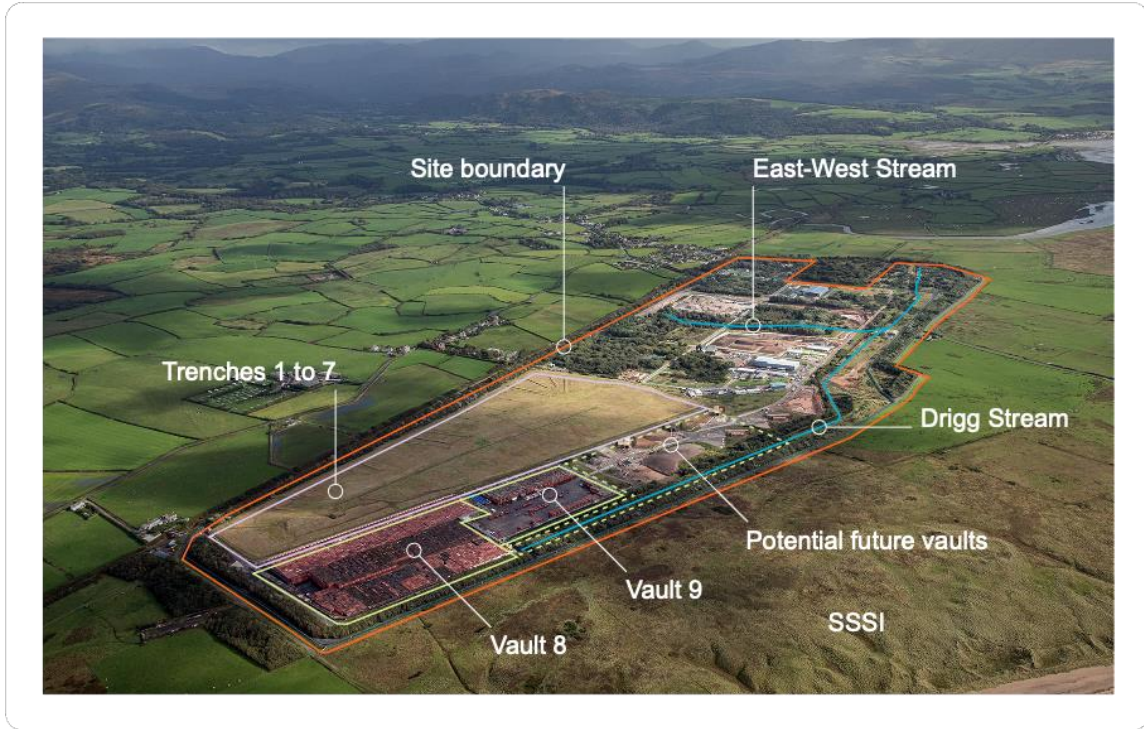


Figure 2.3: Aerial photograph of the LLWR

2.2.2 Surface Waters

The LLWR is located within the Irt-Mite-Esk- Annas Operational Catchment, which drains the western fells of the Lake District, though there are no WFD-protected surface water bodies in the vicinity of the site. The site is approximately 3 km from the Ravenglass Estuary. The

nearest named watercourse is the River Irt (see Figure 2.1), which flows approximately 500 m to the south-east of the south-eastern boundary of the site.

Surface water features either on, or in the immediate vicinity of, the site are shown in Figure 2.4. These are summarised as follows.

- The East-West Stream. This flows from north-east to south-west following a melt water channel. It is underlain by alluvium on the north side of the railway line and cohesive deposit (clay, silt, sand) as it tracks south towards the coast. The stream is culverted underneath the railway line. It is mostly supported by rainfall and run-off; however, there are localised contributions of groundwater discharge from deeper Quaternary deposits and bedrock (where mounding exists).
- The Drigg Stream. This was formed from the straightened lower reaches of the East-West Stream where it turned south. This was extended along the western boundary of the LLWR site. Drigg Stream receives flow from the East-West Stream and also from the north-west part of the LLWR site. The Drigg Stream flows from north-west to south-east, following the south site boundary, through the marsh areas and drains into the tidal stretch of the River Irt. Historically, the Drigg Stream used as a permitted discharge route for routine discharge of leachate from the facility. As a result of this practice, the stream sediments have become contaminated with very low levels of radionuclides. The Drigg Stream remains a permitted discharge route but would only be used in exceptionally high rainfall events. There are localised discharges of groundwater from the groundwater to the Drigg Stream.
- The railway drain (GF0004). This is alongside the railway line which runs through a cutting along the north-east site boundary and flows into the East-West Stream. The historical presence of elevated tritium levels in the railway drain indicates the presence of a pathway from the trenches to the railway drain. In order to try to prevent, or reduce, the migration of contaminants from the LLWR, an interim cap was constructed over the trenches and a cut-off wall was constructed. Current and historical monitoring data from the railway drain show that tritium levels have significantly reduced in the years following installation of the interim cap and cut-off wall [3, 48].
- The Marine Pipeline. This is a permitted discharge route for leachate from the trenches and vaults and other trace active drains. The pipeline discharges to the Irish Sea, about 1.2 km offshore.
- There is a series of ponds on the SSSI to the west of the site. These ponds are not supported by upwelling groundwater and therefore are not receptors for contaminated groundwater [49].
- As part of the cap construction work, a new Sustainable Urban Drainage System, including multiple lagoons to aid sediment removal, has been installed across the site to support surface water management relating to this work.

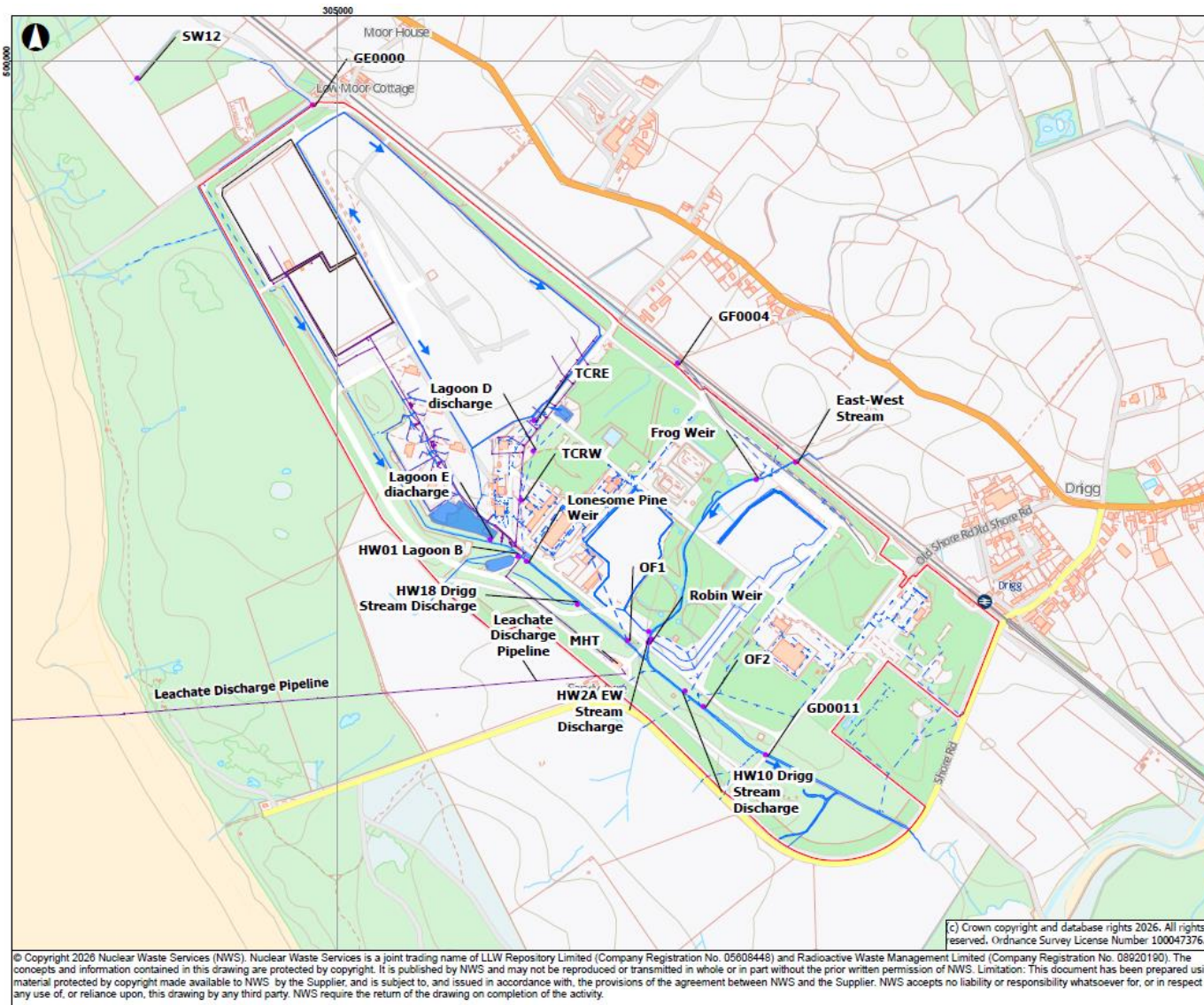


Figure 2.4: Streams, surface water drains and leachate drains near the LLWR

As shown in Figure 2.5, the majority of the LLWR is located within Flood Zone 1 (land assessed as having an annual probability of river or sea flooding of less than 10^{-3}). There is a very low risk that this area will flood from the surface water features listed above. A small area extending from the River Irt to the southern end of the site is classified as Flood Zone 3 (land assessed as having an annual probability of river flooding of more than 10^{-2} and an annual probability of flooding from the sea of more than 5×10^{-3}) [50].

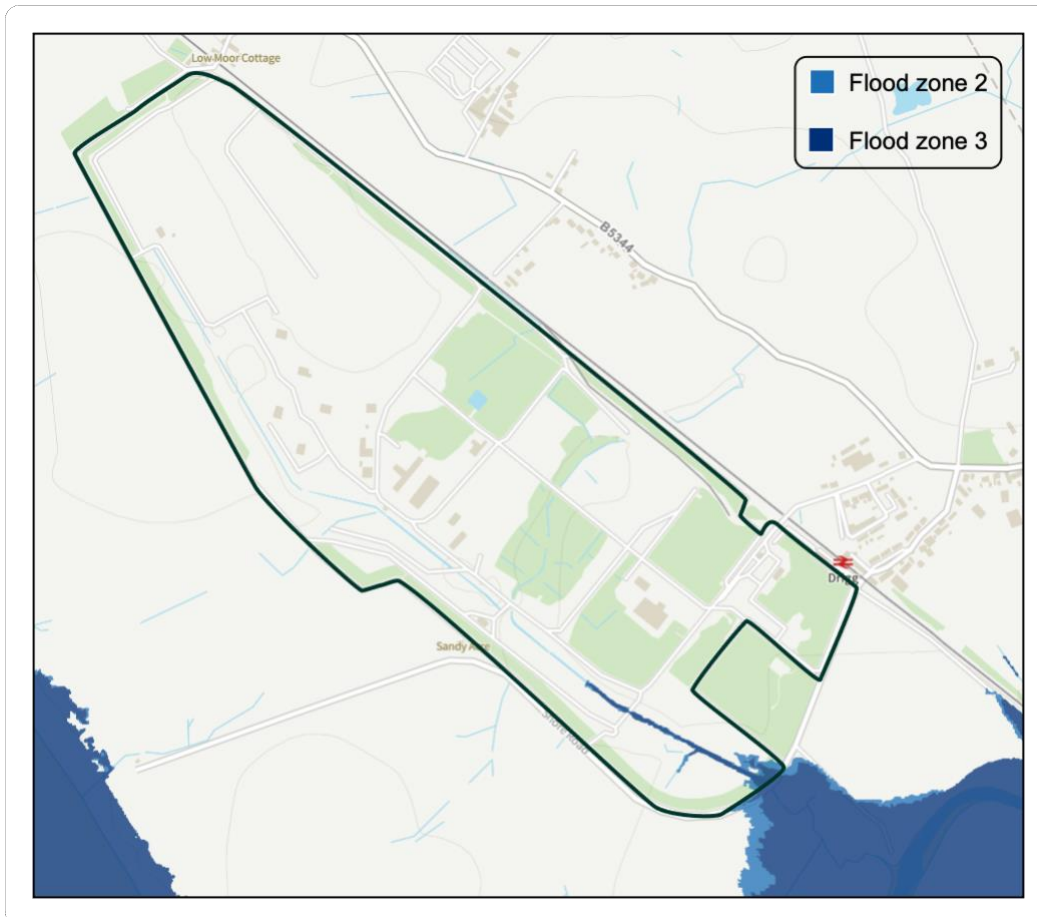


Figure 2.5: Flood risk areas at the LLWR [50]

2.2.3 Geology

The LLWR site and its surrounding area have been the subject of extensive surface and sub-surface investigation for decades (particularly since the 1990s). The sub-surface in the region consists of thick (up to 70m) Quaternary age (last 2.6 million years) deposits overlying bedrock. The Quaternary deposits are particularly important for the hydrogeology because the LLWR is a near-surface facility. A detailed review of the geology of the area and the development of the geological conceptual model is provided in our '*Hydrogeology*' report [7]. Recent investigations have contributed substantially to the current geological understanding of the formation of bedrock and Quaternary deposits in the region. A new geological conceptual model of the Quaternary deposits has been derived using an integrated stratigraphic methodology, applying multiple approaches, including an event stratigraphy and lithostratigraphy [51]. This allows better understanding of the geological complexity, processes and the connectivity of different units in areas where there are little data.

The LLWR site has been subject to glacial processes during the Quaternary period, which have given rise to a complex sequence of sedimentary deposits. The deposits consist, for the most part, of interbedded deposits of clayey diamicton, sandy diamicton, glacio-fluvial sands and gravels and glacio-lacustrine silts and clays. At the LLWR, the Quaternary deposits overlie Triassic sandstone of the Ormskirk Formation (around 240 million years old). The complexity of the Quaternary sediments has been revealed through the extensive investigations that have been undertaken at the LLWR site.

Made ground occurs across the majority of the LLWR as a result of site development. The thickness of the made ground is variable across the site and may, in some locations, be over 2.5 m. Made ground typically comprises a mixture of sand and clay with some gravel, representing reworked natural materials originating from within the LLWR, with occasional construction waste such as brick and concrete fragments.

2.2.4 Hydrogeology

The '*Hydrogeology*' report [7] provides a detailed review of the work done since the 2011 ESC in the developing the hydrogeological understanding. This includes the update of the hydrogeological conceptual model [52], to include recent developments in the geological model [51]. However, these developments have not changed the basic understanding of the groundwater flow. In the vicinity of the LLWR, groundwater generally flows sub-horizontally from the Lakeland fells towards the coast. Groundwater flow occurs within the Quaternary drift deposits and in the underlying Ormskirk Sandstone. Figure 2.6 shows a schematic representation of the hydrogeological conceptual model.

The Regional Groundwater occurs within the deeper Quaternary deposits and the underlying bedrock. It is distinguished from the Upper Groundwater by differences in the characteristics of the groundwater head. In the Regional Groundwater, there is not a significant vertical head gradient in the measured heads. Instead, there is a weak horizontal gradient that is generally perpendicular to the coastline. Flow is roughly from north-east to south-west, driven by the weak horizontal gradient. In areas where the upper part of the bedrock is sandstone, flow in the upper part of the bedrock makes a significant contribution to the regional groundwater flow.

Although the data suggest that it is possible to distinguish between the Upper Groundwater and the Regional Groundwater, they are not separate systems. Groundwater flows between them.

The flow is generally downwards in the Upper Groundwater, although, in places, the flow has a significant horizontal component. In discharge areas, which onshore are generally near streams, the flow has an upwards component. The flow in the Regional Groundwater is roughly horizontal, ultimately discharging into the sea.

There are localised discharges from the groundwater to the streams near the LLWR: the Drigg Stream and the East-West Stream. A component of the Regional Groundwater passing under the south-eastern part of the site discharges to the River Irt and the Ravenglass Estuary. Several engineered structures – including the northern section of the Drigg Stream,

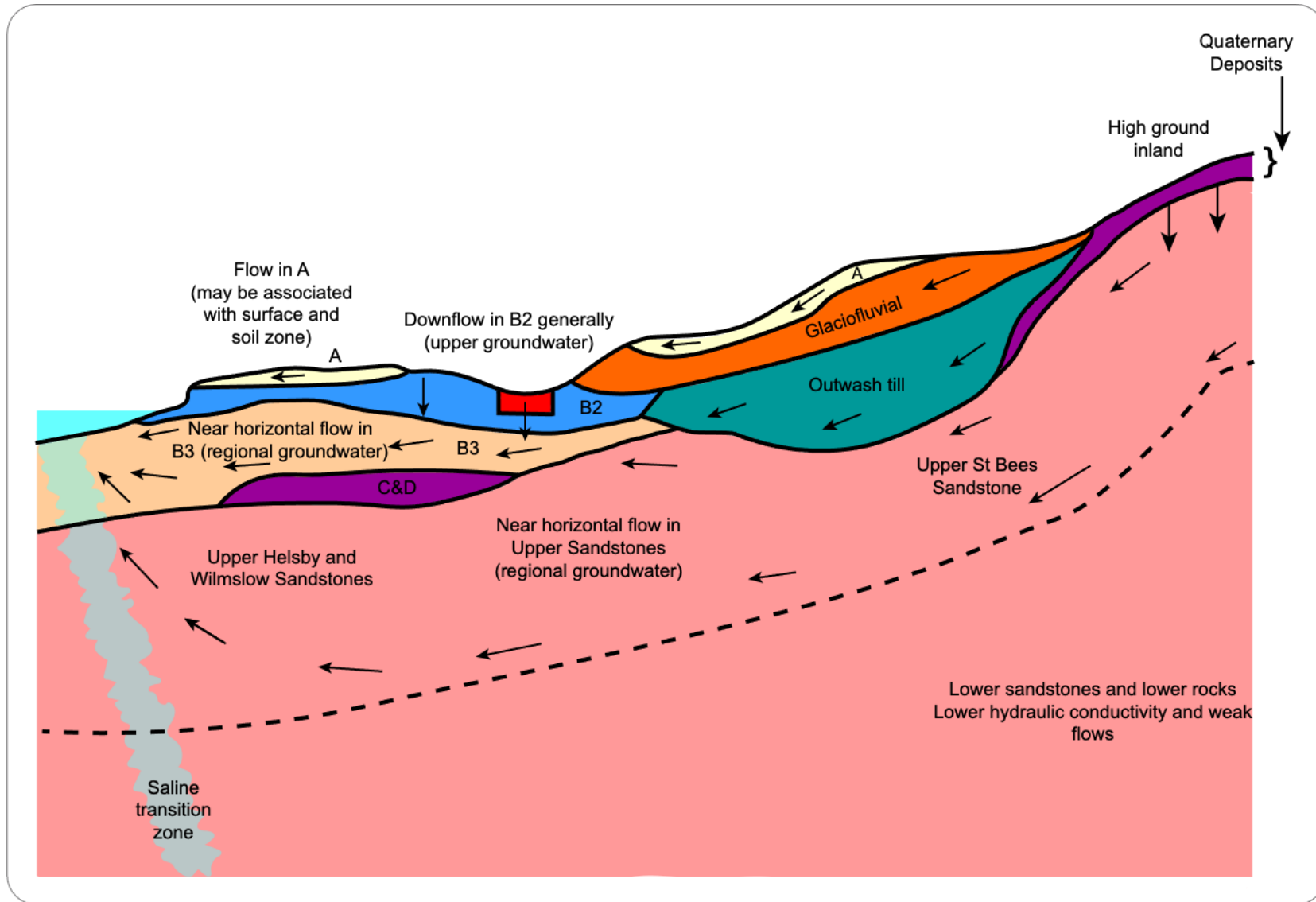


Figure 2.6: Hydrogeological conceptual model [52]

a railway line and associated drain (running in a cutting along the north-eastern site boundary), trench and vault engineering and a historical surface and sub-surface drainage system – all affect groundwater flow to a greater or lesser extent.

Measurements of the groundwater head in the vicinity of the site, and in monitoring wells with response zones at varying levels in the geological sequence, have allowed identification of the groundwater systems present at different depths. Based on the screen location the groundwater monitoring data is subdivided into the following categories:

- Upper Groundwater: is present within the upper Quaternary deposits (B2 in Figure 2.6) and overlies the Regional Groundwater. It is most evident in the north-west and central parts of the site, where it has a groundwater flow pattern that is distinct from that of the underlying Regional Groundwater, being predominantly downward, with very little lateral flow. Groundwater monitoring wells with screen bottoms above 6 m OD in shallow superficial deposits are considered to lie in the Upper Groundwater.
- Regional Groundwater: is observed in the basal Quaternary drift deposits (primarily in B3 in Figure 2.6) and within the underlying Ormskirk Sandstone. The groundwater flow direction in the Regional Groundwater at the LLWR is generally to the south-west. Groundwater from this zone discharges to the inter-tidal zone and further offshore. To allow further delineation of groundwater chemistry, the Regional Groundwater is split into two further subcategories, as follows:
 - Medium Groundwater: monitoring wells with screen tops between 6 m OD and above 10 m OD, which are screened in deep superficial deposits and the upper sandstone, are considered to lie in the Medium Groundwater.
 - Deep Groundwater: monitoring wells with screen tops below 10 m OD, which are screened in the sandstone aquifer, are considered to lie in the Deep Groundwater.
- Intermediate Groundwater: is not considered to be a distinct groundwater system but covers monitoring wells with screen depths crossing the Upper and Regional Groundwater boundary.

Towards the south-eastern boundary of the site, the direction of groundwater flow in the Upper Groundwater system merges with that in the Regional Groundwater system and, therefore, the two systems cannot be differentiated in this area.

Over time, global climate change is expected to lead to loss of part, or all, of the polar ice caps, leading to sea-level rise due to melt water from the ice caps. As a result, the sea level will rise near the LLWR. More details on the effects of climate change and coastal erosion are provided in the '*Site Evolution*' report [8]. There is large uncertainty about the magnitude and timing of sea-level rise and three climate scenarios have been proposed. The upper limit on the sea-level rise for the reference climate case is about 6 m and the upper limit on the sea-level rise for the high climate case is about 12 m [8]. Global climate change is expected

to lead to a pattern of warmer, wetter winters and warmer summers, compared with present day at the LLWR. This will affect the growth of vegetation and the recharge of water.

It is also expected that the coastline will recede as a result of erosion by the sea. It is considered that the initial disruption of the repository site – defined as the point when active coastal erosion first exposes or breaches the vault structures – could occur within a timeframe of 500 to 2,000 years After Present (AP). The complete erosion and removal of the repository materials from the site are estimated to take place between 1,500 and 4,000 years AP, with the specific timing dependent on the emissions scenario.

Sea-level rise and coastal erosion are not expected to change the broad features of the flow described above. As a result of sea-level rise, the head in the Regional Groundwater may rise (dependent on the accompanying climate conditions). The head drop across the Upper Groundwater may therefore reduce, leading to a reduction of vertical flow through the Upper Groundwater.

Groundwater quality and abstraction of groundwater

Groundwater in the Permo-Triassic sandstones in West Cumbria is classified as being a 'principal aquifer' [53] of overall good quality [54]. A principal aquifer is an aquifer that provides *significant quantities of drinking water, and water for business needs* [55].

Groundwater bodies in the Quaternary deposits are classed as 'secondary A' aquifers (typically sands and gravels) or 'secondary undifferentiated' aquifers (mixed deposits). A secondary undifferentiated aquifer is an aquifer for which it is not possible to class as either a secondary A aquifer (permeable layers, supporting local water supplies and providing base flow to rivers) or a secondary B aquifer (lower permeability layers with low yield and storage)

A water features survey carried out in 2023 [56], combined with information on private water supplies in the area was provided by Cumberland Council [57], indicated that there were no private water supplies and licensed abstractions in the vicinity of the site, and that the site does not lie within a source protection zone. The site is not within a Source Protection Zone (SPZ), a Drinking Water Safeguard Zone (Surface Water) or a Drinking Water Safeguard Zone [53].

There are no abstractions currently undertaken between the LLWR and the coast [58]. The designated SSSI is currently under a 999-year lease agreement to the NDA. This ensures that we have control over future development on the SSSI, including the installation of groundwater abstraction wells [58]. We have recently been granted a water abstraction licence [NW/074/0007/012] to allow the use of one of the on-site boreholes to supply water for dust suppression as part of cap construction works. This will be decommissioned before site closure.

2.2.5 Habitats and Designated Sites

Designated sites are protected sites and these include National Nature Reserves, SSSIs, Special Areas of Conservation (SAC), Special Protection Areas (SPA) and Marine Conservation Zones (MCZ). There are five designated sites in the vicinity of the LLWR site:

- Halsenna Moor SSSI;
- Drigg Holme SSSI;
- the Drigg Coast SSSI and SAC;
- the Morecambe Bay and Duddon Estuary SPA;
- the Cumbrian Coast MCZ.

These are shown in Figure 2.7 [59]. The SSSI habitats are referred to as a dune slack and are not groundwater dependent. The ponds on the SSSI are not supported by upwelling groundwater and therefore are not considered to be receptors for contaminated groundwater [49]. The inland sites, Halsenna Moor and Drigg Holme, are up-gradient of the LLWR and hence there is no credible means by which contaminants from the LLWR could migrate to these sites via groundwater or surface water. Consequently, none of the habitats within the designations are groundwater-dependent ecosystems¹⁵.

2.2.6 Past, Current and Future Operations

A comprehensive history of the site is presented in the '*Site History and Description*' report [3]. The site was first developed in 1940 as a ROF and manufactured trinitrotoluene (TNT) to support the war effort. The site was subsequently acquired by the United Kingdom Atomic Energy Authority (UKAEA), which in 1957 was granted authorisation to dispose of radioactive waste at the site. Disposal operations commenced in 1959. Photographic and map evidence from the early 1940s suggests that the principal use of the area, prior to the site being used as a ROF was for agriculture.

The first disposal trench (Trench 1) was formed along a cutting for one of the former site railway lines. As disposals continued and further disposal capacity was required, subsequent trenches (Trenches 2 to 7) were excavated adjacent to the original trench between 1959 and 1995. The trenches were cut into an underlying clay layer to form a low permeability base with bentonite added when clay was not present. A total volume of approximately 800,000 m³ of waste has been deposited in the trenches; this volume was reduced to less than 500,000 m³ as a result of compaction on disposal.

As discussed in Subsection 2.2.2, monitoring data indicated that tritium was migrating from the trenches to the nearby railway drain (GF0004). During 1988, work began on construction of a cut-off wall along the eastern perimeter of the trenches to intercept the migrating tritium. Subsequent monitoring after the installation of the cut-off wall indicated a significant reduction in tritium at GF0004. Tritium plays a key role in site characterisation and understanding at the LLWR due to the fact that it does not sorb to the surrounding media, thus moving through the geosphere at the same rate as the groundwater in which it is transported.

¹⁵ Groundwater-dependent terrestrial ecosystems are wetlands which critically depend on groundwater flows and, or chemistries and are safeguarded by the WFD.

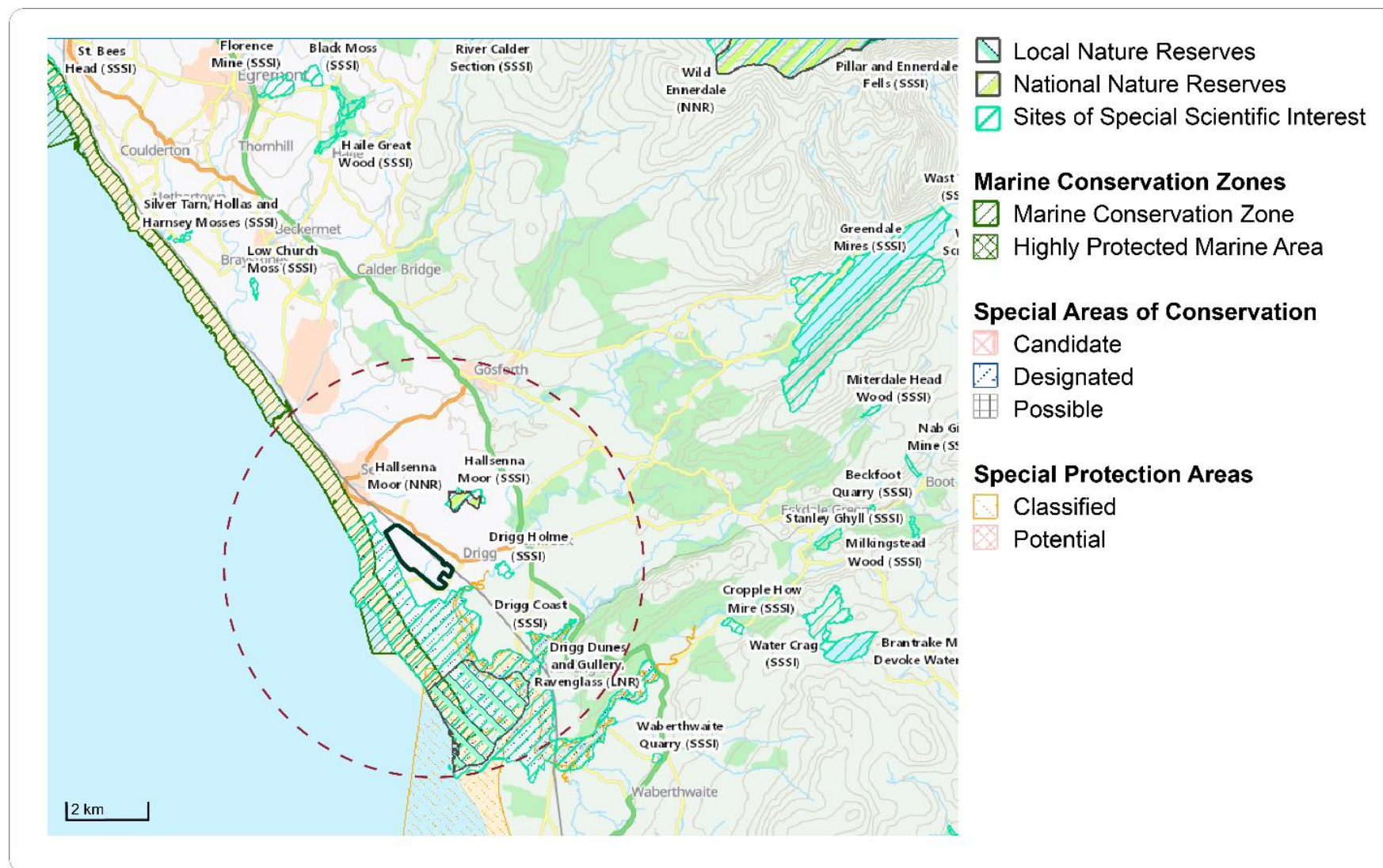


Figure 2.7: LLWR site with habitats and protected sites. The red dashed circle indicates a 5 km radius circle centred on the LLWR site (MAGIC used permission from Natural England) [59]

Disposals of containerised waste to Vault 8 commenced in 1988, with disposals to Trench 7 ceasing in 1995. The trenches have been capped with an interim cap, constructed in 1988 and extended in 1995, to limit infiltration of rainwater and prevent intrusion into the waste. Vault 8 was mostly filled to its original planned capacity by 2009, although final disposals of waste to Vault 8 will take place in 2026.

Construction of Vault 9 began in 2008 and was completed in 2010 with a total capacity of 247,000 m³. Interim storage of waste in Vault 9 began in July 2009, before Vault 9 was finally completed. A revised Permit [43] issued in November 2015 (and subsequent issues 2020 [60], 2021 [61] and most recently 2025 [62]) and planning permission (granted July 2016) now allow for the disposal of waste in Vault 9.

Construction of additional vaults is planned, potentially up to Vault 12 based on current inventory projections although designs extend to Vault 14 to retain flexibility, with waste emplacement ceasing in 2135 [4].

A multi-layer final cap will be placed over the trenches and vaults. It is anticipated that this will be done progressively with cap construction occurring over the same period as construction of additional vaults. The current design of the final cap will include a region of profiling fill, a gas collection layer, a low permeability geomembrane, a gravel drainage layer, a bio-intrusion barrier consisting of large cobbles, a low permeability clay layer, a filter layer comprising sand and coarse gravel and subsoil and topsoil cover. The cap is intended to reduce infiltration and discourage human intrusion. Construction of the final cap over Vault 8 and the northern part of the trenches is expected to commence in 2029. The cut-off wall will also be extended around the disposal area prior to cap construction. Details of the engineering are provided in our '*Engineering Design*' report [5].

We assume, based upon engineering judgement, that a further seven years will be required to install the final elements of the closure engineering, and therefore the site will close in the early-2140s [10].

After site closure, a period of institutional control will begin, during which monitoring will continue. We have reviewed the basis for the length of the period of institutional control as part of work to develop the 2026 ESC [63]. We have concluded, on the basis of that work, that the period of active institutional control will be for 100 years after the installation of the closure engineering, and will end in the early-2240s.

3 Disposal System

The LLWR differs from a landfill in terms of both the waste and the engineered barriers to contaminant release. The facility receives waste from a range of consignors, including nuclear power stations, fuel cycle facilities, defence establishments, general industry, isotope manufacture sites, hospitals, universities and from the clean-up of historically contaminated sites.

Wastes are accepted against WAC [64]. These were derived on the basis of the 2011 ESC and subsequent work undertaken [65, 66, 39] to support an application to vary the Permit [62]. The WAC take into account the potential environmental impacts of radionuclides and non-radiological contaminants in the waste. We detail our approach to revising our WAC on the basis of the 2026 ESC in our '*Implementation*' report [17], while we discuss the waste inventory in Subsection 3.1.

Engineered features at the LLWR site have been optimised during design to provide a multi-barrier approach to minimising environmental impacts from releases of contaminants from the waste [10]. Due to the nature of the LLWR, these features have in many cases been primarily designed to minimise radiological risks from the site. However, it will be demonstrated in this report that they also provide adequate protection against non-radiological hazards and pollution. The various engineered barriers to contaminant release are described in detail elsewhere [5], and the key elements are summarised in Subsections 3.2 to 3.6.

The optimised engineered measures we have identified form part of an optimised Site Development Plan, which we report in reference [10]. The Site Development Plan outlines our view of how the LLWR site will be developed and accommodates currently disposed and emplaced waste, plus estimated future arisings of LLW in the UK. We have considered how the Site Development Plan could be modified to allow the LLWR to accept the disposal of some ILW. The Site Development Plan provides the basis for quantitative modelling and assessments of environmental safety as presented in this and other ESC reports.

3.1 Inventory

Once filled, the trenches and vaults will contain a wide range of waste materials in a variety of forms. These will potentially pose a radiological risk, due to the presence of radioactive materials, and a non-radiological risk from materials that are not radioactive but which nevertheless exhibit a degree of toxicity to humans or to the environment.

The methodology we have used to derive the reference inventory used in the 2026 ESC is described in our '*Disposal Facility Inventory*' [4] report. The reference inventory was developed in stages through iteration with assessments. This is because the disposable inventory is constrained by the radiological capacities and stream constraints derived from

the assessments in the ESC.¹⁶ These assessments have been progressively developed in two phases. As we developed the assessments, we learned more about the radiological constraints and controls that we should apply to the inventory. Three stages of inventory development were carried out during development of the ESC [4]:

- Stage 1: inventory developed using capacities and stream constraints derived from work carried out to support NDA strategy;
- Stage 2: inventory developed with constraints as per Stage 1, but with some targeted improvements arising from early learning in Phase 1 assessments;
- Stage 3: inventory developed using constraints that were generally derived from Phase 1 assessments, cross-checked with early learning from Phase 2 assessments.

The assessments reported in this document are based on the Stage 2 inventory, which is presented in reference [67]. The Stage 2 inventory provides a suitable basis to assess the performance of the repository.

The reference inventory includes:

- wastes disposed of in the trenches between 1959 and 1995;
- wastes disposed of in Vault 8 and consigned to Vault 9 prior to 31st March 2022;
- a forward inventory, including arisings profile, of waste for disposal derived from the 2022 UKRWI.

For each of these, the inventory includes information on:

- the quantity of waste disposed of, which informs our understanding of how many vaults may be required to accommodate the waste;
- the quantity of bulk materials disposed of, which informs our understanding of chemical conditions within the repository;
- the quantity of non-radiological contaminants present in the wastes;
- the radioactivity present in the wastes.

The forward inventory includes LLW declared in the 2022 UKRWI. It also includes some ILW which may be suitable, based upon radiological assessment calculations, for disposal in a

¹⁶ Assessment calculations have been used to derive limits on the total activity which could be disposed of in the repository and the stream constraints, which are maximum activity concentrations in the wastes. These limits and stream constraints ensure that radiological impacts from the disposed wastes will be consistent with relevant regulatory standards. In addition to meeting these radiological constraints, waste would only be accepted for disposal if it complied with WAC, which would cover a full range of environmental and operational safety considerations. We have not assessed waste streams against WAC when deriving the reference inventory. Therefore, inclusion of wastes in our reference inventory does not imply that a waste stream would comply with WAC and be acceptable for disposal. Our approach to developing WAC is presented in our '*Implementation*' report.

near-surface facility.¹⁷ This does not imply that a decision has been taken to accept ILW for disposal at the LLWR.

There is significant uncertainty around the volume of LLW that will require disposal at the LLWR. Waste receipts are significantly lower than previous UKRWI forecasts. The forward inventory volume for LLW is considered to be a best estimate based on the waste currently expected to be routed to the LLWR according to the 2022 UKRWI. Diversion of wastes that are planned to be managed by other routes (e.g. treatment or landfill) has been taken into account.

As discussed in reference [4], the quantity of ILW that is potentially suitable for disposal at the LLWR exceeds the radiological capacity of the repository. We have therefore had to develop an approach, and appropriate assumptions, to derive an inventory of ILW. The approach and assumptions are presented in reference [4].

In this approach, we have assessed the potential suitability of waste streams for disposal based on waste stream data provided in the 2022 UKRWI. This assessment has not considered variability within a waste stream, which means that:

- portions of waste streams judged to be acceptable radiologically may not be acceptable;
- similarly, portions of waste streams judged to be unacceptable radiologically may be acceptable.

As decommissioning plans mature at waste producer sites, characterisation of the ILW will improve. This may change the understanding of the number of waste streams, as well as the volume and activity of the waste streams, which will be reflected in subsequent issues of the UKRWI.

In addition to complying with the constraints derived from assessment calculations, waste would only be accepted for disposal if it complied with WAC, which would cover a full range of environmental and operational considerations. We have not considered whether ILW waste streams would comply with other aspects of the WAC and we have not considered whether the wastes could be transported to the LLWR.

These considerations mean that there is significant uncertainty around the ILW inventory and the wastes which are accepted for disposal may be different to those we consider within the reference inventory. Notwithstanding these uncertainties, our view is that the inventory provides a suitable basis to explore the disposal of ILW at the LLWR.

There is also significant uncertainty around the inventory of non-radiological contaminants in LLW and ILW. Historically, characterisation of non-radiological contaminants has not received as much attention as the radionuclide inventory. Efforts are ongoing across the nuclear industry to improve the quality and quantity of information for non-radiological

¹⁷ That is, the ILW streams do not result in exceedances of the repository's radiological capacity, or contravene the stream constraints.

contaminants for future wastes. Since 2016, we have requested data on non-radiological contaminants from consignors. Data collection has improved over time, but the mass amounts in the disposed inventory are not well understood, particularly for disposals prior to 2016.

Our approach to the disposed inventory is as follows.

- Our understanding of the trench inventory is presented in reference [4], where we note that there is limited information on the inventory of non-radiological contaminants, with good estimates only available for boron and lead. We do, however, have a confident understanding of the bulk materials disposed of in the trenches, and we apply reasonable assumptions about the composition of those materials to derive contaminant masses. This approach is presented in more detail in reference [25].
- For the vaults, for wastes disposed of prior to 2016, we use the quantities of bulk materials disposed of, along with assumptions about the composition of those materials, to derive contaminant masses. This is the same approach as we use for the trenches. For wastes disposed of after 2016, the non-radiological inventory is based upon consignor information about the bulk materials disposed of in the vaults and the quantities of declared non-radiological contaminants present in the wastes.

For the future inventory, we calculate the masses of non-radiological contaminants based on the quantities of non-radiological contaminants and the quantities of bulk materials reported in the 2022 UKRWI.

Our approach to treating uncertainty in the future inventory is to derive appropriate radiological and non-radiological controls and capacities which we will use to manage waste acceptance and ensure that radiological and non-radiological impacts are less than relevant regulatory guidance levels. These capacities do not require knowledge of the contaminants present in the inventory. This is because they are calculated using our assessment models based upon an assumed initial mass of contaminant present.

The assessment models are parameterised based upon our understanding of chemical conditions in the repository. These are discussed in our '*Near Field*' report [6]. Chemical conditions in the repository are dictated largely by the quantities of mild steel (present in the form of waste containers) and grout in the repository [6]. Uncertainties in the material composition of the wastes therefore have only a very minor effect on the evolution of chemical conditions. This means that our assessment model parameterisation and our calculated capacities are robust with respect to uncertainties in inventory and provide an appropriate basis on which to control acceptance of waste.

3.1.1 Identification of Contaminants of Interest

The inventory presented in reference [4] contains information on 153 radionuclides. Only a subset of these radionuclides will contribute significantly to the radiological impacts and therefore warrant inclusion in the groundwater assessment. We have developed an

approach to identify the radiologically most significant radionuclides, which is presented in reference [15].

The non-radiological contaminants to be considered in the HRA are presented in Subsection 4.4. Given a mass of each contaminant, our assessment model, which we discuss in more detail in Section 4, calculates the concentration of non-radiological contaminants in groundwater. The mass of each contaminant of interest is derived from the inventory presented in reference [67] using a methodology discussed in reference [25].

3.2 Trench Disposals

Between 1959 and 1995, disposal was by tumble-tipping into a series of seven parallel trenches, as shown in Figure 3.1. As one trench reached full capacity, another was excavated. The base of each trench was excavated, where possible, into a clay-rich layer of the natural geology at a depth of 5 to 8 m below ground level. Where the natural clay layers were incomplete or absent at the excavated depth, imported bentonite was often used rotovated into the trench base.

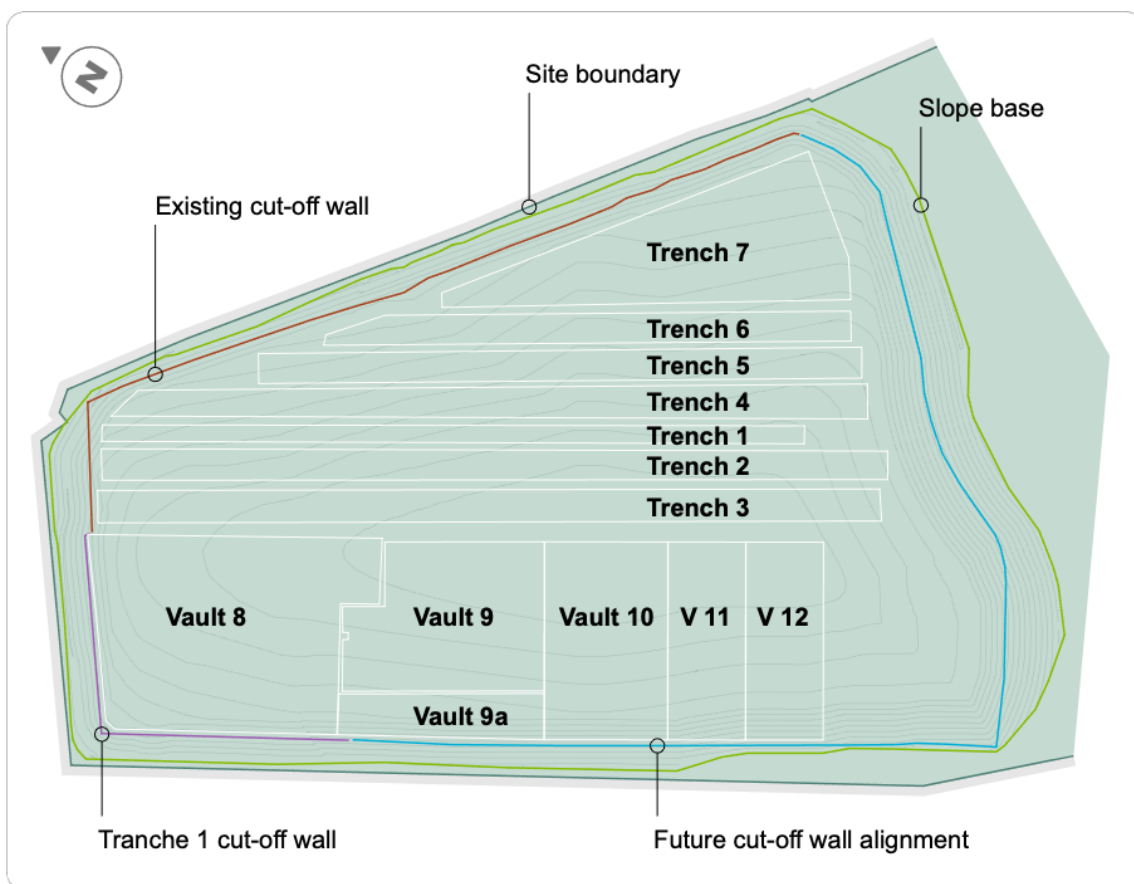


Figure 3.1: Layout of the trenches and vaults at the LLWR

A total of around 800,000 m³ of wastes were disposed to the trenches, occupying a volume of approximately 500,000 m³ in situ [5] as a result of degradation and compaction. The wastes were untreated and generally loose tipped. There was routine daily cover using local

materials excavated from the trenches. These materials varied in composition but typically comprised sandy clays for most of the disposed material.

Leachate drains were constructed at the bottom of the trenches and leachate is collected from the southern end of each trench and discharged to sea via the Marine Holding Tanks (MHT). Further discussion of leachate management is presented in Subsection 3.4.

A cut-off wall was installed from the north-east corner of Trench 8 to the southern end of Trench 7 between late 1989 and 1995 to limit the potential for lateral migration of leachate from the trenches to the adjacent railway cutting. In addition, an interim cap, extending over the top of the cut-off wall, was completed over Trenches 1 to 6 in 1989 with the following objectives:

- to isolate the waste from the near-surface environment, limiting the possibility of disturbance and protecting the waste during the initial phases of waste settlement;
- to control the release of gas from the trench disposal area;
- to limit the infiltration of rainwater into the disposal area and, hence, the volume of leachate generated.

In combination with the interim cap, the cut-off wall also supports the minimisation of leachate production by controlling the lateral migration of meteoric groundwater into the trenches.

The interim cap was subsequently extended to cover Trench 7, following completion of trench disposal operations in 1995. In association with these post-emplacement actions, the disposal of loose wastes was phased out, and was replaced by the emplacement of compacted, containerised and grouted wastes in engineered concrete vaults, as described in the following subsection.

Considerable effort has been expended in the past to determine whether any remediation of the trenches is required [68, 69]. We have considered a wide variety of actions that were identified as having the potential to reduce the radiological risk associated with past disposals to the LLWR. These options were assessed from the perspective of the potential magnitude of reductions in dose and risk they may be capable of delivering, set against the wider implications that would be associated with their implementation. Consistent with the underlying objective of continuing to provide capacity for the disposal of LLW at the facility, the emphasis in the options analysis was on the appropriateness of actions that would 'target' the retrieval of wastes from, or the implementation of other types of remedial action on, specific areas of the trenches. These are where localised high concentrations of key radionuclides are present that play a significant role in determining overall impacts from the facility. Our options assessments demonstrate that the achievable scale of risk reduction (below what is already a low risk by comparison with the regulatory guidance level) is small compared with the costs and disruption that would necessarily be associated with targeted retrieval or remediation. We have not therefore taken forward any plan for targeted retrievals [10].

Recent work identified that the interim trench cap was not performing as well as intended. We discuss the optimised approach to upgrade of the interim trench cap in Subsection 3.4.1.

3.3 Vault Disposals

Our inventory analysis [4] indicates that we are required to construct Vaults 9a, 10, 11 and 12 to accommodate the volume of waste. We recognise, however, that there is uncertainty around the potential future inventory [4] and that it is advantageous, therefore, to retain the flexibility to respond to changes in inventory understanding. Our reference design, therefore, includes disposal capacity up to Vault 14.

3.3.1 Vault 8

Waste form

The majority of disposals consigned to Vault 8 comprise wastes grouted into mild steel ISO-freight containers. Some items consigned to the vault were too large to be disposed in ISO-freight containers. These items were placed into prepared areas of the vault and mostly grouted in situ, both internally within the waste to eliminate voidage and also externally around the wastes, using mobile grouting facilities.

Although the cementitious grout used was originally intended to fill voids, it will provide other long-term safety benefits relevant to the HRA:

- the grout will ensure that high pH conditions are established in the vault after final capping, which will promote sorption and low solubility conditions for certain key contaminants;
- the grout will provide a substrate for contaminant sorption.

The grout therefore has an important role in reducing releases of contaminants from the repository to groundwater.

Voids within the vault will be filled with free-draining, inert, low-fines granular material that will be emplaced between container stacks – wherever the stacks are wide enough accept such material – before installation of the engineered cap.

The void fill will be emplaced to support cap stability, to provide preferential flow-paths for waters around the waste containers, and to provide lateral constraints across the vaults to ensure the continual alignment of stacks and the associated vertical drainage routes between the stacks. This will help ensure redundancy of vertical drainage routes across the vault. This will ensure that infiltrating water is able to flow to the vault base and prevent near-surface discharges of leachate from the facility.

The majority of containers in Vault 8 are stacked four-high, with a small number of six-high stacks. Our understanding of voidage and container performance [10] indicates that:

- we cannot substantiate 'higher stacking' of containers on the top of these existing stacks;

- the containers will deform under closure loads.

In light of this understanding, we have undertaken extensive studies to determine an optimised approach to closure of Vault 8. We have concluded [10] that we will subject the wastes in Vault 8 to surcharge. That is, the containers will be loaded such that container deformation is expressed prior to installation of the final cap, rather than during its installation. This will provide a stable base for construction of the cap, which is essential for confidence in cap constructability and confidence in its long-term performance.

Some container damage is expected during the surcharging process, which will reduce the contribution of the containers to the multi-barrier system. They will still, however, provide some containment, as the granular backfill between stacks will constrain the side panels of the containers, ensuring they remain in place. Given the expected level of performance of the final cap, the effect on repository performance will only be very minor. As the containers are not considered in our assessments of the groundwater pathway, there is no effect on assessed impacts.

Engineering design

The design and construction of Vault 8, in 1986 and 1987, were to contemporary standards and best practice. Vault 8 has a depth of about 7.7 m, an east-west length of about 177 m and a north-south length of about 227 m, leading to a vault volume of about 308,000 m³. A plan of the vault is shown in Figure 3.2.

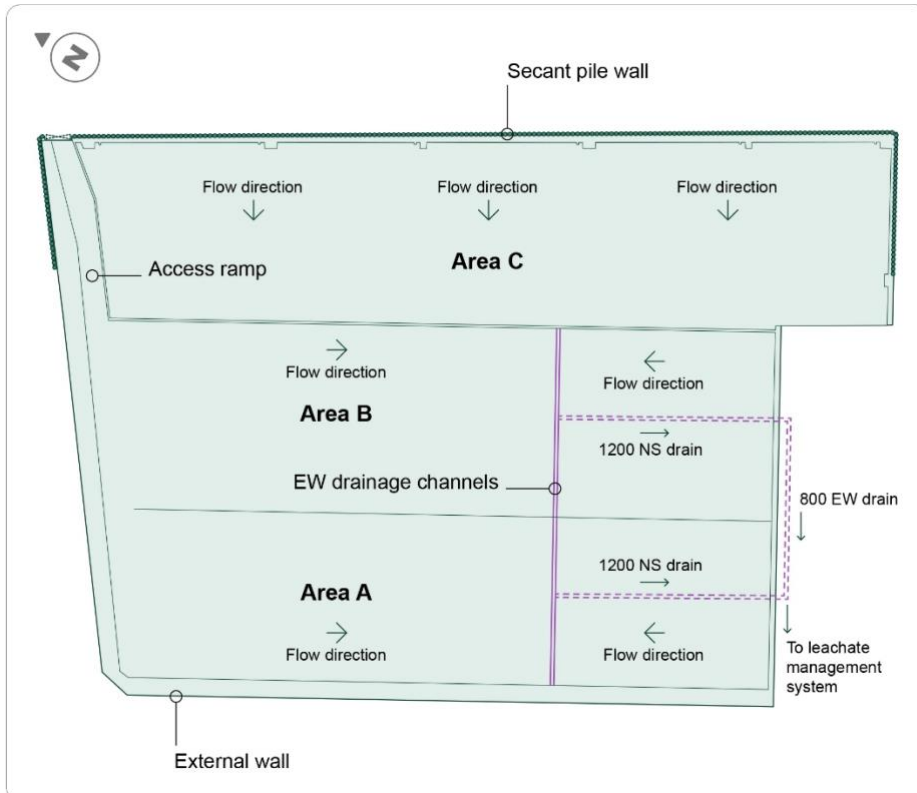


Figure 3.2: Schematic plan of Vault 8

It is a concrete-lined vault, comprising reinforced concrete walls and base slab. A secant pile wall (see Figure 3.2) provides structural support along the eastern side of the vault, adjacent to the trenches.

The reinforced concrete base slab was formed in three bays, with the first stage, western bay Area A 250 mm thick. The middle bay, Area B, and eastern bay, Area C, have base slabs 300 mm thick. There are reinforced concrete walls between the bays for operational purposes; these walls are typically 600 mm thick and to the whole height of the vault [5]. The exterior north and west walls are of reinforced concrete 350 mm thick.

The base was cast in sections with joints as it was only intended to provide an operational running surface for disposal operations and was not intended to function as a hydraulic barrier. It was expected that some cracking would occur upon loading by disposals, and some small cracking has been observed [5].

As the vault base slab was not intended to act as a barrier to water flows, it was constructed on top of a minimum of 1 m of clay or 300 mm of bentonite-enhanced sand (BES).¹⁸ This low permeability mineral layer provides a barrier to release of leachate from the vault to

¹⁸ It was originally intended that existing underlying clay would provide a low permeability layer. The clay layers were, however, found to be intermittent when the vault was constructed. Where the clay was insufficient, or of low quality, BES was emplaced instead under the base slab to provide the low permeability layer. There is no reliance placed upon the low permeability of existing ground materials for Vault 9 or any of the future vaults.

groundwater. Assessments provide confidence that the low permeability mineral layer will remain intact and functional after installation of the final cap over Vault 8 [5].

3.3.2 Vault 9 and 9a

Waste form

Vaults 9 and 9a will contain two distinct populations of LLW, which are summarised below.

- **Existing and committed LLW.** This is LLW that is either present in the vault now or will be consigned to the vault in an existing ISO-freight container design. Similarly to the Vault 8 wastes, these containers will be insufficiently strong to withstand the loads that will be imposed during installation of the final cap. We have determined that the optimised approach to managing these containers is to subject them to surcharge [10]. As with Vault 8, this will provide a stable base for cap installation, with associated benefits for cap constructability and confidence in long-term cap performance, albeit at the expense of a minor reduction in the level of containment provided. These containers will be placed at the northern end of Vaults 9 and 9a, and covered with a cap seal as soon as is practicable after closure of Vault 8 to ensure that post-closure conditions are established.
- **Future LLW.** This is LLW for which there is the opportunity to change the disposal container. Our optimised approach to these wastes will be to dispose of them in a modified ISO-freight container design which has greater load-bearing capacity than the current ISO-freight container [10]. The container will be strong enough to support the load of the overlying cap and profile fill and to allow higher stacking. The uppermost container of each stack will be fitted with a reinforced concrete container protection unit (see Figure 3.3). The container protection unit will transfer the loads of the overlying cap and profile fill to the load-bearing elements of the container, thus protecting the weak container lid from damage. To ensure that the contribution of these containers to the multi-barrier system is preserved, they will be stored in temporary warehouses prior to emplacement in their final disposal locations.

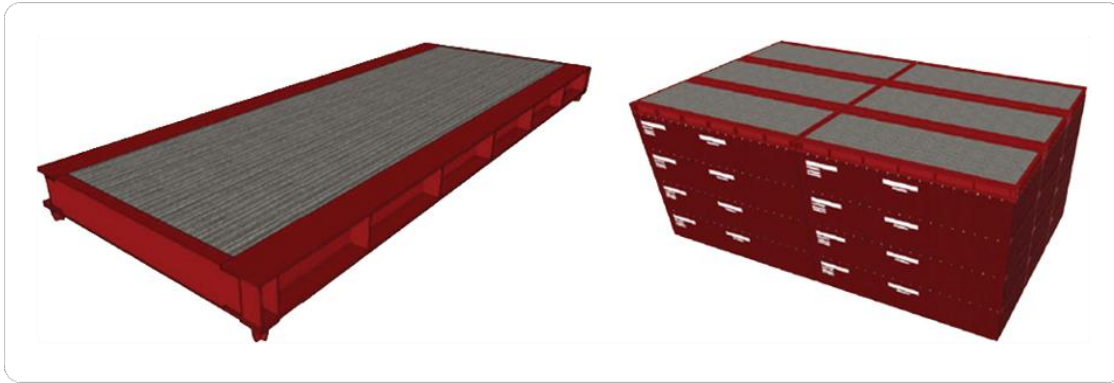


Figure 3.3: Illustration of the container protection unit approach for future LLW

Wastes in both categories will be grouted into the containers. As with Vault 8 wastes, the cementitious grout will reduce voidage, as well as contributing to retarding contaminant release from the repository.

If ILW were to be disposed at the LLWR, then Vaults 9 and 9a would also be used to dispose of 'ILW that can be managed as LLW'. This is ILW that leads to external dose rates similar to those from LLW, such that additional arrangements such as in-vault shielding would not be required to manage the waste.

For the ESC, we have assumed that this waste would be disposed of in a small, strong container which could be transported to the site in a Standard Waste Transport Container (SWTC) if required. These SWTC-compatible strong boxes would be sufficiently strong to withstand the loads imposed during, and after, installation of the final cap. They would also be capable of being higher stacked. The topmost container of each stack would be fitted with a container protection unit to prevent the weak lid of the container being damaged by the load of the cap.

These containers would be disposed of in the vaults alongside strengthened ISO-freight containers, although differences in container sizes would mean that they would not be co-located in the same stacks. To protect the containers from environmental effects including precipitation and wind-blown salt, they would be stored alongside strengthened ISO-freight containers in temporary warehouses prior to emplacement in their final disposal locations.

A cementitious grout would be used to fill voids within the containers. The grout would provide the same safety functions as the grout in the LLW containers.

As with Vault 8, the voids between stacks will be filled with an inert free-draining material.

Engineering

Vault 9 was constructed between 2008 and 2010. In 2009, it was partly commissioned to take some waste for storage only, with full commissioning in December 2010, again for storage only. The Permit issued in November 2015 [43] (and subsequent issues 2020 [60], 2021 [61] and most recently 2025 [62]) and planning permission (granted July 2016) allow for the disposal of waste in Vault 9. The dimensions of Vault 9 are approximately 174 m

north-south by 177 m east-west, by around 9 m depth. This leads to a vault volume of about 270,000 m³. A schematic view is shown in Figure 3.4.

A secant pile wall provides structural support along the eastern side of Vault 9, adjacent to the trenches. The secant pile wall was not designed to be water-retaining. It is not relied on to provide an impermeable barrier but will remain present in the long term.

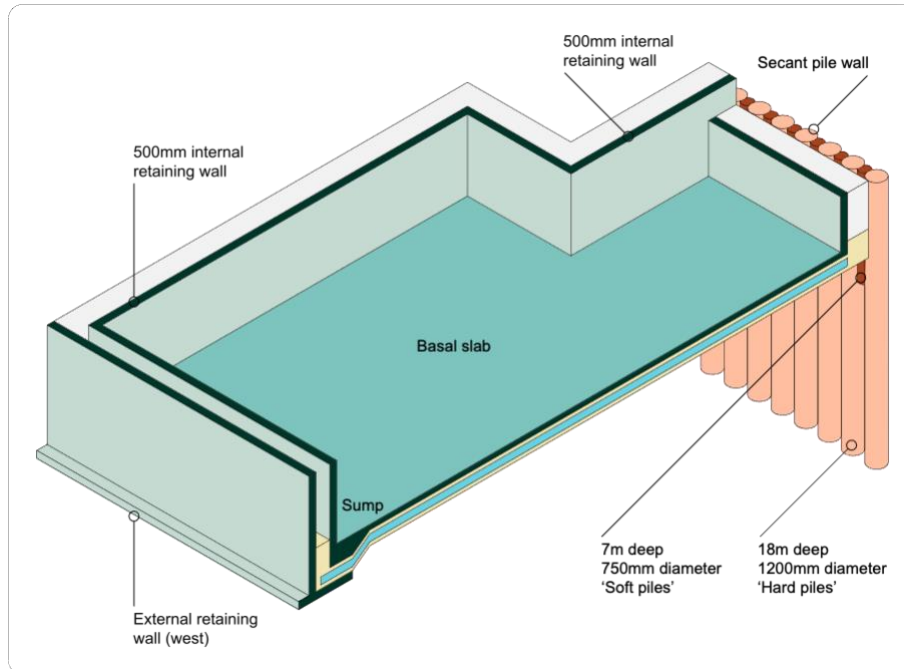


Figure 3.4: Schematic view of Vault 9 (not to scale)

The Vault 9 base consists of a reinforced concrete slab, underlain by a double composite liner system. These are shown in Figure 3.5. The concrete slab is primarily intended to be a running surface for waste disposal operations. While it was not designed to be impermeable, it will nonetheless provide a significant barrier at early times, although it will degrade over time. The composite liners underneath the basal slab will prevent the release of leachate to groundwater during the operational period and will continue to provide a barrier in the long term, albeit one which will degrade.

Vault 9 incorporates a leachate detection and collection layer beneath the upper and lower composite liner systems.¹⁹ The intent was that leachate (if any) penetrating the upper composite liner during operations and active control stages would be collected and monitored. The intent was to provide confidence in the performance of the system. The optimised design does not require this layer for the future vaults.

¹⁹ The leachate detection and collection layer has been 'placed out of service'.

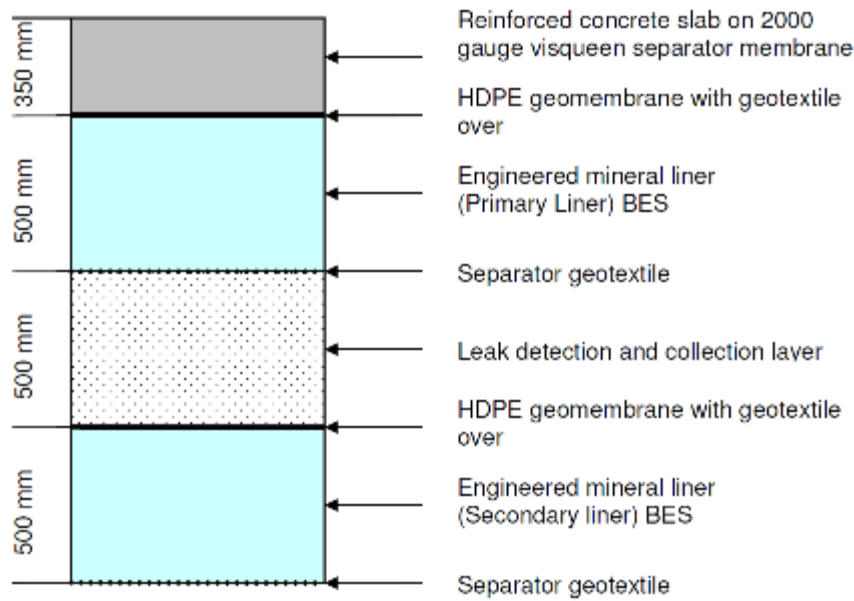


Figure 3.5: Structure of Vault 9 composite liner

The perimeter west and internal north walls of Vault 9 are comprised of double-leaf reinforced concrete walls each 500 mm thick. The east wall is similar, but with the reinforced concrete secant pile wall providing the outer wall.

The double leaf walls are separated by 1 m or more of infill with BES to provide an internal liner of low permeability. The BES and geomembrane extend up the perimeter walls, providing continuity with the basal liner, up to a height of 1 m above the base slab, where they are capped with concrete. The existing void space at and above this level will be infilled with free-draining stone and penetrations will be engineered into the vault walls at this level. These measures will allow the routing of leachate to the drainage blankets of future vaults and thence into the underlying geology, which will promote unsaturated conditions in the waste and minimise the potential for near-surface discharges as a consequence of overtopping to occur.

The internal south wall is a single wall of reinforced concrete construction to 2 m high, over about half the length of the southern boundary to operations. It identifies the limit of Vault 9 but is not intended as a retaining wall.

The first of the future vaults, Vault 9a, will be constructed alongside the existing Vault 9. It will be constructed and filled immediately following Vault 9, before or at the same time as Vault 10, to enable it to be capped at the same time as Vault 9. Vault 9a will have the same design as the other future vaults (see Subsection 3.3.3). The optimised design is simpler than that of Vault 9.

3.3.3 Future Vaults (Vaults 10, 11 and 12)

We anticipate that, by the time these vaults are constructed and operational, we will have phased out the current design of ISO-freight container. All LLW in these vaults will therefore be disposed of in strengthened ISO-freight containers as described in Subsection 3.3.2.

If ILW were to be disposed at the LLWR, then in addition to ILW that can be managed as LLW, these vaults would also contain ILW that cannot be managed as LLW. This population of ILW would lead to higher external dose rates, and it would therefore be necessary to provide shielding to all vault disposal. We have determined that the optimised approach to this would be to construct shielded modules in the vaults [10]. These are shown in Figure 3.6.

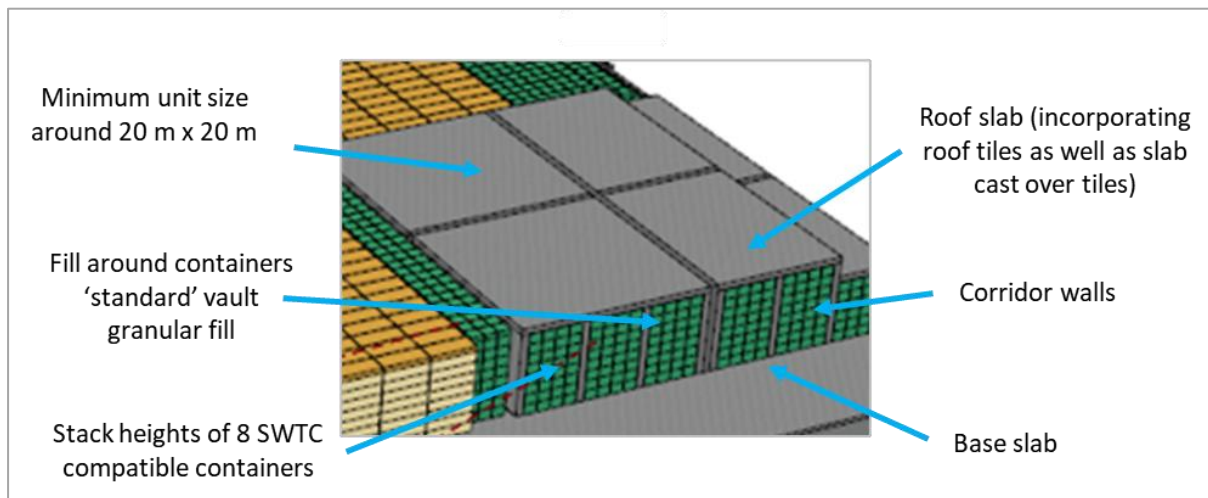


Figure 3.6: Illustration of shielded modules. Note that a ‘cut-away’ representation of the corridors is shown, which would in practice be closed at all times when not involved in active emplacement operations.

The shielded modules would be constructed on a reinforced concrete base slab that is locally thickened compared to the standard future vault bases for LLW disposals, to ensure rigidity of the overall external structures system.²⁰ Otherwise, the base unit and underlying engineering will be as for the standard future vault bases.

The modules would consist of reinforced concrete walls and roofs which would be of sufficient thickness to provide shielding and structural support. Container protection units would not, therefore, be required. The shielded modules would protect containers from precipitation and wind-blown salt, so interim storage in warehouses prior to emplacement in final disposal locations would not be required for these wastes.

The sizes of shielded modules (i.e. the extent of the future vault slab areas they occupy) would intentionally be kept flexible, to match the evolving projections of disposals with time.

²⁰ The requirement for a locally thickened base slab means that it would be easier to incorporate shielded modules into the design for future vaults than it would be to construct them in an existing vault. In addition, it would be beneficial to separate disposal operations from construction operations. For these reasons, we have determined that these structures would not be built in Vault 9 or Vault 9a.

On the basis of engineering input to options work to date, and the need to create a stable and efficient disposal structure, a minimum size of 20 m by 20 m has been identified. In practice, units are expected to be larger than this minimum. The default assumption is that shielded modules would always be the maximum height that will fit under the cap, for volumetric efficiency reasons.

The shielded modules would include the following measures to manage infiltrating water into the vaults.

- Drainage channels would be incised in corridor bases to ensure passive drainage. Arrangements would be made (small holes with coves) to allow drainage through the corridor sides also whilst ensuring shielding would be maintained.
- The corridors would be void filled with standard vault drainage material, which would also be emplaced around the shielded module units, consistent with the overall vault concept for maximising redundancy in drainage capacity, as well as minimising voids for cap resilience reasons.
- Bunding or drain arrangements would be used to isolate waters and leachate associated with the shielded module units from those associated with adjoining vault disposal modules for other categories of wastes. This would include intercepting clean waters shed from the unit roofs during the operational period so they would not become leachate. This would be consistent with the principle of minimising overall leachate volumes.

For the ESC, we have assumed that ILW consigned to shielded modules would be contained in SWTC-compatible strong boxes. Cementitious grout would be used to fill voids within the containers and to help limit contaminant release from the repository.

In our groundwater assessments, we do not model the shielded modules. This is cautious, as it neglects any containment function that the modules would offer.

In the future vaults, disposal of ILW that can be managed as LLW would be the same as in Vaults 9 and 9a.

Vault design

Future vaults (Vaults 9a, Vault 10, and ongoing vaults as required)²¹ will follow the same basic approach as for existing vaults. These vaults will be built along the west side of the trenches (see Figure 3.1).

As they are developed, the secant pile wall along the west side of the trenches will be extended southwards to provide support to the western edge of Trench 3. The design of the

²¹ Inventory projections for the 2026 ESC [4] indicate that it is only necessary to create disposal capacity up to Vault 12 to accommodate all relevant projected disposals. However, designs to date have included vaults up to Vault 14. This approach has been continued in the current design, to provide flexibility with regards to inventory assumptions and national strategy. As the vaults are modular and this does not affect cap design for the first few tranches, this is considered useful in retaining flexibility.

secant wall will be similar to that of Vault 9. The secant wall is only intended to have a structural role and it will not be designed, or relied upon, to provide a low permeability barrier.

The levels of the future vault bases will be stepped to follow the site topography. The vault bases will be simpler than that of Vault 9 and will comprise the following.

- A composite basal liner consisting of geomembrane on BES. The thickness of the geomembrane will be at least 2 mm, while the BES will have a minimum thickness of 500 mm. The composite basal liner will limit releases of leachate to groundwater during the PoA and will continue to provide some barrier function for the long term.
- A reinforced concrete base slab of nominal 350 mm thickness (subject to detailed design) on top of the basal liner. The concrete slab is principally intended to provide a running surface for waste emplacement operations but it may assist in leachate management during the PoA, depending on the presence of cracking. The concrete base slab will not be specifically designed to provide any long-term barrier function, but it may nevertheless provide a barrier to flow at early times, degrading over time.

The perimeter west and east walls for each of the future vaults will be of reinforced concrete around 350 mm thick (subject to detailed design) with composite 2 mm geomembrane and 500 mm thickness of BES on the outside, both in continuity with the basal composite liner. This will provide a composite low permeability liner system to 1 m level intended for outer vault walls. The crest will be level on both sides so that eventual spill-over to the long-term drainage system to east and west will be similar. The concrete will extend over the BES across the crest to give it suitable physical protection.

Above the 1 m level there will be stone drainage material connected to the basal drainage system. The drainage material will be free-draining granular fill, similar to the under-vault drainage blanket (see below). These free-draining 'walls' will be laterally separated with a minimum of 2 m width of low permeability material at each vault boundary so as not to present a preferential north to south drainage pathway for leachate overtopping the 1 m walls. The separators will be of BES or locally available clay materials having permeability not less than 10^{-9} m s^{-1} .

A drainage blanket will be installed under each vault. These will be formed of free-draining stone, 500 mm thick and specified to have a permeability not less than 10^{-4} m s^{-1} and generally in accordance with standard specifications [5]. The basal drainage blankets to each vault module will be separated axially from each other with at least 2 m width of low permeability ($k < 10^{-9} \text{ m s}^{-1}$) material, which may be the local clay, at each level step in vault boundary. This is so as not to present a preferential north to south drainage pathway for leachate, in order to spread distribution to the underlying geology as evenly as possible and avoid a possible flow concentration at the lowest, southern end of the system, i.e. modules associated with Vault 14.

The role of the 1 m walls and the underlying drainage blanket in long-term leachate management are described in Subsection 3.4.3.

3.4 Leachate Management

Any water that has come into contact with disposed or stored waste is considered to be leachate. Leachate management is a key part of the overall management of the disposal facility and, together with other active controls, is an integral component of the optimised design, which ensures that radiological risks are ALARA. These controls also ensure that the non-radiological impacts of the LLWR are appropriately low. Further information on the leachate management arrangements is provided in our '*Engineering Design*' report [5], while our arguments that they are optimised are presented in our '*Optimisation and Site Development Plan*' [10].

3.4.1 Operational Period

Trenches

The final cap will be installed incrementally over the vaults and adjacent strips of the trenches as disposals to each vault complete. This will reduce the quantity of water infiltrating into the trenches and reduce the quantities of leachate generated. The first strips of the final cap will be installed over Vault 8 and the adjacent trenches.

The remaining trenches will not be covered with the final cap for several years. Investigations have revealed that the interim trench cap has not performed as originally anticipated [10]. We have determined that, for these trenches, the optimised approach to improving hydrological performance is to install an improved interim trench cap, comprising a geosynthetic clay liner (GCL). The GCL is currently being installed. The GCL will be removed prior to installation of the final cap.

Any trench leachates that are generated will be collected by the trench leachate drains, which lead to collection sumps. These historically discharged to Drigg Stream, but the system was refurbished in 1991, since when they have discharged to the MHT for sampling, analysis and recording. Effluent is then periodically discharged into the Irish Sea through the Marine Pipeline.

The collection of leachate in trenches has not been completely effective, and evidence exists that radionuclides have entered groundwater [9]. Whilst installation of the GCL and the final cap will substantially reduce the rate at which any releases occur to groundwater, future leakage cannot be ruled out. Monitoring will continue to review whether the engineered measures satisfactorily control leachate or whether any other measures are warranted.

In addition to the interim cap, the partial cut-off wall provides a downstream function to limit near-surface uncontrolled discharges from the trenches. Historical tritium levels in the railway drain were elevated before the installation of the interim cap and bentonite cut-off wall in the early 1990s. Since then, monitoring shows a significant reduction in tritium concentrations, to the point where there is no longer a pathway between the trenches and the railway drain.

Vaults

Prior to capping, the containers, container protection units, shielded modules and interim warehouses will limit contact between rainfall incident on the vaults and the wastes.

Leachate generated in the vaults will therefore only have very low levels of contamination.

The vault bases (details of which are provided in reference [5]) allow leachate generated in the vaults to be directed towards sumps, whence the water will be directed²² to the MHT for sampling, analysis and recording prior to discharge to the Irish Sea via the marine pipeline.

The operational system is currently constrained by the MHT pumping capacity. In practice, the system can comfortably manage anticipated heavy rainstorms. In the case of extreme events, there is provision within the system to manage overflows by use of the automatic diverter valve system, which would direct discharges to the Drigg Stream, or by management intervention to limit discharges from the vault pumps to the MHT.

Installation of the final cap will greatly limit infiltration and the quantities of leachate produced. This will preclude the need for additional discharge capacity.

The MHT system has a consent of 6,500 m³ per day and quantities are anticipated to continue to be well within this. There are also existing controls on leachate quality under the site discharge consent [70]. It is anticipated that these will continue to be met, with planned operations expected to result in improvements in quality with time.

3.4.2 Active Institutional Control

Once waste disposal operations have completed and the final closure engineering has been installed, the site will enter a period of active institutional control until the end of the PoA.

One of the objectives of active institutional control over the site is to build confidence through monitoring that the facility and system are behaving as expected. The baseline assumption for the ESC is that leachate will continue to be actively managed while waste emplacement continues and after final capping of the facility has been completed until the end of the PoA.

Within the vaults, this will be achieved by a gravity-fed system which will drain leachate from sumps installed in the vault bases. Gravity drains leading to the southern perimeter of the facility provide for leachate collection from the trenches. All leachate will be routed to the MHT for monitoring prior to permitted discharge via the Marine Pipeline to the marine diffuser and into the Irish Sea.

Continued active leachate management and monitoring is assumed, consistent with established practice for landfill operations. The leachate management system, including MHT and Marine Pipeline, will be maintained for a period after closure, until it can be demonstrated via monitoring that they are no longer needed (i.e. that there is no significant leachate generation). This is expected to occur relatively soon after installation of the final cap and extension of the cut-off wall are completed; following this the leachate management

²² In Vaults 8 and 9, water leachate is currently actively pumped to the MHT. We intend to move to a gravity-fed system, which is likely to be installed when Vault 9a is constructed. For future vaults, we will only employ a gravity-fed system.

system will be decommissioned. Monitoring of MHT discharges is expected to continue as a Permit requirement up to the point the system is decommissioned. The leachate monitoring points, in the trenches and vaults, are expected to remain in use until Permit surrender or that monitoring confirms that they are no longer required. At that point all remaining leachate monitoring points will be decommissioned and the cap penetrations sealed.

3.4.3 Period Following the Withdrawal of the Environmental Permit

Although the cap design is intended to minimise infiltration as far as is reasonably practicable, water cannot be excluded completely and the rate of infiltration is expected to increase over the long term as cap performance degrades [13]. After the period of active control, under passive conditions and with no collection of leachate from the vault sumps, the potential for saturation over the depth of the waste column in the future vaults is minimised by ensuring that leachate retained by the base overflows the 1 m containment walls on the east and west sides to a drainage blanket beneath the vault base, subsequently flowing beneath the vaults into the underlying geology. This will also minimise the potential for overtopping and discharges to the near-surface environment. The internal north and south walls of the vaults are set slightly higher than the east and west walls to give preferential drainage pathways to the sides, but with ultimate hydraulic continuity (to provide contingency in the event that free drainage does not occur) along the vaults to the south.

Vault 8 and Vault 9 do not currently have hydraulic connectivity with under-vault drainage blankets. To minimise saturation over the height of the waste column in these vaults, passive drainage connections will be installed at closure at the 1 m level between Vaults 8 and 9 and the future vaults.

3.5 Closure Engineering

We have identified engineering design options, taking into account the types of passive control that are relevant to the main threats to isolation and containment of the wastes in the post-closure period, associated with [10]:

- natural disruption of the facility – in particular as a result of coastal erosion and/or inundation;
- disruption of the facility by future human actions – in particular those actions that have the potential to cause disturbance of the wastes, as well as direct or indirect exposures to contaminants within the wastes;
- release of contaminated gases;
- generation and release of contaminated leachate by water entering the facility and contacting the wastes.

For each of these threats, we examined each engineering option in terms of the safety functions associated with the different passive controls. A set of design features was subsequently identified that provide passive safety [10].

- Future vault walls and bases will contribute to the control of leachate that may be produced within the facility. We have considered the implications of alternative strategies for leachate control beneath the cap under both normal (as built) and degraded cap performance conditions.
- Each of the vaults will be connected with an extensive sub-basal drainage layer that will interface with the underlying geology over a wide area. This will maximise drainage, supporting the maintenance of low saturation conditions within the disposed wastes, and will promote ready dispersal by avoiding unnecessary concentration of leachate.
- The cut-off wall is a low permeability wall that will surround the LLWR site including all of the vaults and trenches. The cut-off wall is discussed in Subsection 3.5.1.
- The cap is an engineered barrier that will cover the region within the cut-off wall. The cap is discussed in Subsection 3.5.2.

3.5.1 Cut-off Wall

The main roles of the cut-off wall are:

- to provide reassurance against the possibility of near-surface release close to the facility in the event of degraded cap performance by directing any resulting leachate downwards into deeper groundwater;
- to minimise lateral flows into the trenches and sub-vault drainage.

To deliver these roles, the main requirement for the cut-off wall is that it should provide a permeability contrast with the surrounding geology.

The existing cut-off wall, which runs alongside the north and east sides of the trenches, will be extended to encircle the entire facility during the closure process. The existing cut-off wall is 1 m wide and extends approximately 7 to 9 m deep, into underlying clays. The proposed extension will be constructed to 2 m below the bottom of the vaults, to ensure it extends below the sub-vault drainage blanket. It will be a cement-bentonite slurry wall of similar mix to the section of the cut-off wall already in existence. The cut-off wall will be integrated with the low permeability elements of the final cap and will be constructed consistent with the strip capping schedule.

3.5.2 Cap

Capping of the LLWR site fulfils a number of functions, principally to:

- act as a barrier to reduce the likelihood of, and impacts arising from, human intrusion;
- isolate and protect wastes;
- reduce water infiltration rates into the trenches and vaults;
- control gaseous emissions arising from waste degradation;
- reduce direct and indirect external irradiation doses.

All aspects of cap design and function are discussed in detail in reference [5]. A summary of the details of particular relevance to the current HRA is presented here.

As described in Subsection 3.2, an interim cap was installed over Trenches 1 to 6, being completed in 1989. This was extended to cover Trench 7 following its closure in 1995. The interim trench cap was, as discussed in Subsection 3.4.1, found to be performing less well than expected, and we are currently installing an improved interim trench cap over the southern part of the trenches. The improved interim cap will be removed prior to installation of the final cap.

On the basis of optimisation studies [10], the final cap will have a single dome profile, primarily because of its resistance to erosion compared with alternative designs. This design ensures that rainfall will be shed to the perimeters away from the wastes as efficiently as possible, as indeed is normal practice for landfills, repositories and similar cover systems.

The final cap will be installed in strips across each vault and adjacent area of trenches once each vault is full. It will thus be developed from the north to the south in conjunction with preparation of each subsequent vault. For each strip, a temporary seal will be constructed at the leading edge of the cap to ensure protection of containers under the cap from environmental effects.

The current assumed schedule for installing sections of the final cap is detailed in Table 3.1, although it should be recognised that there is a high degree of uncertainty in the schedule for installation of the final cap. This is because the capping schedule has been derived based upon waste arisings profiles, which are subject to significant uncertainties, as discussed in reference [4].

Table 3.1: Capping schedule assumed in the ESC. The final cap over the trenches is installed in sections at the same time as the adjacent vault is capped.

Trench or Vault	Interim capping	Final capping
Trench 1	1989	Phased
Trench 2	1989	Phased
Trench 3	1989	Phased
Trench 4	1989	Phased
Trench 5	1989	Phased
Trench 6	1989	Phased
Trench 7	1995	Phased
Vault 8	N/A	2036
Vault 9	N/A	2057
Vault 9a	N/A	2057
Vault 10	N/A	2097
Vault 11	N/A	2118
Vault 12	N/A	2142

The final cap is an engineered multi-layer barrier that will cover the trenches and vaults, and the optimised design is shown in Figure 3.7. As described in our '*Engineering Design*' report [5], the layers function together as an integrated engineering system. The key layers of relevance to the HRA are described briefly below.

- The geomembrane and BES layers are the primary layers responsible for limiting water ingress. These layers form a composite barrier system, where each layer complements the other to enhance overall hydraulic performance.
- The drainage and protection layers (above the low permeability layers) assist performance by minimising the hydraulic head and providing preferential flow paths to the surface water drainage channels around the cap perimeter. The geosynthetic protection components safeguard the integrity of the low permeability layers during construction activities.

- The bio-intrusion layer provides a dual function, providing additional drainage capacity as well as protecting the low permeability layers from burrowing animals and deep rooting vegetation, achieving the latter by providing a low nutrient zone.
- The upper layers support vegetation, which contributes to cap performance by reducing hydrologically effective rainfall (HER) through evapotranspiration. Additionally, the root systems enhance slope stability by reinforcing the upper soil structure.

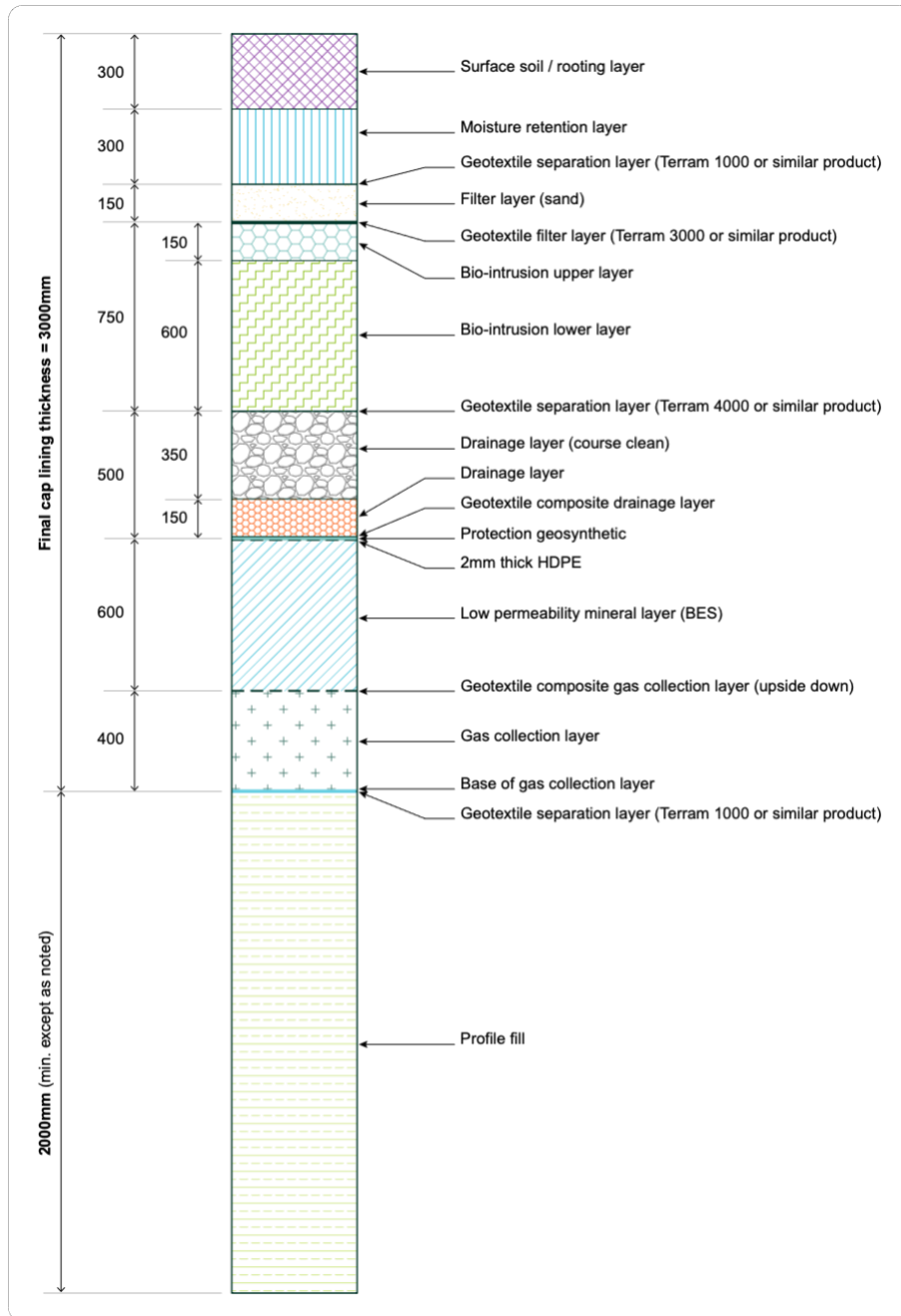


Figure 3.7: Layers of the final cap [5]

The single dome profile will be created using a layer of profiling fill above the wastes and below the cap. This will provide a suitable foundation for cap construction and will ensure adequate cross-falls for drainage and stability. The thickness of profiling fill will vary across the cap to suit the profile but will be a minimum of 2 m thickness and generally more. The profiling also has a function to spread the influence of differential settlements, thus protecting cap performance.

We will only use free-draining, low fines granular material for profile fill over the vaults. Over the trenches, we will additionally use excavated materials from construction of future works, as well as imported cohesive material of suitable specifications

The final cap will act as a barrier to the inflow of water to the trenches and vaults, minimising infiltration to the extent that it is less than the combined drainage capacity of the underlying geology and the under-vault drainage blanket, thereby keeping the wastes unsaturated for as long as reasonably practicable. The cap will also minimise the likelihood of disturbance by future human actions and bio-intrusion.

Gas venting arrangements

The cap will contain a passive gas venting arrangement during the PoA. This is to ensure that the generation of bulk gases cannot damage the cap and thus reduce cap performance. The planned venting arrangement is shown in Figure 3.8. This can be considered to be a mature basis for the ESC, as it has been part of the cap design for several decades. Nevertheless, this is an area of continuing design development and optimisation, as the gas vent does not need to be implemented in the first (or indeed second) strip of the cap.

The vent arrangements are based upon the gas collection layer (labelled 11, purple) connecting with the drainage layers above the low permeability layers (these drainage layers are labelled 5 – biointrusion barrier; and 6 – drainage layer - in this figure. These layers have been combined in the updated cap design). The vent will form a continuous ring around the crest of the cap.

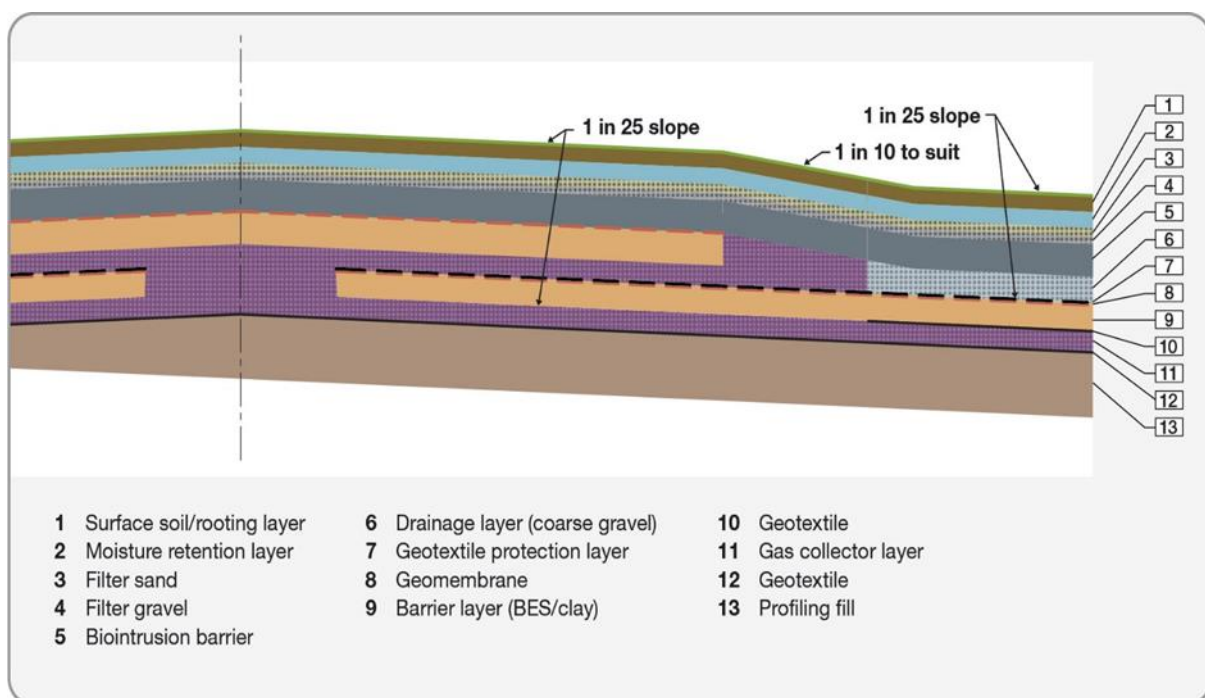


Figure 3.8: Gas collection and vent detail

The penetration at the peak of the cap will be covered by a BES layer, which will limit the potential for water incident upon the peak to infiltrate into the repository via the gas vent. The gas vent and, above the geomembrane, the material down-gradient of the vent, will consist of granular material. This will be free-draining and will also act as a capillary breaking layer. It is not expected that significant clogging will occur. These considerations, together with the slope of the cap, will lead to water infiltrating into the gas vent material being directed

towards the cap perimeter, and away from the penetration in the membrane and the BES at the peak of the cap. We expect, therefore, that the gas vents will not result in increases in water flows into the repository.

We anticipate that bulk gas generation will continue for a significant period of time [6]. Our assumption is that the gas vents will be decommissioned and sealed at the end of the PoA but we retain the flexibility to leave the vent open at the end of the PoA if necessary. We do not need to take a decision on vent closure until towards the end of the PoA. Any decision will be informed by many years of years of monitoring, modelling and analysis to provide the required levels of confidence in the understanding of bulk gas evolution.

Cap resilience

As the final cap is the most significant barrier governing repository performance, we have considered in detail the resilience of the cap to settlement and other processes that could affect performance (e.g., erosion and seismic effects) [13, 71]. We have a high degree of confidence [5] that the cap will be resilient to potential general and differential settlements for a prolonged period. Measures which contribute to this confidence include the following.

- The minimum profile fill thickness of 2 m is intended to help distribute strains, while the choice of granular profile fill over the vaults provides more extensive strain distribution than that provided by cohesive materials. Together, these measures will reduce strains imposed on the geomembrane and BES by differential settlement.
- The decision to surcharge relevant categories of waste in Vault 8 and Vault 9. This will result in the expression of settlement prior to installation of the final cap, thus reducing the potential for settlement after cap installation.
- The assumption of a move to stronger containers in the future for both LLW, and any ILW to be disposed to the vaults. The high-pH anaerobic conditions which will be established under the final cap will lead to low corrosion rates [6]. We expect, therefore, that these containers will provide a stable base for the cap and profile fill for a substantial period of time and, potentially, until the site is disrupted by coastal erosion.
- Updates to the understanding of the strain tolerance of geomembranes and BES compared to assumptions in the 2011 ESC.
- Current and future controls (via WAC) on available and future voidage for vault disposals will reduce the potential for settlements which would exceed the strain tolerance of the geomembrane and BES to occur.

This understanding has been used in our EPA, where we have examined in detail the potential evolution of the cap after installation [13]. Based upon these considerations, we have assessed the performance of the cap. We discuss the EPA further in Subsection 4.1.9.

3.6 Key Aspects of the Disposal System Design

As set out within our '*Optimisation and Site Development Plan*' [10], we have optimised the design to provide good performance in relation to the release of radionuclides to the environment. Our optimised approach is to try to keep the wastes unsaturated for as long as practicable and to direct releases to the underlying geology, which we aim to achieve through the following engineering measures:

- the installation of an engineered cap which will perform to a very high standard and will reduce the infiltration of water into the waste;
- 1 m-high sidewalls in future vaults which, together with the passive drainage system, will direct water which overtops the vault walls to an under-vault drainage blanket and thence to the underlying geology;
- passive drainage connections installed at closure between Vaults 8 and 9 and the future vaults, to ensure long-term passive drainage performance in these vaults;
- a cut-off wall which will help direct discharges to the underlying geology, rather than to the near-surface environment, and which will help to limit lateral flows into the vaults and the under-vault drainage blankets.

This approach will mitigate against the eventual possibility of localised discharge of contamination to the surface environment around the LLWR. This approach is equally effective in limiting the release of non-radioactive contaminants.

In developing the design, we have taken into account the approaches used in landfill engineering, for example in relation to the repository cap. However, neither the vaults nor the trenches conform to a conventional landfill design. As explained in reference [10], we have developed an optimised design that addresses and satisfies the requirements set out in the GRA [19]. As noted in Subsection 2.1.1, a key requirement is to demonstrate a level of protection that is 'no less stringent' than that required to achieve 'nationally acceptable standards for disposing of hazardous waste'.

During the PoA, releases from the vaults are expected to be insignificant. The vault bases will ensure minimal release of leachate from the vaults, thereby allowing any leachate collecting at the bottom of the vaults to be managed and discharged via the Marine Pipeline as required. Collection of leachate will be effective because the waste containers will remain substantially intact during this period. Contact between any waste and the water will also be limited because of the presence of the waste containers. The system will be monitored to confirm that behaviour is as expected. The final cap will be installed incrementally over the course of the PoA. After it has been installed, water inflow into capped vaults will be very low for the remainder of the PoA.

Releases from the trenches during the PoA will be more significant than releases from the vaults. For the radionuclide inventory, we have considered the merits of remediating the trenches (e.g. by partial retrieval of the wastes) and have concluded that this would not be

appropriate. We consider that the same arguments apply in relation to the non-radioactive components of the inventory.

Leachate from the trenches is collected and released via the marine pipeline during the PoA. Some leachate is, however, released to groundwater through the trench bases. As with the vaults, the trenches will be capped incrementally. Once the final cap is installed and the cut-off wall constructed water inflow into the trenches will reduce significantly. Consequently, releases from the trenches will reduce significantly.

After the end of the PoA, as the final cap and the other engineered barriers (and, for the vaults, the containers) degrade, releases from the repository will begin to increase, although they are expected to remain very low.

Our design for the vaults is expected to be more effective in limiting releases of non-radioactive contaminants than would be the case for a landfill in the following respects.

- The waste in the vaults is contained in mild steel containers, which will provide substantial containment during the PoA, whereas non-radioactive wastes disposed to landfill are often uncontainerised.
- Together with the vault bases, the persistence of the mild steel containers will ensure that pumping of leachate will be highly effective in limiting releases to groundwater during the PoA.
- The repository's engineered cap will contain a membrane and BES layer which work in concert to provide a very effective barrier to infiltration of water into the repository for a substantial period after the end of the PoA. We expect that this composite barrier will perform better than the simpler barriers installed in many landfills, which are often designed to perform for much shorter timeframes than we consider [13, 72].
- The vault bases will limit releases of contaminants to groundwater until water levels increase such that either heads are sufficiently high to result in water flows through the bases, or levels increase until the low side walls overtop.
- The low sidewalls in the future vaults, together with the cut-off wall, will make local discharge to the surface environment much less likely than would be the case with a landfill.
- The use of low fines, inert granular material between container stacks will provide redundancy with respect to vertical drainage routes through the vaults, helping to direct infiltrating water to the vault base and hence minimise the potential for near-surface discharge of leachate.
- The wastes in the vaults are grouted to reduce voidage, promoting the stability of the cap. Once the final cap is installed, the grout will help establish chemical conditions which will limit the release of contaminants. The chemical conditions will also limit the corrosion of mild steel containers, ensuring that they continue to provide substantial containment and contribute to cap stability after the PoA. The profile fill will act as a

strain distribution layer, minimising the effects of any settlement on the BES layer and geomembrane. The measures that we will take to enhance cap stability in the long term contrast with practice in many landfills, which typically include different types of wastes with higher initial and total potential voidage. Landfills are generally not designed with a focus on the very long term and typically have reduced levels of engineering to protect the cap [72].

- The period of active institutional control will be substantial, allowing monitoring of the evolution of the engineered features. This will help improve our understanding of their evolution, as well as providing the opportunity to remediate any defects identified as a result of these monitoring operations.

These observations are part of our argument that the level of protection provided by the LLWR is no less stringent than would be provided under the regulations pertaining to hazardous wastes.

4 Risk Assessment

4.1 Assessment Approach

4.1.1 Nature of the Hydrogeological Risk Assessment

We have considered two possible approaches to calculate non-radiological impacts to groundwater:

- the 'landfill' approach;
- the 'repository' approach.

In the 'landfill' approach, an assessment would be conducted using the LandSim [73] software. This would use the observed concentrations of contaminants in leachate to simulate the rate of loss of contaminants from the landfill and calculate the future concentrations of hazardous substances and non-hazardous pollutants.

The LandSim approach could apply to the trenches and we have considered the use of trench leachate data to characterise the trench source term. This approach would entail some significant limitations:

- there are only leachate data for a small number of contaminants, and these only provide a view of contaminant concentrations over a limited time period and at particular locations;
- the leachate data provide little information about spatial variation of contaminant concentrations in the trenches and no information about their evolution with time.

Conversely, LandSim would not be appropriate for the vaults at the LLWR. This is because of the following considerations.

- Vault wastes are predominantly disposed in mild steel ISO-freight containers, which limit contact between incident water and the wastes. Consequently, contaminant levels in vault leachate are low. Vault leachate monitoring data do not, therefore, provide an appropriate basis on which to characterise the vault source term. As the LLWR is the only facility of its kind in the UK, appropriate analogue data are not available.
- LandSim does not represent release pathways in a sufficiently flexible manner. For the optimised design for the LLWR, we expect, in the long term, that overtopping of the vault sidewalls and direction of overtopping waters to the underlying geology will be more significant than transport through the degraded base of the facility. LandSim is focused on release through the base of a landfill.
- The HRA covers the same timescales as the radiological assessment of the groundwater pathway, which encompasses the PoA and several thousand years following the withdrawal of the environmental Permit. Over these timescales, we

need to consider the evolution of the engineered barriers (in particular the engineered cap) and the implications for the environmental performance of the facility. We have undertaken an EPA [13] which considers these aspects in some detail. We require an assessment model approach which has the flexibility to incorporate the findings of the EPA. LandSim lacks this flexibility.

- LandSim has no facility for representing radionuclide chains and we prefer an approach where a consistent methodology and software are employed for both radioactive and non-radioactive contaminants.
- There is an expectation that numerical limits are set on the quantities of radionuclides, non-radiological contaminants and organic complexants that are disposed to the LLWR. Using a landfill-type approach would mean that there is no simple link between disposed inventories and the near-field pore water concentration because the LandSim calculations are based on measured concentrations in pore water.

Viewing the HRA within the wider context of operations at the LLWR, we consider it appropriate that potential non-radiological impacts be assessed in a manner that is as consistent as possible with the approach adopted for the assessment of radiological impacts.

In view of these considerations, we developed, as part of the 2011 ESC, a bespoke performance assessment model to evaluate both radiological and non-radiological impacts from the LLWR. We developed this model further to undertake the 2017 HRA, with alterations and improvements where necessary to meet specific recommendations made in the Environment Agency's review of the 2011 ESC and taking account of new information and improved understanding arising since 2011. Key developments included:

- an improved representation of the geology and hydrogeology leading to an improved hydrogeological model;
- a more sophisticated treatment of the B2 lithofacies unit in the assessment model;
- the assessment of releases within one model that is run from the time of the first disposals into the post PoA period (rather than having separate models for the PoA and post-PoA periods).

We have developed this model further as part of the 2026 ESC to evaluate radiological and non-radiological impacts from the LLWR. Significant updates are listed in Subsection 2.1.4.

4.1.2 Assessment Framework

As discussed in Subsection 4.1.1, we adopt the same assessment approach for the HRA as we do for our radiological assessments. This is a well-established approach, as described by the Nuclear Energy Agency (NEA) [74] and by the International Atomic Energy Agency (IAEA) [75]. This includes the following activities:

- 1) acquisition of information and data, and development of understanding of the disposal system and the uncertainties present, including those related to waste form, site characteristics, engineered structures and their evolution;
- 2) identification of the pathways potentially leading to the transfer of radionuclides and non-radiological from the repository to humans and the environment, and refinement/definition of those pathways taking account of system-specific understanding and data;
- 3) definition of the broad scenarios for the long-term evolution of the disposal system and its environment, and identification of the cases that should be assessed taking account of the model capability;
- 4) developing and testing of conceptual and mathematical models of the behaviour of the disposal system and its components, and implementation in quality assured numerical models and codes;
- 5) application of all of the above in structured and comprehensive analysis to yield the required assessment endpoints for comparison to regulatory requirements, including assessment of the relevant uncertainties;
- 6) evaluation of the robustness of the analysis and its results, and identification of key uncertainties that have bearing on the interpretation of the results and conclusions that should be drawn from the assessment.

We have set out this approach in our '*Assessments Manual*' [47].

The acquisition of information and development of understanding (item 1) is set out in other ESC reports [3, 4, 5, 6, 8, 9]. In the following subsections, we outline our approach to treatment of uncertainties, identification of scenarios and cases, and achieving model quality. Assessment endpoints are discussed in Subsection 2.1.1 and 2.1.2.

4.1.3 Lifecycle Phases

The lifecycle phases of the LLWR are shown schematically in Figure 4.1. The HRA considers a period from the time of the first disposals to Trench 1 in 1959, extending through the PoA and some thousands of years into the period thereafter. During the first part of the PoA, operations continue, but in the latter part of the PoA, the site is managed and the repository is monitored, while no further disposals occur.

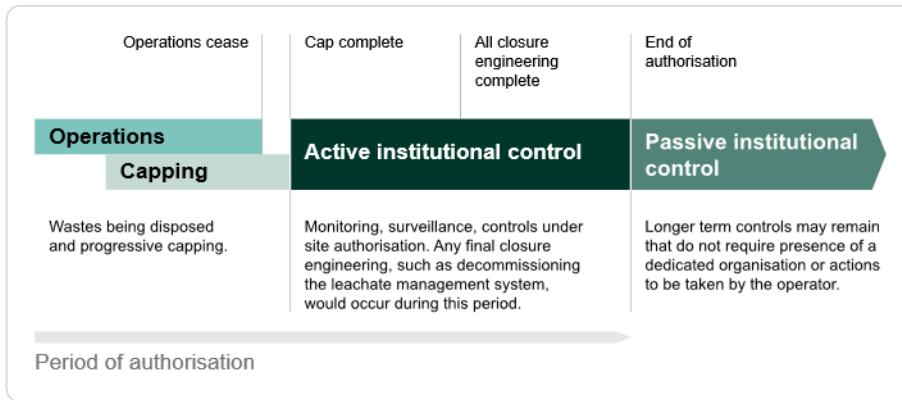


Figure 4.1: Lifecycle phases for the LLWR

The operational phase runs from the time of first disposals (1959) until the emplacement of the last tranche of the final cap. This is anticipated to occur, based upon our understanding of the reference inventory and engineering judgement, in 2242.

Following this there will be a further period of active institutional control, during which monitoring and active leachate management will continue. We have reviewed our approach to the period of institutional control [10] and we have concluded that a period of 100 years is appropriate. We assume that the site will be released from regulatory control at the end of the period of institutional control. After this point, the site will no longer be managed.

The reference case for the HRA addresses a timeframe that includes the PoA and runs for approximately 1,000 years after the site is released from regulatory control. This is because of the expectation that the repository will be eroded by the sea. At its north-western corner, the LLWR site is only around 450 m from the high-water mark, so the site is vulnerable to sea-level rise and coastal erosion. Thus, consideration of sea-level rise and coastal processes and assessment of their effects are significant aspects of the ESC. We discuss these aspects and their treatment within the assessment in Subsection 4.1.4.

4.1.4 Scenarios for the 2026 ESC

In this subsection we discuss how our understanding of the site's lifecycle is used to arrive at scenarios for consideration in the assessment.

General definitions

Within our assessment approach, we define scenarios as follows.

- Scenarios are the broad alternative future events, circumstances, conditions or their evolution, that are characteristically different and provide a framework for analysis that is useful in providing an understanding of the environmental safety of the disposal system.

Scenario is a high-level term denoting a substantial difference of site development or long-term evolution. We acknowledge that circumstances that are intermediate between the selected scenarios may occur, but analysis of performance under the selected scenarios is

designed to provide representative illustrations on which an analysis of environmental safety can be based.

There may be relatively few scenarios. On the other hand, a large number of calculation cases may be required to adequately explore all aspects and uncertainties within a scenario. Thus:

- A calculation case is a specified combination of events, circumstances, conditions or their evolution, including specification of model boundary conditions and data, which represents a particular realisation of the disposal system, its evolution and radionuclide or contaminant release path of concern.

Alternative calculation cases are defined to investigate the effect of particular assumptions or uncertainties, or combinations of assumptions, e.g. of near-field evolution, geosphere path, etc.

Evolution scenarios

To assess the environmental performance of the LLWR, we assess impacts assuming development of the facility according to an optimised Site Development Plan accommodating currently disposed and emplaced waste, plus estimated future arisings of LLW. We have also considered how the Site Development Plan would require modification to allow the LLWR site to dispose of some ILW.

We also consider the evolution of the surrounding environment and its effect on the repository. Currently, the closest point of the LLWR vaults lies approximately 350 m inland from the present-day coastline. It is therefore expected that the LLWR and its surrounding environment will be affected by the various manifestations of climate change. The main impacts of climate change are expected to be:

- sea-level rise;
- coastal erosion;
- changes in precipitation and temperature.

In the absence of coastal defences, which would be of uncertain long-term efficacy and cannot be assumed in a safety case, sea-level rise and coastal erosion will result in the disruption of the repository, with consequent dispersal of wastes into the coastal environment. Details of the expected development of the coastline are given in reference [8].

There are large uncertainties over future global greenhouse gas emissions and climate change and, consequently, the timing and magnitude of sea-level rise. The magnitude of sea-level rise affects the timing and rate of erosion, with higher sea-level rise resulting in earlier and faster erosion of the repository.

To explore the effect of these uncertainties on repository performance, we have defined low, reference and high greenhouse gas emissions scenarios. For each scenario, we have developed integrated projections of greenhouse gas emissions, sea-level rise and coastal erosion [8].

Based on these studies, we expect that the repository is liable to be first disrupted by coastal erosion within a time frame of several hundred to thousands of years. For a given scenario, there are still significant uncertainties as to when the repository would be disrupted. Given these uncertainties, we judge it inappropriate to provide a best estimate for the timing of disruption of the facility. Rather, we have identified reasonable estimates for the purposes of modelling. These are provided below:

- in the low greenhouse gas emissions scenario, the timescale for first exposure of the repository is 1,750 years after present, with complete loss of the northern part of the repository 3,750 years after present;
- in the reference greenhouse gas emissions scenario, the timescale for first exposure of the repository is 1,250 years after present, with complete loss of the northern part of the repository 2,750 years after present;
- in the high greenhouse gas emissions scenario, the timescale for first exposure of the repository is 750 years after present, with complete loss of the northern part of the repository 1,750 years after present.

For each of these scenarios, we use the Site Development Plan as the basis of the assessment. Each scenario encompasses the broad expectation for evolution of the disposed wastes, engineered barriers and natural environment. This is based on our characterisation and understanding of the wastes [4, 6], engineered barriers [13], hydrogeology of the site [7] and environmental processes and their impacts [8]. We take these into account in our assessment model.

For the reference scenario, we explore the effect of uncertainties through the definition of appropriate assessment cases (see Subsections 4.1.6 and 4.3).

In addition to the different emissions scenarios, we consider a 'what-if' delayed erosion assessment case. We stress that the delayed erosion scenario represents a climate change scenario that is not expected to occur in reality. It does, however, provide the opportunity to investigate the consequences of survival of the LLWR site in the long term.

In this scenario, we consider the site to remain un-eroded 250,000 years into the future. As this evolution is not aligned with a particular climate scenario, we have assumed that the infiltration rates and flows are the same as at the present day. We note that a landfill HRA would not generally consider such aspects or such long timescales.

Our assessment cases, and their mapping to the emissions scenarios, are presented in Subsection 4.3.

4.1.5 Disruptive Events

Prevention and mitigation of disruptive events are considered in our '*Engineering Design*' report [5]. The emphasis is on prevention of disruption where possible, with the provision of sufficient safeguards to mitigate against any adverse environmental impacts in the case that disruption does occur. For example, in the event of degraded cap performance, the cut-off

wall and passive drainage arrangements will work together to provide reassurance against the possibility of near-surface release close to the facility.

The facility will be actively managed during the PoA. Disruption during this period will therefore either be prevented or, if it occurs, remedial action would be considered.

In the discussion below we consider:

- natural disruption;
- the effects of future human actions;
- other disruptive events.

Natural disruption

Considering the timescales over which the radiological component of the LLWR waste could pose a potential environmental threat and the proximity of the site to the coast, the most significant natural disruptive event is likely to be coastal erosion. As discussed in Subsection 2.1.3, consideration of non-radiological contaminants during coastal erosion is not required.

Extreme rainfall events could potentially cause localised flooding. During the operational phase, uncapped vaults will present large open areas that could be susceptible to flooding, most notably the combined Vault 8 and Vault 9 area. The leachate drainage system and pumps have been designed [5] to cater for a wide range of rainfall events. The repository design includes provision of safe temporary waste storage for possible extreme rainfall events in excess of this.

Post-closure, the final cap has been designed to minimise the impacts of extreme rainfall events on infiltration [5]. Final design of surface water features, including surface drainage and the diverted Drigg Stream, will cater for post-capping drainage flows consistent with contemporary design standards. In the event of extreme rainfall beyond the design basis, there may be localised surface ponding. Such localised ponding is not considered to be a threat to the performance of the repository design [5].

The implications for the LLWR of potential seismic events were considered for the 2002 Post-closure Safety Case (PCSC) [76]. It was confirmed that the repository design conforms to standard UK nuclear provisions in this regard. These same provisions would be expected to provide adequate safeguards against non-radioactive releases resulting from seismic events. While most concern relates to the operational phase, cap stability in the longer term was considered in the 2002 PCSC [77]. It was concluded that stability would be satisfactory for slopes up to 1 in 4. As the slope of the final cap is only as high as 1 in 5 at the edges [5], problems with stability are not expected to arise.

More recently, we have undertaken geotechnical modelling and material strain assessments [71, 78, 79] which confirm that the cap system is resilient to dynamic loading. Strain levels induced by seismic-like deformation remain well within the material tolerance thresholds (as described in the EPA [13]), ensuring long-term integrity [78].

Our position with respect to the risk of a tsunami affecting the LLWR was set out [80] in response to a Technical Query raised by the Environment Agency in review of the 2011 ESC. In summary, whilst tsunami events cannot be ruled out for the north-west coast, estimates of wave height, based on past events and postulated scenarios, do not exceed 1 to 2 m, with potential local effects up to a height of 4 m. Given that the high ground between the LLWR site and the coast exceeds 20 m, we consider it very unlikely that a tsunami would have an impact on waste currently stored or disposed at the site. The potential for a tsunami-like event to impinge on the site increases as sea level rises and coastal recession occurs. It is considered, however, that once the facility is capped, there would be no significant impact from such events beyond some limited erosion of the upper layers of the cap.

Disruption by future human actions

During the PoA, disruption of the site by human actions is precluded by site management arrangements, including operational procedures, the use of barriers and other controls to prevent access to the site. Following the PoA, human disruption of the site could conceivably be limited by measures such as preservation of information relating to its presence and content, planning and legal controls over use of the site, and the use of signage and markers. Such active post-PoA measures cannot be and are not relied upon, however, and so minimisation of the long-term impact of disruption by future human actions relies upon using passive engineering measures.

The primary passive engineering control to reduce the likelihood of disruption is the depth of waste burial beneath the ground surface controlled by the thickness of the final cap, including profiling material.

As discussed in Subsection 2.1.3, the impacts of non-radiological contaminants do not need to be considered as part of the human intrusion assessment. We have considered the effect of a human intrusion event upon subsequent repository performance [15]. We have concluded that the effects on repository performance would be very minor.

Other disruptive events

In response to a Technical Query raised by the Environment Agency in review of the 2011 ESC, we conducted an assessment [81] of the potential for contaminant release due to short-lived waste fires after the PoA. Subsequent work [82] established that the potential for a long-term fire was negligible.

Due to the nature of the LLWR, this assessment was focused on the potential for release of radionuclides. We concluded that:

- the materials and conditions in the trenches at any time after the PoA are very unlikely to support combustion;
- any fire events that might occur must be confined to very small volumes of waste with little or no potential for release of radionuclides.

We consider these same arguments to apply to any potential adverse impacts related to non-radiological releases from the site following a fire.

As discussed in Subsection 3.5.2 we have concluded, based on detailed assessments, that the final cap will be resilient to potential waste settlements. Such settlements will therefore have minimal impact on system performance.

4.1.6 Treatment of Uncertainty

As discussed in the GRA (paragraphs 6.3.23 to 29 and 7.3.8 to 20), uncertainties, their representation and their management are central to assessments of repository performance and the wider ESC.

Sources of uncertainty

Uncertainty originates from lack of knowledge of parts of the studied system, from the random nature of some phenomena, and from approximations included in the models and tools we use to represent the system.

Lack of knowledge can arise from limited measurements, uncertainty in interpretation of measured data, insufficient understanding of processes, spatial variability and the uncertain occurrence of events. In the context of the LLWR and its assessment, uncertainties arise from:

- uncertainties concerning past and future waste disposal operations, e.g. the inventory, future engineering and waste management choices;
- uncertainties in the characteristics of the waste disposal facility and its environment, e.g. due to measurement uncertainty, uncertainty in interpretation, applicability of literature values, and variability of parameters in time and space;
- uncertainties about the long-term evolution of the waste disposal facility and its environment and about the future events that may have an impact on the disposal facility and its environment;
- uncertainties concerning future human actions and behaviour.

It is necessary to characterise the relevant uncertainties and analyse them to the extent needed to guide the development of the facility and assure its safety and consistency with regulatory requirements. The treatment of uncertainty may, however, be constrained by limited data and the models that are available.

Our goal for treating uncertainty

Our goal within the HRA is to present a qualitative understanding of the key uncertainties, and quantitative illustrations of the effects of quantifiable uncertainties, such as to inform both our own and the Environment Agency's judgement as to whether our assessment results are consistent with the requirements of the GRA. In addition, we identify priorities for further work by which key uncertainties might be reduced. Some uncertainties will remain unresolved, but this does not prevent conclusions being drawn about the safety of the LLWR

taking account of the uncertainties. As stated in the GRA, paragraph 7.3.8: '*Uncertainties themselves are not obstacles to establishing the environmental safety case, but they do need proper consideration and including in the structure of the environmental safety case as appropriate.*'

Approach in this assessment

We adopt the classification of uncertainties that is conventional in radioactive waste disposal assessment, focusing on their mode of treatment in the safety assessment, thus:

- scenario uncertainty – definition of a scenario, or scenarios, sufficiently broad to represent the possible evolutions of the disposal facility and its environment;
- model uncertainty – models, including alternative model assumptions, that adequately represent the features, events and processes (FEPs) that are important to radionuclide release, transport and exposure to humans;
- parameter uncertainty – variation of model parameter values within their realistic or possible ranges.

Within the 2026 HRA, we have:

- identified reference and alternative scenarios for the future evolution of the natural environment, and identified cases for assessment within each (Subsection 4.3);
- identified expected and alternative evolution cases for the engineered barriers of the repository and, using these, defined assessment cases (Subsections 4.1.9 and 4.3);
- developed sound conceptual models, including considering alternative assumptions (Subsection 4.2), and implemented the models in quality assured codes (Subsection 4.6);
- applied these models to analyse and assess the groundwater pathway including taking account of alternative assumptions and parameter uncertainty (Subsection 4.7);
- undertaken FEP and bias and uncertainty audits, which indicate how FEPs, biases and uncertainties have been addressed within the models and cases analysed in the HRA (Subsection 4.1.8).

With regard to parameter uncertainty, in most cases parameter values, together with upper and lower bounds are drawn from standard sources or established by recorded judgements.

4.1.7 Treatment of Variability

Variability arises because the properties of media within the repository system will vary spatially and over time. In the 2026 HRA we have considered both spatial and temporal variability when developing the assessment approach and model.

Temporal variability

We have identified where temporal evolution is sufficiently important that it requires consideration within the assessment. For these parts of the system, we have undertaken work to characterise the temporal variability for use in either the assessment model, or in key supporting modelling (e.g., the hydrogeological modelling). In particular:

- We have identified different expected evolution scenarios and used these to parametrise sea level rise, rates of coastal erosion and precipitation. This parametrisation provides boundary conditions for our hydrogeological modelling and allow us to model the changes in head and water flows in the repository and the geosphere over the timeframe of the assessment.
- We have undertaken work [13] to understand the evolution of the repository's engineered barriers. Based upon this understanding, we have derived parameters which allow us to represent the temporal variation in barrier performance over the course of the assessment.
- We have modelled the increase in inventory over the operational period of the repository. We use a simple approach in which the inventory is assumed to be added linearly over the operating lifetime of a trench or vault.
- We model explicitly the construction of each trench and vault. Prior to construction, a cell is assigned the same properties as the surrounding lithofacies unit. Once a trench or vault is constructed, the cell is assigned the properties appropriate to a trench or vault filled with waste.
- We model incremental installation of the final cap and construction of the cut-off wall.

More details of the treatment are provided in reference [25].

In our near-field work [6] we have considered the evolution of chemical conditions within the near field over timescales of several thousand years. For the trenches and the vaults, we expect that chemical conditions will remain sufficiently constant over the assessment timeframe that it is appropriate to use solubility and sorption parameters which are appropriate for the initial chemical conditions throughout.

The physical state of the containers will evolve as they corrode over time. There will also be spatial variability in the physical state of the containers. We expect, based upon the anticipated chemical conditions, and the measures we will take to protect containers prior emplacement, that the containers will provide a significant barrier to contaminant release for a significant period of time. In our assessment, however, we do not quantitatively model the containers and we do not, therefore, consider temporal evolution or spatial variability of the containers.

Spatial variability

Our aim, when considering spatial variability, has been to develop a treatment which allows us to represent spatial variability at length scales commensurate with our level of knowledge

of the system. In the main, we have treated variability through discretisation of the system into cells based upon our understanding of the engineering design, inventory, chemical conditions and hydrogeology. For example:

- The trenches are significantly different to the vaults in terms of engineering design, chemical evolution and disposed inventory and therefore we model trenches and vaults separately.
- We discretise the trenches into subsections. The inventory of each subsection is calculated as the total trench inventory divided between the relative volumes of the subsections. This provides a simplified representation of the heterogeneity of the trench wastes, but is reasonable because chemical conditions are not expected to vary significantly over the spatial extent of the trenches. We do not have detailed information as to distribution of contaminants in waste items and therefore we cautiously assume that all contaminants are accessible by infiltrating water
- We represent each vault separately in the assessment, which allows us to account for differences in the design of each vault which are important when assessing performance (e.g., vault base permeabilities). This also allows us to account for differences in the inventory disposed, or anticipated to be disposed, in each vault. We assign the waste in each vault the same effective, averaged density and porosity which accounts for the material composition of the waste in each vault.
- We do not have detailed information as to the distribution of waste within Vault 9 and future vaults, and we therefore treat waste as being uniformly distributed throughout each vault and cautiously assume that all contaminants are accessible by infiltrating water. Consequently, in the assessment model we do not consider sub-vault spatial variability in the distribution of contaminants, the distribution of waste types and any resulting variability in chemical and physical properties. This is an appropriate approach because:²³
 - it is consistent with our level of knowledge and, in our assumption that all contaminants are accessible to infiltrating water, is cautious;
 - chemical conditions in the vaults will be dictated largely by the quantities of container steel and grout present and we expect, therefore, that localised variations in chemical conditions will have limited influence on contaminant release from the wastes and transport in the near field;
 - contaminants from different regions of a vault will mix in the saturated region at the base of a vault prior to release into the geosphere, which means that releases

²³ For Vault 8 we do have more detailed information as to the spatial distribution of the wastes within the vault. In the assessment, however, we treat waste in Vault 8 as being homogeneously distributed, as we do for Vault 9 and future vaults. The considerations in the following bullet points imply that the bias in doing so is not significant.

will tend towards those which would be expected from a homogeneous mass of waste.

- We explicitly represent the B2 and B3 lithofacies units in the assessment model, reflecting the different hydrogeological properties of each unit. Each lithofacies unit is treated as homogeneous in the assessment, with flow characteristics specified in terms of effective, averaged parameters. This is consistent with site characterisation studies which can only provide a high-level description of the subsurface stratigraphy in the region of the LLWR. We have undertaken work to consider the effects of heterogeneity on flows in the geosphere [7] and we have used the results of that work to define probability density functions (PDFs) for water flows in the B3 unit.

We have, where appropriate, used alternative approaches as follows.

- We have developed a 'dual porosity' model which explicitly models stacks of waste in the vaults. Whereas our main assessment model considers uniform advective flows of water and contaminants through a grouted waste mass, the dual porosity model considers diffusion of contaminants through the grouted waste mass to the gaps between the stacks, wherein the contaminants are advected in water flowing between the stacks. This model therefore allows us to explore spatial variability in waste properties, water flows and contaminant transport on the length scale of a stack. As the model considers stacks of ISO-freight containers and stacks of smaller SWTC-compatible strong boxes, it allows us to consider variability in diffusion length scales in the assessment.
- Degradation of the final cap is likely to display some spatial variability. This is for two reasons. Firstly, the cap will be installed in strips over the duration of disposal operations at the repository, which may be around a century. Thus, the first strips of cap will be around a century old when the last strips are installed. It is reasonable, therefore, to anticipate that strips of cap laid earlier will show signs of degradation earlier than those laid later. Secondly, degradation of the membrane in the cap is likely to commence at areas of localised high strain, before more widespread failure occurs [13]. Within the assessment we have explored the effect of such variability on system performance through the definition of a localised cap failure case (see Subsection 4.3).

4.1.8 Approach to FEPs

Identification of the FEPs relevant to the assessment of a disposal system, and assurance that the important processes and uncertainties have been adequately represented is central to confidence in the completeness of safety assessments. In the case of the LLWR, a high level of understanding of the relevant FEPs has been developed through iterative assessments and supporting studies carried out over the past two decades, and the pathways that are to be assessed are well known.

Thus, we considered it most appropriate to:

- use this understanding when developing our conceptual models and representing these conceptual models in assessment models;
- audit the resulting models against appropriate lists of FEPs.

The assessment was divided into two phases. At the end of Phase 1, an audit of biases and uncertainties was undertaken to identify whether any biases could be removed or uncertainties reduced. At the end of Phase 2, a further audit was undertaken to identify remaining biases and uncertainties.

4.1.9 Engineering Performance Assessment

We have undertaken a formal analysis of the evolution and performance of the repository's engineering as part of our EPA, which is documented in reference [13]. The EPA is an integral part of our assessment approach and considers both the PoA and the period thereafter.

Within the EPA we have:

- developed a conceptual understanding of the evolution and degradation of each component of the repository, including engineered barriers;
- considered and quantified uncertainties in the evolution of each component;
- identified parameter values which quantify the evolution of component performance and which we use in our assessment models and in supporting models such as our near-field and hydrogeological models;
- derived reference engineering evolution cases considering expected evolution for each component and for the repository system as a whole, which inform the assessment cases considered in our assessment models;
- derived alternative engineering evolution cases considering alternative plausible and low probability 'what-if' evolutions for each component and for the repository system as a whole, which inform the assessment cases considered in our assessment models.

4.2 Sources, Pathways and Receptors

4.2.1 Sources

The main potential source of contamination at the LLWR is the waste disposed to the trenches and the vaults, including past, ongoing and future disposals. The nature of these wastes is discussed in Subsection 3.1. Non-radiological hazardous substances and non-hazardous pollutants present in the waste are identified in Subsection 4.4 and the associated inventories are derived in reference [25]. The key aspects of the disposal system are discussed in Section 3.

The trench and vault wastes have the potential to generate leachate containing both radioactive and non-radioactive contaminants. Any water collecting in the vaults during the

PoA is treated as leachate and discharged via the marine pipeline. Leachate is collected via drains from the trenches and similarly discharged, but some leachate is released to groundwater through the trench bases. Such active management of leachate is projected to continue until the end of the PoA. The leachate management system is discussed in Subsection 3.4. Despite the leachate management system and the various other passive engineering measures discussed in Section 3, contaminant releases from the waste to groundwater beneath the site are expected, particularly in the long term after the end of management control. In Subsection 4.7 we quantify and provide understanding of the environmental impacts of such releases through a detailed performance assessment of the groundwater pathway.

Apart from the trench and vault wastes, other sources of non-radioactive contamination at the LLWR site include:

- the legacy of the ROF operations and buildings [3];
- storage of chemicals within the PCM magazines and adjacent area [3];
- potential contaminant migration from off-site, across the site boundary;
- LLWR operations other than waste disposal, including transport, waste conditioning and herbicide application;
- drainage systems and foul sewers.

As discussed in Subsection 1.2, the radiological and non-radiological impacts arising from legacy or more recent contamination have been assessed in the '*Environmental Safety During the Period of Authorisation*' [14] report and in a separate site characterisation report [23]. The quantitative risk assessments indicate that the risks from contaminated land at the LLWR are low given the current land use and management procedures that are applied. Our plans for the treatment of legacy or more recent contamination are set out in the '*Waste Management Plan*' [11]. They are not considered when assessing impacts to groundwater in this HRA.

4.2.2 Contaminant Release from the Near Field

Contaminant release and water flows in the near field

Contaminant release from the near field (i.e. the repository) occurs as a result of contact between infiltrating water and the wastes. Some contaminants may be readily available for leaching on contact with this percolating water, while others may only be available for leaching after the waste form in which they are bound undergoes a degradation process. For example, constituents of metal wastes are only released on corrosion of the disposed metal.

Water will enter the wastes principally through the cap, but lateral flows from the surrounding geology may be significant for the trenches. The cap and the cut-off wall are designed to limit these flows for as long as practicable, thereby limiting the release of contaminants. As these engineered features degrade, water flows through the near field will increase.

Within the EPA a significant amount of work was conducted to understand the evolution of the infiltration through the final cap [83]. As part of that work, we developed a numerical model to calculate infiltration through the cap. We considered the following when developing the model:

- the timing of geomembrane degradation and the elicitation of a PDF characterising the uncertainty in the time of geomembrane failure;
- the frequency of defects in the geomembrane, which takes into account the different types of defects, and their evolution with time;
- the evolution of the BES layer with time.

The numerical model produces a PDF of infiltration rates for different time snaps.

On the basis of the EPA [13], the final cap at LLWR is expected to perform effectively for a prolonged period, with the geomembrane component likely to last up to or beyond 2000 years. During this period, water flows through the cap will be localised to places where there are defects in the geomembrane [13]. Only waste stacks in the immediate vicinity of a defect will be exposed to infiltrating water [6].

The rates of water infiltration through defects will be very low [13] and most of the infiltration will be consumed by corrosion and waste degradation reactions. In the vaults, container corrosion will be the main sink for water, but water present in the grout will also be consumed in corrosion and degradation reactions, and consequently the wastes will desaturate. The lack of water flow means that contaminants will be retained in the wastes in this period.

As the cap degrades, water infiltration into the vaults increases and becomes more uniform over the vault area. The grout will begin to saturate, as water is drawn into the grout via capillary action. The high capillary suction of the grout means that no water will drain from the grouted wastes until the grout is fully saturated.

We expect this will only occur when the cap's performance degrades significantly. The paths of water flow in the vaults will depend upon the state of the containers.

- Surcharged wastes (Vault 8 and existing and committed LLW in Vaults 9 and 9a). These containers will have some degree of damage as a consequence of the surcharging process and it is likely that the upper container of each stack will have a damaged lid. We expect, therefore, that water infiltrating through the cap will enter the top container, saturating the grout and then any voids in the wastes. Once the container is fully saturated, water will be able to spill over and run down the sides of the lower containers in the stack. Water may be consumed in corrosion reactions with the metal of the container and, where lower containers are damaged, some water may enter these containers and saturate the grout.
- LLW in strengthened ISO containers and ILW which can be managed as LLW. The container protection units will divert flows of water into the gaps between the waste

stacks, where it may be consumed corroding the container external surfaces. The majority of these containers will only be able to take up water, other than possibly water vapour through the gas vents, when corrosion has perforated the walls so water flowing in the gaps cap can percolate into the grout. Given chemical conditions in the vaults, corrosion rates will be low [6] and therefore we expect that there will only be limited corrosion of containers. Water ingress into, and contaminant egress from, containers will therefore be limited.

- ILW in shielded modules. The shielded modules will act to divert the water into the drainage gaps between the bays. Containers in these modules will not experience water flows unless the module roof fails due to degradation of the reinforced concrete by the infiltration through the cap. Analysis in the EPA suggested that failures of the units would be rare before disruption by coastal erosion [13]. We would only expect, therefore, limited releases of contaminants from shielded modules.

Saturation of the grout and consumption of water in chemical reactions (e.g., corrosion of the containers and metal wastes) will therefore act as water 'sinks'. Water flows to the vault base and contaminant transport from the wastes will only be significant when there is excess water. That is, when the rate of water input into the system exceeds the rate of water consumption in these water sinks.

The release of contaminants from the disposed waste and their subsequent transport through the near field will depend on the ambient chemical conditions. It is expected that, for a significant period of time after repository closure (hundreds or thousands of years) the prevailing chemical conditions in the trenches will be reducing with neutral pH. In the vaults reducing conditions will be established within a few years and maintained until coastal erosion disrupts the cap. Alkaline conditions (a pH greater than 12) will be maintained in the grout pore water for thousands of years, while the pH in the gaps between the stacks will be 11 or lower [6].

In general, contaminants will sorb to available surfaces in the near field. Important substrates include soil, grout, concrete, the wastes themselves, and in the vaults the products from the corrosion of the waste containers. In the assessment model we:

- assume linear equilibrium sorption to soil in the trenches and grout in the vaults;
- ignore sorption to other surfaces.

The concentrations of some contaminants in solution also depend on their solubility limits in the chemical conditions of the trenches and the vaults. The chemical conditions which will be established in the vaults will limit the solubility, and favour the sorption, of key contaminants. This will limit the mobility of these contaminants in the near field.

Water levels in the near field

Once active leachate management ends, we expect that water flows into the repository will be very low for an extended period of time as the final cap will perform very well. Infiltration will increase as the cap degrades.

The resulting water levels in the vaults will depend upon the relative magnitudes of water flow into the repository minus the rate of water consumption, and the hydraulic conductivities of the various near-field barriers. In the assessment model we do not account for the consumption of water by corrosion or degradation processes. This is a cautious treatment.

Given anticipated low infiltration rates, we do not anticipate that water levels will rise significantly before the coastline reaches the site. We expect, therefore, that only a small proportion of waste containers will be in contact with standing water.

Our optimised approach to future vaults will ensure that, even if infiltration rates into the vaults increase significantly, water levels cannot rise above 1 m. Water that rises to the top of the 1 m vault walls will be directed to the drainage blanket under each vault and thence into the underlying geology. Vault 8 and 9 do not have under-vault drainage blankets. To minimise saturation over the height of the waste column in these vaults, passive drainage connections will be installed at closure at the 1 m level to allow water to flow into the future vaults and thence into the under-vault drainage blankets.

4.2.3 Pathways in the Geosphere

Our hydrogeological conceptual model is set out in Subsection 2.2.4 and remains consistent with that set out in the 2011 ESC and the 2017 HRA. Thus, as shown in Figure 2.6, once contaminants are leached from the near field, as described above, they are transported largely vertically through the B2 lithofacies unit, then largely horizontally through the B3 lithofacies unit being discharged eventually in local coastal waters or sediments.

As contaminants are transported through the geosphere, they will sorb to materials in the lithofacies units and will be subject to dilution and dispersion. In principle, they would also be subject to solubility limitation, although the contaminant concentrations in groundwater are expected to be sufficiently low that solubility limitation will not be relevant.

We have developed a regional-scale and a site-scale hydrogeological model which we have calibrated against onsite monitoring data [30]. We use this model to understand water flows and contaminant transport through the repository and the surrounding geology. The relationship between the hydrogeological model and the assessment model we use to calculate radiological and non-radiological impacts for the groundwater pathway is discussed in Subsection 4.6.

In Figure 4.2, we show pathlines calculated using the site-scale hydrogeological model superimposed over the lower lithofacies units. The pathlines describe the migration of contaminants based on advection alone, i.e. neglecting dispersion, sorption, radioactive decay, and ingrowth. We have set the pathlines to originate in several locations in the trenches and Vault 8. They therefore provide a first indication of the advective paths that would be followed by contaminants migrating from the waste.

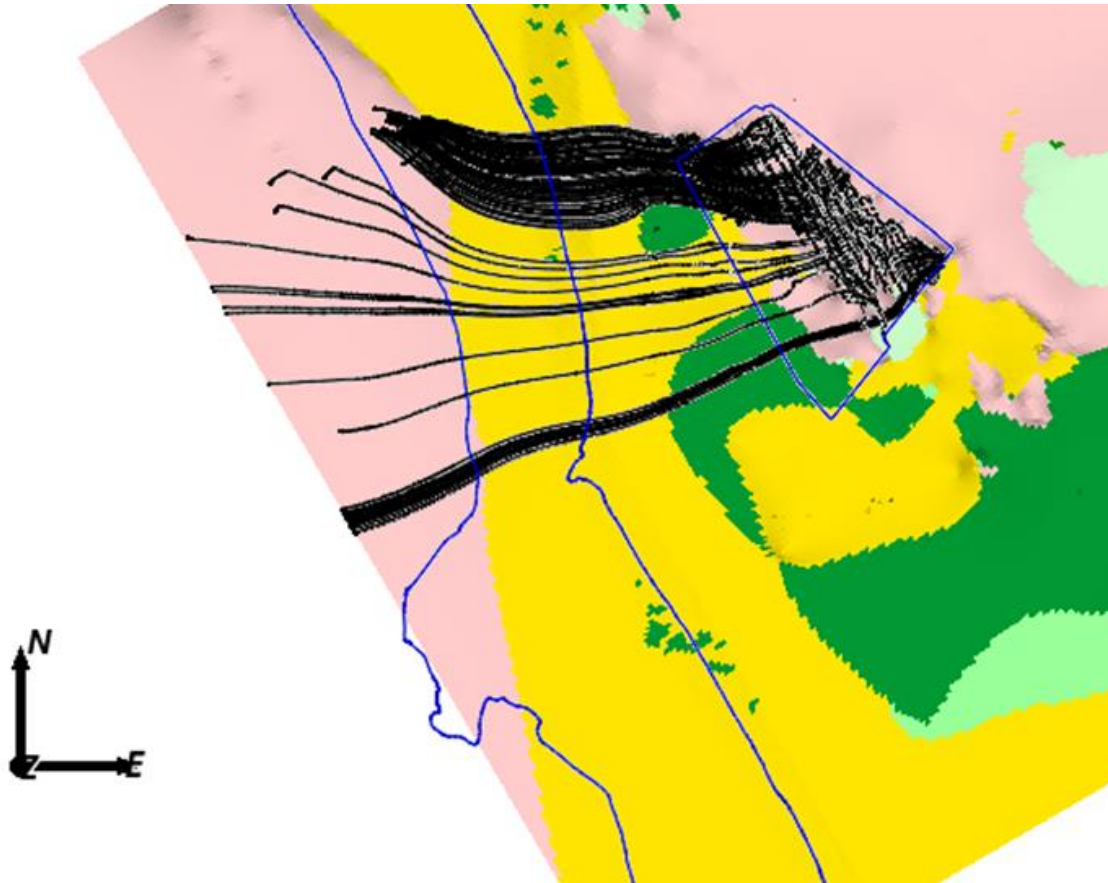


Figure 4.2: Pathlines calculated using our site-scale hydrogeological model [25]

The pathlines suggest that most of the groundwater travels from the LLWR through lithofacies unit B3 to the sea to the west of the site. Our hydrogeological modelling [25] indicates that the discharge point for contaminants is dictated by the offshore extent of the B2 lithofacies unit.

The calculated pathlines suggest that contaminants are not transported to the Drigg Stream or the East-West Stream via groundwater flows. Monitoring data, however, indicate low levels of tritium (less than 100 Bq l^{-1}) in the Upper Groundwater in the area to the south of the trenches and at two points in the surface water drainage system which discharge to the Drigg Stream [84]. There is uncertainty as to whether these derive from a leak in the leachate line at the southern end of the trenches or are evidence of a groundwater pathway between the disposal facility and the Drigg Stream. Extension of the cut-off wall as part of installation of closure engineering will remove this pathway.

As discussed in Subsection 2.2.2, in the past elevated levels of tritium were observed in the railway drain, which indicated the presence of a pathway from the trenches to the railway drain. Current and historical monitoring data from the railway drain show that tritium levels have significantly reduced in the years following the installation of the cap and cut-off wall to the extent that there is no longer a pathway between the trenches and the railway drain [85].

Our analysis of surface water features summarised in Subsection 2.2.2 indicates that other surface water bodies on the site are unlikely to be connected to groundwater.

4.2.4 Receptors and Compliance Points

The discussion in Subsections 2.2.2 and 2.2.4 indicates that groundwater beneath and down-gradient of the site is the main receptor for groundwater-borne contamination arising from wastes in the LLWR.

In line with regulatory guidance for landfill risk assessment [86], in Subsection 4.7 we calculate groundwater concentrations of the non-radioactive contaminants at compliance points in the geological units underlying the site. The locations of compliance points are different for hazardous substances and non-hazardous pollutants.

Compliance points for hazardous substances

For hazardous substances, there is one compliance point located below each trench and vault, in the B3 unit, adjacent to the edge of the discharge area for that particular trench/vault. The discharge area includes the immediate surrounding geology. Inclusion of the immediate surrounding geology ensures that all possible discharges into the B3 lithofacies unit are captured.

The situation is illustrated in Figure 4.3, which indicates the discharge area and its edge for a single trench or vault. The compliance point used in the assessment calculations is shown in red. Such location of compliance points is in keeping with the hydrogeological conceptual description (Subsection 2.2.4) of predominantly vertical groundwater flow down through B2 into B3, where flow is largely lateral towards the coast.

In calculating hazardous substance concentrations at these compliance points, account is taken of immediate dilution occurring after discharge from B2 to B3. Full dilution within B3 is not accounted for. An immediate dilution factor is calculated according to the groundwater flow across the width of the discharge area and within the vertical mixing depth, as described in reference [25]. The vertical mixing depth is shaded in blue in Figure 4.3.

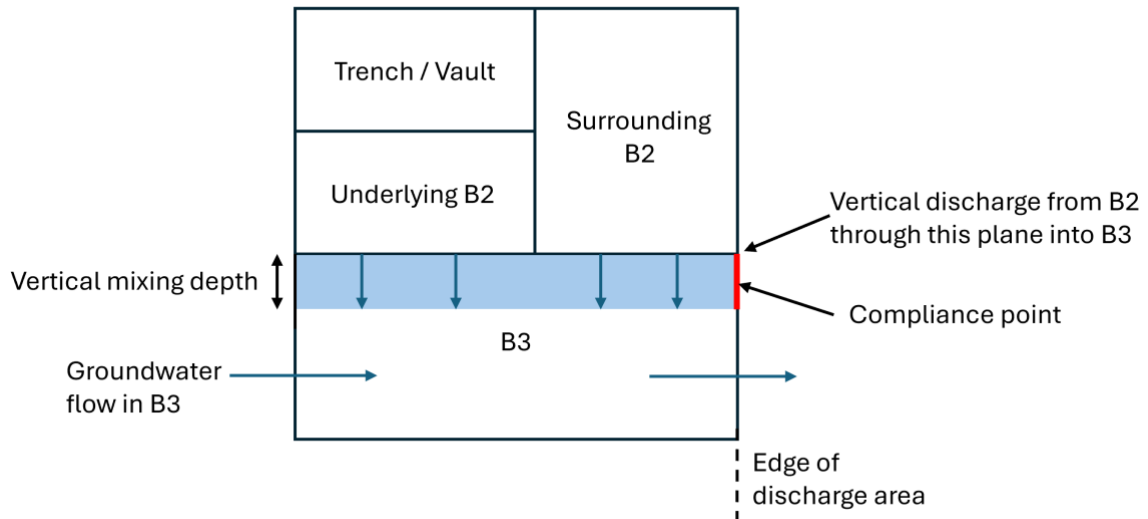


Figure 4.3: The discharge area and edge of discharge area for hazardous substances, taking into account the surrounding geology. In this figure, only a single block of surrounding geology is shown. For some trenches and vaults, there will be more than one region of surrounding geology to be included. The red line denotes the compliance point used in the assessment calculations.

Compliance points for non-hazardous pollutants

For non-hazardous pollutants, we calculate concentrations in lithofacies unit B3, directly below the site boundary. Although we refer to this as the ‘compliance point’ for non-hazardous pollutants, it is important to note that contaminant concentrations will vary along the width of the site boundary because of the differing contributions from the trenches and vaults across the width of the boundary. This is illustrated in Figure 4.4. For assessment purposes, we use the maximum concentration (at any given time) across all such points lying below the site boundary.

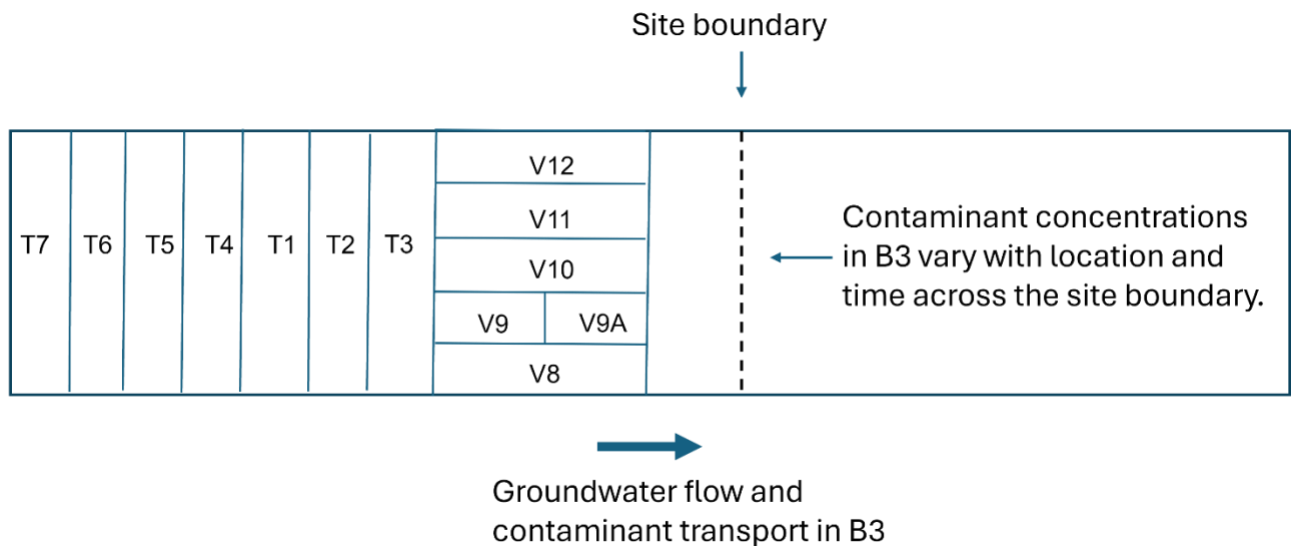


Figure 4.4 : Compliance points for non-hazardous pollutants

4.3 Assessment Cases

Drawing upon our understanding of the evolution of the natural environment and our system understanding, we have identified a set of cases which allows us to explore:

- the performance of the LLWR given the reference future evolution of the natural environment and the engineered barriers;
- the performance of LLWR given alternative future evolutions of the natural environment;
- the effect of uncertainties in the evolution of the engineered barriers on system performance;
- the effect of alternative representations of the near field on system performance;
- the effect of uncertainties in the geosphere on system performance;
- the effect of uncertainties in near-field properties on system performance;
- the contribution of the near-field chemical properties to retarding contaminant releases from the repository;
- the contribution of non-radiological contaminants present in the grout to assessed impacts.

These cases are listed in Table 4.1. For each case, we have undertaken deterministic calculations.

Table 4.1: Assessment cases

Reference	Title	Description	Comments
RC	Reference case	<p>The reference case which uses:</p> <ul style="list-style-type: none"> • sea-level rise and coastal erosion as set out in our reference emissions scenario; • evolution of the engineered barriers as set out in the reference engineering evolution scenario in the EPA; • best estimate values for parameter values. <p>The model considers advective transport of water flowing through the grouted waste mass in the vaults (see Subsection 4.6).</p>	
RC-DP	Dual porosity model	<p>As for case RC, but explicitly considers stacks of grouted waste containers and models diffusion of contaminants from the grouted wastes into gaps between containers where advective flows occur (see Subsection 4.6).</p>	
COAST-HI	Rapid coastal erosion	<p>As for case RC, but with coastal erosion assumed to occur according to the high emissions scenario.</p>	<p>These two cases explore uncertainty in the evolution of the environment in the vicinity of the LLWR.</p>

Reference	Title	Description	Comments
COAST-LO	Delayed coastal erosion	As for case RC, but with coastal erosion assumed to occur according to the low emissions scenario.	
DELAY_CE	Significantly delayed coastal erosion	As for case RC, but assumes that coastal erosion does not occur for at least 250,000 years.	A 'what if?' calculation which considers a climate change scenario that is not expected to occur in reality. It does, however, provide the opportunity to investigate the consequences of survival of the LLWR site in the long term
INFIL	Enhanced infiltration through the cap	As for case RC, but with enhanced infiltration through the cap overlying the trenches and vaults.	This case addresses alternative engineering evolution cases in the EPA which consider accelerated reduction in cap performance and major failure of the cap.
VBASE	Enhanced vault base permeability	As for case RC, but with enhanced permeability for the vault bases and walls.	This case addresses alternative engineering evolution cases in the EPA which consider faster than expected deterioration or more extensive failure of the vault base.
CLOGGING	Earlier or greater than expected clogging of the	Case to consider situations where clogging of the under-vault drainage blanket is earlier or to a	This case addresses alternative engineering evolution cases in the EPA

Reference	Title	Description	Comments
	under-vault drainage blanket	greater extent than expected. Considered to be a lower probability case.	which consider earlier than expected, or worse than expected clogging.
GROUT	Non-radiological contaminants in grout	As for case RC, but considering only the inventory of non-radiological contaminants associated with grout in the vaults.	This case requires an adaptation of the vault near field model to represent leachate concentrations in grout. The case does not consider non-radiological contaminants present in the wastes.
CHEM	Near-field chemical barrier	As for case RC, but with zero sorption and unlimited solubility in the trenches and vaults.	This case investigates the effectiveness of the near-field chemical conditions in retarding contaminant releases from the repository.
GEO_LOW_SORPTION	Geosphere chemical conditions	As for case RC, but with zero sorption in B2 and B3 lithofacies units.	This case investigates the contribution of the geosphere in limiting contaminant releases.
COMPLEX1	Uncertainty in complexant loading	As for case RC, but with complexant inventory enhanced by an order of magnitude.	
COMPLEX2	Complexants in the geosphere	As for case RC, but with no complexation in the geosphere.	Addresses uncertainty in the degree of complexation expected once contaminants leave the near field.

Reference	Title	Description	Comments
HYDRO-HI	Hydrogeological parameters - increased flow	As for case RC, but with groundwater flow rates in B3 increased by 3 standard deviations (1.5 orders of magnitude).	These cases consider uncertainties in the hydrogeological characteristics of flow in the B3 lithofacies unit.
HYDRO-LO	Hydrogeological parameters - reduced flow	As for case RC, but with groundwater flow rates in B3 reduced by 3 standard deviations (1.5 orders of magnitude).	
CORROSION	Uncertainty in metal corrosion rates	As for case RC, but with enhanced metal corrosion rates in the trenches and vaults.	The metal corrosion rate governs the release of non-radiological contaminants from metals into pore water. This case explores the significance of uncertainty in corrosion rates for system performance.

4.4 Non-radiological Contaminants to be Modelled

The non-radiological contaminants considered in the HRA have been chosen on the basis that they:

- are explicitly declared in historical or future waste streams; or,
- are considered to be present in declared bulk materials; or,
- are known to be present in grout.

In determining whether contaminants are considered to be present in declared bulk materials, we have considered whether they have been detected in trench leachate or other site characterisation studies.

In some cases, representative contaminants have been used in place of groups of chemically similar substances, for example vinyl chloride is used to represent chemically similar organohalogen compounds, as discussed in reference [25]. This is based on practical considerations, since it would not be feasible to represent the full suite of organohalogen compounds explicitly.

We laid out our approach to identifying contaminants to assess in our previous HRA [44]. Given updates to our understanding of the inventory, including assessing potential ILW disposal in the ESC, we have undertaken a review to determine whether there are additional contaminants which should be included within the assessment [87].

The resulting set of contaminants considered is present in Table 4.2 and Table 4.3. These tables indicate whether a particular contaminant is included on the basis of the declared inventory and/or whether it is known to be present in grout.

We have not assessed contaminants which have been detected in leachate, but for which we do not have inventory data. This is because our assessment approach requires inventory data to define our source term. As discussed in Subsection 4.1.1, we do not judge leachate data an appropriate basis on which to determine a source term for use in our assessment of non-radiological impacts from the wastes.

While we cannot calculate impacts for contaminants for which we do not have inventory information, we can calculate non-radiological capacities for these contaminants. This is because the determination of a capacity (i.e., the maximum quantity of a contaminant which could safely be disposed of) does not require inventory information. The magnitudes of the calculated capacities provide a measure of the impact associated with a contaminant - the smaller the impacts, the greater the capacities. Capacities are used where appropriate to manage the disposal of non-radiological contaminants and ensure that assessed impacts remain less than the relevant LLWRAS. This means that, even if we do not have inventory information for a contaminant of interest, we can ensure that we protect groundwater from any future disposals of that contaminant.

Our approach to calculating capacities, along with the calculated capacities, is described in more detail in references [17, 25].

Table 4.2: Hazardous substances considered in the HRA

Contaminant	Declared in inventory?	Measured in leachate?	Present in grout?	Comments
Arsenic	Yes	Yes	Yes	
Benzo-a-pyrene (BAP)	Yes	Yes	No	Taken to be representative of polycyclic aromatic hydrocarbons present in tarmac, asphalt, bitumen and coal tar.
Benzene	Yes	No	No	Taken to be representative of non-halogenated aromatic organic compounds. In our assessment we assume that condensation polymers, thermoplastics, non-halogenated plastics, non-halogenated rubber, unknown rubber, hydrocarbons, organic ion exchange resins and unknown organics are comprised entirely of benzene.
Chromium (VI)	Yes	No	No	
Lead	Yes	Yes	Yes	
Mercury	Yes	Yes	No	

Contaminant	Declared in inventory?	Measured in leachate?	Present in grout?	Comments
Tributylphosphate (TBP)	Yes	Yes	No	TBP is taken to be representative of organophosphorous compounds.
Vinyl chloride	Yes	Yes	No	Vinyl chloride is taken to be representative of chemically similar organohalogen compounds detected in leachate. In our assessment we assume that halogenated plastics and rubber degrade to 1 ppm vinyl chloride.

Table 4.3: Non-hazardous pollutants considered in the HRA

Contaminant	Declared in inventory?	Measured in leachate?	Present in grout?	Comments
Aluminium	Yes	No	Yes	
Ammonium	Yes	Yes	No	
Antimony	No	Yes	Yes	
Barium	Yes	Yes	No	
Beryllium	Yes	Yes	No	

Contaminant	Declared in inventory?	Measured in leachate?	Present in grout?	Comments
Boron	Yes	Yes	Yes	
Cadmium	Yes	Yes	Yes	
Chromium(III)	Yes	Yes	Yes	
Cobalt	Yes	Yes	No	
Copper	Yes	Yes	No	
Fluoride	Yes	Yes	No	
Iron	Yes	Yes	Yes	
Magnesium	Yes	No	No	
Molybdenum	Yes	Yes	Yes	
Nickel	Yes	Yes	Yes	
Phenol	Yes	Yes	No	Phenol is taken to be representative of persistent hydrocarbons and bioaccumulable organic toxic substances. In our assessment we assume that ion exchange materials are comprised entirely of phenol.

Contaminant	Declared in inventory?	Measured in leachate?	Present in grout?	Comments
Phosphate	Yes	Yes	No	
Selenium	Yes	Yes	Yes	
Tin	Yes	Yes	No	
Titanium	Yes	Yes	No	
Uranium	Yes	Yes	No	
Vanadium	Yes	Yes	Yes	
Zinc	Yes	Yes	Yes	

For each contaminant declared to be in the inventory, we require a contaminant mass in order to be able to carry out a quantitative assessment. Using the Stage 2 inventory reported in reference [67], we obtain contaminant masses using:

- declared masses of contaminants where available;
- a set of assumptions as to the composition of bulk materials (e.g., we assume that stainless steel in the inventory is composed of 65% iron, 18% chromium, 14% nickel and 3% molybdenum).

The assumed material compositions, along with the approach used to derive the contaminant masses, are provided in reference [25] and remain the same as those used in the 2011 ESC and the 2017 HRA. The resulting inventory which we have used in this HRA is also presented in reference [25].

Our grout source term derives from column leaching experiments carried out by the Buildings Research Establishment on pulverised fuel ash (PFA) and ordinary Portland cement (OPA) grouts which we consider to be reasonably representative of LLWR grout [88]. Reference [88] presents data for contaminant concentrations in grout against the total volume of water that is flushed through the grout samples.

In the current assessment, we have set the mass of contaminants in the grout to be a nominally very high value and we have also set solubility limits for contaminants in grout to be equal to the peak concentrations observed in the leach tests. We have cautiously treated these solubility limits as constants over the assessment time frame. This approach ensures the maximum calculated pore water concentration is equal to the peak observed leachate concentration.

4.5 Review of Technical Precautions

Operations and facilities at the LLWR have been optimised [10] to minimise the environmental impacts resulting from release of radionuclides. The precautionary measures taken will be equally effective in limiting the release of non-radioactive contaminants. The technical precautions include aspects of the trench and vault disposal systems, cap, cut-off wall and the leachate management system, as discussed in Section 3.

It is our contention that the level of protection provided by the LLWR is no less stringent than would be provided under the regulations pertaining to hazardous wastes. Key aspects of the disposal system design in this regard were discussed in Subsection 3.6.

4.6 Numerical Modelling

4.6.1 Overview of Assessment Model

We have used the conceptual description of the system provided in Subsection 4.2 to develop an assessment model which considers:

- infiltration of water into the repository through the engineered cap and laterally through the cut-off wall and trench and vault walls;
- changes in water levels in the repository;
- release of contaminants from the waste to porewater subject to solubility limitation;
- contaminant transport through the near field, where contaminants are subject to reversible linear sorption, and we account for the effects of complexing agents on contaminant mobility;
- vertical transport of contaminants in the B2 lithofacies unit, where contaminants are subject to reversible linear sorption and complexation, which affects mobility;
- horizontal transport of contaminants in the B3 lithofacies unit, where contaminants are subject to reversible linear sorption and complexation, which affects mobility.

The assessment model is used to model releases over the full lifetime of the repository, from the first emplacement of waste in the late 1950s through to disruption of the facility by coastal erosion. Construction of, and disposal of waste to, each trench and vault in the repository is represented within the model. We use a simple approach in which inventory is added linearly over the operating lifetime of a trench or vault. We discuss the operational lifetime of each trench and vault in Subsection 4.6.5.

For non-radiological contaminants, we have used the assessment model to calculate contaminant concentrations at the compliance points discussed in Subsection 4.2.4. We have not therefore modelled non-radiological contaminant transport through the biosphere.

For radionuclides, the appropriate compliance points are in the biosphere (see reference [14] and reference [15]). We have used simple linear models to calculate the transfer of radionuclides through the biosphere and to calculate the incurred radiation doses and risks. The data required to carry out these calculations (e.g., habits data, sorption coefficients for biosphere environmental media, transfer factors and concentration ratios) have been identified based on the characterisation of the biosphere.

The assessment model has three component parts:

- the near-field model, which calculates releases of contaminants from the wastes and transport of contaminants through the near field;
- the geosphere model, which models transport of contaminants through the geosphere and the resulting porewater contaminant concentrations;
- the biosphere model, which models transfer of radionuclides through the biosphere and calculates the resulting radiation doses and risks to exposed groups.

The assessment model has been implemented within GoldSim. This provides a flexible framework for modelling various systems including repositories for radioactive waste. GoldSim is widely used within the industry to assess the radiological and non-radiological impacts arising from disposal facilities. Within the 2026 ESC, we have used GoldSim to

assess the impacts arising from the coastal erosion, gas and human intrusion pathways, in addition to using it to assess radiological and non-radiological impacts to groundwater.

The GoldSim assessment model makes use of outputs from other models as follows.

- PFLOTRAN. We have used PFLOTRAN to model the evolution of pH and Eh over the assessment time frame. This understanding has informed our view of complexing agents within the vaults and has informed our identification of appropriate corrosion and sorption and solubility parameters in the vaults and trenches (see Subsection 4.6.5). We have not directly used PFLOTRAN outputs (e.g., contaminant fluxes) in the assessment model.
- A finite element groundwater flow model [30]. A detailed model of the area has been developed using the ConnectFlow finite element software [89]. This integrates the geological model with understanding of the site hydrogeology and is calibrated against a range of different information sources including groundwater levels from boreholes, trench probes, stream flow gauges, trench leachate flows, and transient fluctuations in water levels. The hydrogeological model is implemented on two levels: a large-scale regional model, and a site-scale model which includes a representation of the repository engineering for the purpose of modelling groundwater flows in and around the repository. The lateral boundary conditions of the site-scale model are derived from the output of the regional-scale model. We have used outputs from the ConnectFlow models in the following ways.
 - We have calculated flowpaths in the ConnectFlow models to provide a first indication of the advective paths that would be followed by contaminants migrating from the waste. This understanding has informed our development of the GoldSim assessment model.
 - Flows in the B3 unit calculated by the regional-scale ConnectFlow model have been used directly by the assessment model (see Subsection 4.6.5) to calculate contaminant transport through the B3 unit.
 - The site-scale ConnectFlow model is too complex for implementation in an assessment model and we have therefore developed a simplified representation of the hydrogeology suitable for inclusion in the assessment model (the Compartment Flow Model, which we discuss in following bullet point). We have used the site-scale ConnectFlow model to provide boundary conditions for the Compartment Flow Model within the near field and in the B2 lithofacies unit and to verify levels and flows calculated within the Compartment Flow Model.
- The Compartment Flow Model [30]. This is simplified model of the hydrogeology which has a smaller domain than the finite-element model, extending vertically from the repository down to the B2 geological layer and laterally to a small distance outside of the cut-off wall. The model is constructed as a system of inter-linked compartments, each representing a particular vault, part of a trench or part of the

surrounding or underlying geology. The equivalence between the Compartment Flow Model and assessment model compartments allows the Compartment Flow Model to be embedded directly within the assessment model. At each timestep, the assessment model calls the Compartment Flow Model to calculate water flows between pairs of compartments and the water levels in the repository near field and in the B2 lithofacies unit. The calculated flows and water levels are used to determine the transport of contaminants through the repository near field and through the B2 lithofacies unit.

This relationship of the non-radiological assessment model to the other models outlined above is shown in Figure 4.5, where the arrows denote inputs into the assessment model. We emphasise that the arrows do not always represent 'direct' modelling relationships. Often, we apply expert judgement in extracting the results of one model and translating them into an input for another model.

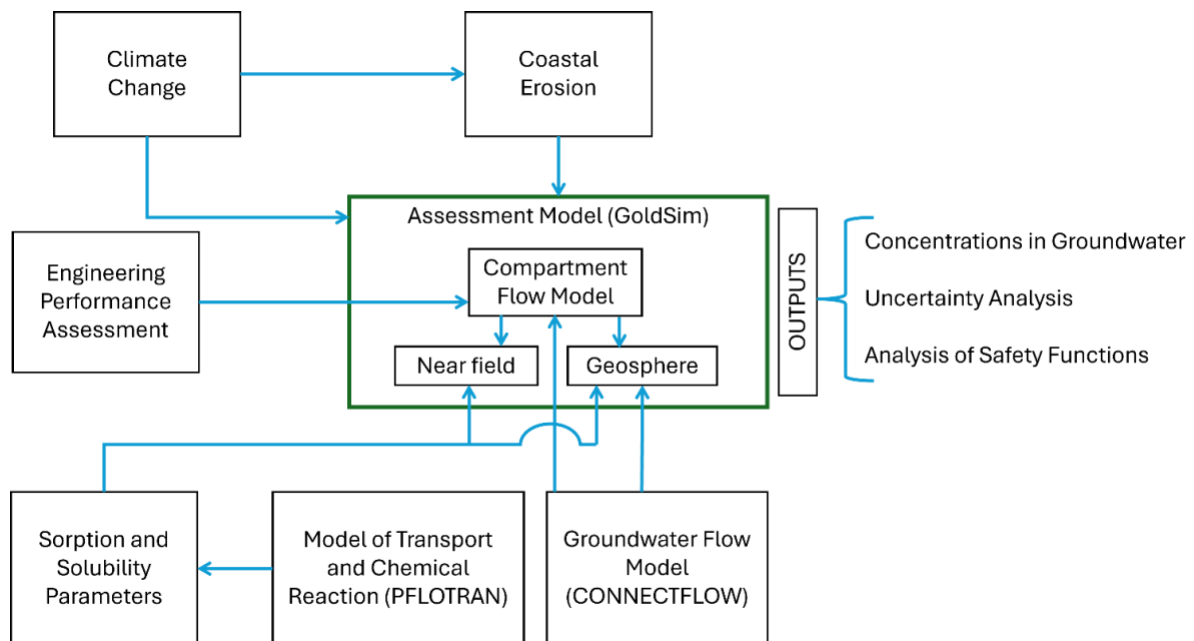


Figure 4.5: Model structure in the non-radiological assessment model and relationship to other models

We briefly discuss the components of the assessment model in the following subsections.

4.6.2 Near-field Model

Within our assessment model we have two different representations of the vault near field.

- A 'uniform flow' model, which is a development of the model used in the 2011 ESC and in the 2017 HRA, and which models flows of water, and advective transport of contaminants, through connected pore space in well-mixed compartments. Contaminants are uniformly distributed through these compartments.
- A 'dual porosity' model, in which we explicitly represent stacks of grouted waste in the vaults. In this representation, water flows between the stacks of waste, rather than

through the pore space of the grouted waste mass itself. Contaminants diffuse through the grouted waste mass to the gaps between the stacks, where they are entrained in infiltrating water and transported to the vault base and thence to the underlying geology.

Only the treatment of the waste within the vault near field differs in the two models.

The dual porosity model provides a representation of the near field that is more consistent with our view of water flows in the vaults (see Subsection 4.2.2) than the representation in the uniform flow model. The dual porosity model, however, assumes that all waste containers are well-grouted. Given the known variability in the quantity of grout present in containers which have been disposed of in the repository, our view is that the uniform flow model provides a more appropriate basis for a reference case. The dual porosity model nevertheless allows us to explore the effects of a different near-field representation on assessed impacts.

We summarise the two models in the following subsections.

Uniform flow model

Within the uniform flow model, we represent each vault as one region, while the trenches are presented as between four to six distinct regions. The level of discretisation is dictated by a requirement to represent explicitly the key engineered features of the repository and also allow the representation of progressive capping within the assessment model.

Each region is assigned average physical and chemical properties. These properties reflect the composition and proportions of the waste and soil (for the trenches) and grout (for the vaults) assumed to be present (or anticipated to be present) in the region.

Thus, for the vaults, sub-vault heterogeneities in physical and chemical properties are not considered. Different waste types are not explicitly represented within the assessment model. The level of discretisation means that the assessment model does not treat individual containers or stacks of containers.

In each region of the repository the saturated and unsaturated zones are each modelled as a well-mixed compartment. The height of the water table, and hence the volume of each compartment, is determined by the Compartment Flow Model. Each compartment can be considered to be a finite volume through which contaminants are uniformly mixed. Variations in contaminant concentrations within a compartment cannot therefore be modelled, although variations in time are modelled.

We use first-order rate constants to determine the release of contaminants from the wastes. Once released from the wastes, contaminants are treated as being instantaneously available for dissolution into porewater. In the model we assume that contaminants are able to access the entire water-filled void space available within the trenches or vaults. These are simple and cautious assumptions, which are adopted because there is insufficient knowledge of the characteristics and distribution of contaminants throughout the wastes to support a more detailed and sophisticated treatment.

Some contaminants are assumed to be released sufficiently rapidly from the bound state that the release can be considered to be instantaneous. This is represented by assigning a nominally high rate constant for such releases. Other contaminants are assumed to be released over longer timescales, determined by the corrosion properties of metallic wastes in which they are present.

Contaminant concentrations in porewater are controlled by solubility limitation and sorption, where applicable. Solubility sets a maximum concentration for each relevant contaminant in porewater according to the chemical conditions. Sorption of dissolved contaminants to solids within the near field (soil in the trenches and grout in the vaults) acts to reduce their concentration in groundwater. Organic complexants present in the waste, such as ethylenediaminetetraacetic acid (EDTA) or isosaccharinic acid (ISA), are considered to increase contaminant mobility, by increasing effective solubility or reducing effective sorption.

Once released to porewater, contaminants are transported through the near field via advection. Flows of water are calculated using the Compartment Flow Model. Flows within the unsaturated compartment are vertical, while in the saturated compartment flows may be:

- laterally either to saturated compartments in neighbouring trenches or vaults, or to neighbouring geology compartments; or,
- vertically downwards into the underlying B2 unit.

Dual porosity model

In the dual porosity model, we explicitly represent stacks of waste in the vaults. Each stack is represented as a series of concentric, well-mixed compartments, throughout which contaminants are uniformly distributed.

As the gaps between the stacks are inter-connected, we model them as one single compartment. Water flows through, and water levels in, this compartment are calculated using the Compartment Flow Model. We do not consider water flows through the waste stacks. This is because, as discussed below, the grout is modelled as fully saturated throughout the assessment time frame and water flows will occur preferentially through the inter-stack gaps as these are more transmissive. Contaminant transport through the grouted waste therefore occurs solely by diffusion.

The grouted waste stacks are treated as being fully saturated throughout the assessment time frame. The evolution of saturation in the grout will be complex, as it will depend upon:

- the initial saturation of the grout;
- the quantity of grout in each containers;
- the nature of the waste in each container;
- the quantity of water flowing down a stack;
- the physical state of the containers.

It is likely therefore to exhibit variability across the vaults, particularly when the geomembrane is functional and water flows occur through defects in the membrane. It is not feasible, or necessary, to represent this degree of variability in the assessment model. Assuming complete saturation is a simplifying assumption which avoids this complexity. It is a cautious approach in two respects:

- it maximises diffusive fluxes of contaminants in the grout;
- it removes a water sink from the system, increasing the quantity of water that reaches the vault base.

The model also includes the following cautious assumptions:

- we do not represent the waste containers within the assessment model;
- we apply a zero concentration boundary condition in the gap.

These are cautious because they will maximise the diffusive flux from the grouted waste into the gap.

The release of contaminants from the waste to porewater is treated in the same way as in the uniform flow model. This includes:

- control, where appropriate, of contaminant concentrations in grout porewater by solubility limitation and sorption;
- considering the effects of complexation on contaminant mobility.

Once released to the gap between the stacks, contaminants are transported in advective flows of water, calculated using the Compartment Flow Model. Flows may occur:

- laterally either to saturated compartments in neighbouring trenches or vaults, or to neighbouring geology compartments; or,
- vertically downwards into the underlying B2 unit.

Representation of near-field engineering

The assessment model represents the near-field engineering implicitly through the use of water flows and levels calculated by the Compartment Flow Model. The Compartment Flow Model explicitly represents the profile fill as well as the following components of the near-field engineering:

- the trench drains;
- the trench bases²⁴;
- the cut-off wall;
- the Vault 8 concrete base slab and underlying drainage layer and BES;

²⁴ The trench walls are not represented within the Compartment Flow Model.

- the secant pile wall along the eastern edge of Vaults 8 to 12;
- the Vault 9 concrete base slab and underlying geomembranes and BES layers;
- the 1 m-high BES lining in Vault 9;
- hydraulic connections between Vault 8 and 9a and between Vault 9 and 9a at the 1 m level;
- the concrete base slabs and underlying BES in Vaults 10 to 12;
- the internal 1.1 m-high walls between Vaults 10 and 11 and between Vaults 11 and 12;
- the external 1 m-high walls for Vaults 10 to 11;
- the drainage blankets underneath Vaults 9a to 12 and the free-draining material behind the 1 m-high side walls which provides hydraulic connectivity to the under-vault drainage blankets.

Using the outputs of the EPA [13] we represent the degradation of these engineered features by modifying their hydraulic properties over the assessment time frame.

The vault leachate management system is represented implicitly in the Compartment Flow Model as described in reference [25]. Similarly, the interim trench cap and final cap are implicitly represented as boundary conditions determining the quantities of water which will enter the upper surface of the Compartment Flow Model. They are treated as follows.

- The infiltration rate through the interim cap is determined based upon HER and the cap efficiency, which is inferred from our monitoring work. Cautiously, we do not represent the upgrade of the interim trench cap in the model and therefore the cap efficiency is taken to be constant over the assessment time frame. This approach will result in higher releases of contaminants from the trenches to groundwater than we anticipate will occur after the interim trench cap is upgraded.
- As part of the EPA, we have considered how the hydrological performance of the different layers of the final cap will evolve with time. We have used this to develop a model which allows us to calculate the amount of water which will be able to enter the repository through the final cap [13]. Infiltration rates used in the assessment are reported in reference [25]. As the final cap degrades over the assessment time frame, the quantities of water calculated to infiltrate through the final cap increase.

Progressive construction of each trench and vault and installation of the interim cap, final cap and extension of the cut-off wall over the lifetime of the LLWR are represented within the model. As we model the progressive installation and degradation of engineered features, water flows into and out of the repository will vary with time over the course of the assessment.

4.6.3 Geosphere Model

The geosphere model includes a representation of the B2 and B3 lithofacies units, which we discuss in turn in this subsection. The geosphere model is the same for both the uniform flow and dual porosity models.

Within the assessment model, we represent:

- volumes of the B2 unit that are adjacent to the trenches and vaults;
- volumes of the B2 unit that underlie the trenches and vaults.

These regions are shown schematically in Figure 4.3.

Each volume of the B2 unit is modelled as a stack of five well-mixed compartments. This ensures that the degree of contaminant dispersion during transport through the B2 stack is reasonable (it corresponds to a Peclet number for transport of around 10). This contrasts with the approach adopted in the 2011 ESC, in which only a single compartment was used to represent the lithofacies unit B2 below each vault or trench. The use of a single compartment leads to a highly dispersive system in which some contaminants are transported through the underlying B2 much more rapidly than the expected retarded travel times.

In contrast to the trenches and vaults, all compartments within the B2 stack are assumed, for simplicity, to have the same saturation value. This is computed by calculating the total amount of water present in the stack (information that is passed from the Compartment Flow Model) and assuming that this water is uniformly distributed through the volume of the stack.

Contaminants can enter the B2 stack in one of two ways:

- in vertical flows from the overlying trench or vault;
- in lateral flows from neighbouring trenches, vaults or neighbouring underlying B2 stacks.

For both the inward and outward lateral flows, the lateral flows are assumed to be distributed uniformly across the depth of the B2 stack. This approximation is necessary because the Compartment Flow Model does not resolve the underlying B2 geology in the same way as in the assessment model. Instead, the Compartment Flow Model considers the underlying geology under a particular trench or vault as a single compartment.

Water flows into and out of the B2 stack are determined by the Compartment Flow Model. Discharge from the base of the B2 geology stack is into the underlying B3 lithofacies unit.

Within the B2 unit, we model the effects of dispersion, radioactive decay and ingrowth. We model sorption to the geosphere as a linear, reversible process, modified by the effects of organic complexants as appropriate (see Subsection 4.6.5). Contaminant concentrations in the B2 unit are sufficiently low that solubility limitation is not relevant.

Groundwater flows in the B3 unit are predominantly horizontal, directed from the hills of the Lake District to the Irish Sea (see Subsection 2.2.4). We therefore model the B3 unit as a

series of 'pipe' elements in GoldSim. In each element the assessment model solves a one-dimensional advection-dispersion equation in which we model:

- radioactive decay and ingrowth of radionuclides;
- sorption to solids in the geosphere, which is treated as a linear equilibrium process modified, where appropriate, by the effects of organic complexants;
- advection and longitudinal dispersion of contaminants through the pore space of the lithofacies unit.

The approach neglects:

- solubility limitation in the B3 unit as the concentrations of contaminants are too low for this to be relevant;
- transverse dispersion, which is a cautious treatment;
- diffusion of contaminants in porewater.

For the assessment calculations, the B3 unit is represented at a sufficient resolution to consider separately the passage of contaminants from each trench and vault and to enable concentrations of hazardous substances and non-hazardous pollutants to be evaluated at all of the required compliance points, as identified in Subsection 4.2.4. The discretisation of the B3 unit into pipes is shown in Figure 4.6. In this figure, the groundwater flows (and hence contaminant movements) are taken to be from left to right.

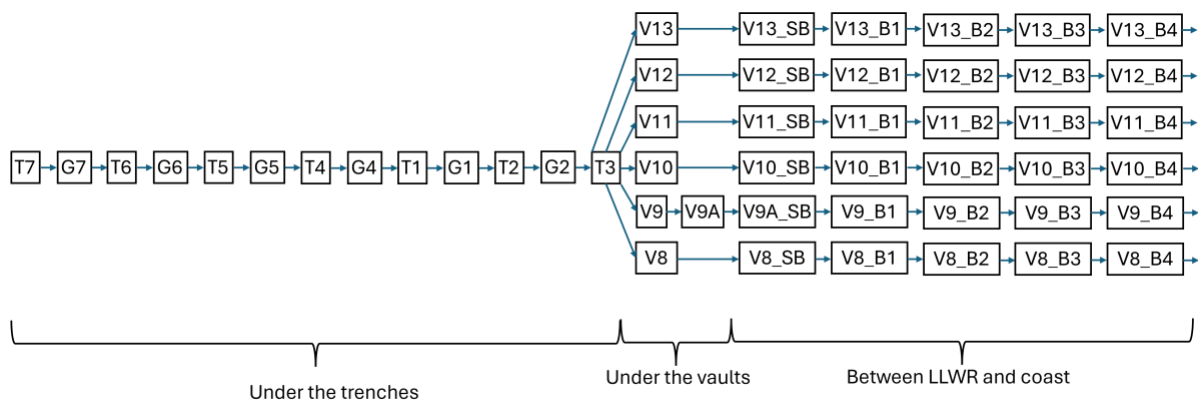


Figure 4.6: Structure of the geosphere (within the B3 lithofacies) within the assessment model. The volume that a possible V13 would occupy is included in the model – at the surface it contains only B2 material (no waste) and below the surface it is modelled as B2 and B3 as for the other vaults. The cells labelled e.g. “V8_B1” and “V8_B2” represent “blocks” in the geosphere that are numbered 1 to 4 (rather than the B2 or B3 lithofacies).

In the assessment we treat both the B2 and B3 units as having homogeneous properties that represent average bulk flows and transport through the groundwater system.

4.6.4 Biosphere Model

The biosphere constitutes that part of the accessible environment where humans and non-human biota are present. Our compliance points for non-radiological contaminants (see Subsection 4.2.4) are located within the geosphere and consequently we do not model transfer of non-radiological contaminants through the biosphere. Notwithstanding this, our assessment model includes a biosphere model as we use the same model to calculate radiological and non-radiological impacts. We provide a brief overview of the biosphere model in this subsection. More detail on the biosphere model can be found in reference [24].

The components of the biosphere relevant to waste disposals at the LLWR are determined by the expected migration pathways for contaminants away from the LLWR and into the surrounding environment. This is informed by our ConnectFlow modelling.

Within the assessment model, the biosphere is modelled as a series of connected, well-mixed compartments. Each compartment is treated as being composed of water and relevant solid materials (soil, sand, sediment). Linear equilibrium sorption to solid materials is assumed. Contaminant transport between compartments is specified in terms of first-order rate constants. These rate constants are determined by considering expected rates of water and solid material transfer between compartments.

For all radionuclides except C-14 and tritium, radionuclide transfer through food chains is modelled using concentration ratios and transfer factors. For C-14 and tritium, as discussed in references [24, 90], we use specific activity models to calculate radionuclide uptake.

4.6.5 Model Parameterisation

In this subsection we provide an overview of the data we have used in the HRA.

Timescales

The timescales for construction and operation of the LLWR include the time of construction of the trenches and vaults, the closure timescales of the trenches and vaults, and the timing of cap emplacement. The timescales for these activities are informed by our understanding of the reference inventory and engineering judgement as to the length of time required to install strips of the final cap. Reference [91] sets out the relevant timescales and their justification and we present the timescales in Table 4.4. The cut-off wall will be extended consistent with the capping schedule laid out in Table 4.4 and we assume that the area to the south and south-east of Vault 12 will be capped at the same time as Vault 12.

We have concluded, based on a review documented in reference [63], that the site will be under active institutional control for a period of 100 years after completion the final cap. Our current intent, therefore, is that the PoA will end in 2242.

Table 4.4: Key dates during the PoA (AD)

Element Name	Build dates	Beginning of disposal	End of disposal	Interim cap build dates	Final cap dates
T1	1959	1959	1963	1989	Phased
T2	1963	1963	1973	1989	Phased
T3	1973	1973	1979	1989	Phased
T4	1979	1979	1981	1989	Phased
T5	1981	1981	1983	1989	Phased
T6	1983	1983	1985	1989	Phased
T7	1985	1985	1995	1995	Phased
V8	1988	1988	2018	N/A	2036
V9	2008	2008	2047	N/A	2057
V9A	2008	2008	2047	N/A	2057
V10	2051	2047	2090	N/A	2097
V11	2094	2090	2113	N/A	2118
V12	2116	2113	2135	N/A	2142

Timescales relevant to the onset of coastal erosion are discussed in Subsection 4.1.4.

Inventory

The derivation of contaminant inventories is discussed in detail and presented in reference [4].

Chemical properties

The two chemical properties of interest in the near field are sorption coefficients and solubility limits. The data used in the assessment are provided in reference [25].

In the near field, sorption coefficients are required for soil in the trenches and grout in the vaults. Sorption to concrete, corrosion products or waste is neglected. We have undertaken reviews of data used in previous assessments and published scientific data to identify sorption parameters appropriate for the chemical conditions we anticipate will be established in the trenches (references [92] and [93]) and vaults [94] for the duration of the assessment timeframe [6]. These reviews have also identified appropriate sorption data for the B2 and

B3 lithofacies units. Our review indicates that available data do not allow the identification of different sorption parameters for the B2 and B3 units. We therefore use the same values for both.

References [92] and [93] also document our review of solubility data, which we have used to identify solubility limits appropriate for the chemical conditions in the trenches. Solubility data for the vaults are documented in reference [95]. Contaminant concentrations in the geosphere will be sufficiently low that solubility limitation is not important. We have therefore neglected solubility limits in the geosphere in our assessment.

We have treated sorption and solubility parameters as constant over the assessment time frame, which is appropriate given the available understanding of chemical evolution over the assessment timescale.

Solubility enhancement and sorption reduction factors

Solubility enhancement factors (SEFs) and sorption reduction factors (SRFs) are factors that modify solubility limits and sorption coefficients, to take account of the presence of complexing agents in the disposed inventory. SEFs and SRFs vary between chemical elements, complexing agents and complexant concentration.

Effective solubility limits in the presence of organic complexants are obtained by multiplying the solubility limits by the SEFs. The SRFs are factors that divide sorption coefficients to obtain effective sorption coefficients in the presence of organic complexants. An SEF or SRF of one indicates that a particular element shows little or no complexation.

In our previous assessments, we only considered the complexing effect of EDTA. This was taken to be representative of aminocarboxylic acids, which were the complexants we expected to be of most significance at the LLWR.

We now expect that high pH conditions will be maintained in the grout pore water for several thousand years [6]. ISA, which can act as a strong complexing agent, may therefore be more significant than we argued in the 2011 ESC and 2017 HRA. This is because the high pH conditions:

- are more conducive to the formation of ISA;
- will inhibit microbial degradation of ISA.

ISA is formed from the degradation of cellulosic wastes. There is some uncertainty, as discussed in reference [6], as to whether cellulosic wastes will be sufficiently isolated from the grout that they evolve at a lower pH than the grout porewater. Within such isolated niches, ISA production would be lower and microbial activity may be possible which would act to reduce ISA concentrations. Given the uncertainties, in our assessment we have cautiously assumed that we need to account for ISA production and persistence under the high pH conditions in the grouted wastes.

We have therefore considered the complexing effects of EDTA and ISA in the vaults in our assessment. On the basis of work reported in reference [96] we have concluded that it is

appropriate to treat contaminants as complexing only with EDTA or ISA. For each contaminant, we have identified the dominant complexant by examining the SRF and SEF data. In the vaults, only iron favours complexation with EDTA over complexation with ISA [96].

In the trenches and the geosphere, only complexation with EDTA is relevant. This is because:

- chemical conditions in the trenches will result in lower production of ISA, and microbial activity will degrade any ISA produced;
- in the geosphere, microbial activity will result in degradation of ISA.

When complexed contaminants enter the geosphere, it is expected that some degree of dissociation of the complexes will occur. Two extreme cases can be considered. These are the 'no dissociation' and 'full dissociation' cases. The 'no dissociation' case neglects the effects of competition with other metal ions and dilution, whereas the 'full dissociation' case assumes that such effects render the impact of EDTA negligible in the geosphere.

In our assessment, we adopt the 'no dissociation' case. Thus, in the B2 and B3 lithofacies units, we assume that EDTA present in those units continues to modify contaminant sorption coefficients in those units. This is a cautious treatment. We further assume that the SRFs for the B2 and B3 lithofacies units are the same as those for the trenches.

As the SRFs and SEFs are a function of EDTA concentration, we have had to consider the concentration of EDTA within porewater. We have done this in the following ways.

- For the trenches, we have assumed that the EDTA concentration is fixed at a value of $1 \times 10^{-7} \text{ mol l}^{-1}$, based upon leachate measurements. We have then used SRFs and SEFs appropriate for this concentration [97].
- For the vaults, we have estimated the quantity of EDTA that may be present in each vault [97]. Using these vault loadings, we have calculated the concentrations of EDTA in porewater in the vaults and in the B2 and B3 units using the contaminant transport equations within the assessment model. We have then used SRFs and SEFs appropriate for the calculated concentrations and a pH of 12.5. We expect this pH to persist throughout the assessment time frame [6].

For ISA, we have calculated SRFs and SEFs based upon the expected concentration of ISA in each vault. These calculations are reported in references [96] and [98].

Groundwater flow in the B3 unit

Groundwater path lengths, flow rates, depths and effective flow cross-sections are required to model contaminant transport through the B3 lithofacies unit. We have determined these as part of our hydrogeological modelling carried out in ConnectFlow [30], [25]. Our ConnectFlow models incorporate several developments in our geological and hydrogeological modelling since the 2011 ESC, so provide a significant update to the properties used in the 2011 ESC [99] and subsequent assessments [44].

We have defined a series of planes, shown in Figure 4.7. For each of these planes, we have calculated flows in the B3 unit. The locations and flows calculated are as follows.

- F1: Flow in B3 into and under the trenches;
- F2: Flow in B3 into and under the vaults;
- F3: Flow in B3 at the site boundary (nearest the coast);
- F4: Flow in B3 at the half-way point between the site boundary and the 2080 coast line;
- F5: Flow in B3 at the coastline.

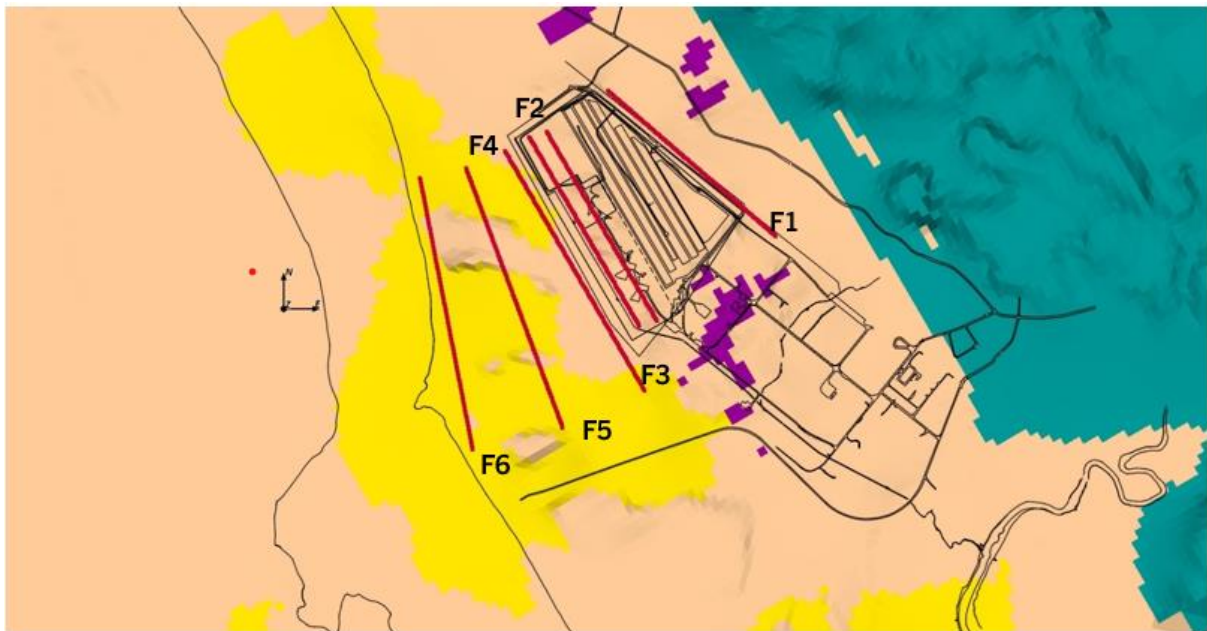


Figure 4.7: Planes used to calculate groundwater flows

We have calculated the flows at these points for the present day. For each climate emissions scenario, we have also built hydrogeological models for the future timesteps given in Table 4.5.

Table 4.5: Snapshot times considered in the hydrogeological modelling

Snapshot times (AD)	Description
2130	As-built (facility at closure)
2230	100 years after as-built (potential end of institutional control)
2430	300 years after as-built (potential end of institutional control)
2780	650 years after as-built (similar to high scenario erosion time)
3280	1150 years after as-built (similar to reference scenario erosion time)
3780	1650 years after as-built (similar to low scenario erosion time)

Each of the future models applies:

- the change in top surface boundary condition (recharge) based on the date and climate scenario from climate studies [100];
- the change in sea level based on the date and climate scenario from climate studies [100];
- the change in topography based on the expected coastal regression and sea level for the date and climate scenario from coastal studies [100];
- updates to the representation of the engineering to represent the as-built engineering (which is the same for all future times but updated from the current-day model);
- the change in hydraulic conductivity of the engineered materials based on the evolution of each engineered material as elicited in the EPA [13].

Using these future models, for each time step we have calculated the flows in the B3 unit at each of the planes shown in Figure 4.7 for each climate emissions scenario. The flows are used as an input to our assessment model implemented in GoldSim. The calculated flows are given in reference [25].

We have characterised uncertainties and variability in the hydrogeological properties [7]. We have used the PDFs derived in that work to inform the groundwater flow rates used in our HYDRO-HI and HYDRO-LO cases.

The pathlengths are derived from the distances from the trenches and vaults to the coast line. Thus, the pathlengths are also time dependent on account of coastal erosion.

Corrosion rates and release timescales

The model makes provision for non-radiological contaminants to be released into solution with trench or vault porewater, as a result of the corrosion of the metallic wasteform in which they are present. In order to calculate the release timescales of such contaminants, we have assumed that metallic wastes are present as sheets that corrode uniformly across the two faces of each sheet. In this case, the timescale for corrosion is specified through a rate constant. Corrosion rate, sheet thicknesses and calculated rate constants are given in reference [25]. The corrosion rates identified are appropriate for the anticipated chemical conditions in the vaults and trenches.

We have considered the physical form in which the contaminant is most likely to be predominantly present in the inventory when selecting corrosion rates. So, for example, for chromium we use the corrosion rate for stainless steel appropriate for the expected chemical conditions. Similarly, for cobalt we use the corrosion rate for stellite and for tin, the corrosion rate of bronze.

For those contaminants for which understanding or data were insufficient to conclude better corrosion performance than stainless steel, the decision was taken to assume instantaneous release of the contaminants into solution.

The characteristic thicknesses of the metals were based on the thicknesses of materials typically expected to be found disposed at the LLWR and constructed from those metals. The bases of the chosen metal thicknesses are given in Table 4.6.

Table 4.6: Basis for chosen thicknesses of metals

Metal	Basis for chosen thickness
Stainless steel	Based on the thickness of Magnox pressure tubes
Mild steel	Cautiously taken to be the same as for stainless steel
Molybdenum	Unknown, taken to be the same as for stainless steel
Nickel and alloys	Taken to be the same as for stainless steel
Lead	Based on typical thickness of shielding
Copper and alloys	Based on thickness of copper wire
Chromium	Taken to be the same as for stainless steel
Cobalt	Taken to be the same as for stellite
Tin	Taken to be the same as for copper

Assuming the characteristic thickness of mild steel to be the same as that for stainless steel is likely to be a cautious assumption.

Physical properties

The assessment model requires the porosity and bulk density of soil in the trenches, grout in the vaults, and lithofacies units B2 and B3. These properties are set out in reference [25] and have been informed by our hydrogeological studies.

The Compartment Flow Model requires hydraulic properties for:

- the engineered barriers and the profile fill, which are identified in the EPA [13];
- the B2 lithofacies unit, which are informed by our hydrogeological studies [7].

The data used in the Compartment Flow Model are reported in reference [25].

Compartment properties

The basis of the assessment model for the LLWR near field is a series of interconnected well-mixed compartments. The compartments represent different spatial regions of the trenches, vaults and surrounding geology, and track the concentrations of the disposed contaminants as they migrate through the near field. The properties of these compartments are presented in Appendix C of reference [25].

Dual porosity model properties

The dual porosity model explicitly considers waste stacks and diffusive transport of contaminants. Based upon assumed stack heights presented in reference [5, 10] and our Stage 2 inventory, we have determined the number of waste stacks in each vault. We have considered stacks of LLW in HHISOs and stacks of ILW, assumed to be disposed in SWTC-compatible strong boxes. The dimensions of these containers define the dimensions of the stacks. Data on stack numbers and container dimensions are presented in reference [25].

In our calculations, we have taken all contaminants to have the same diffusion coefficient ($1.30 \cdot 10^{-10} \text{ m}^2 \text{ s}^{-1}$) [25]. We have ignored the effects of anion exclusion when calculating diffusive fluxes.

4.6.6 Model Quality Assurance

The first requirement is that the assessment model includes the relevant FEPs, and incorporates a correct understanding of the important interactions at an appropriate scale. This is a matter of sufficient information, scientific understanding and judgement.

Development of assessment models may be iterative and generally includes stages of discussion and review. Scoping calculations and results from model testing may also reveal where attention should be focused to improve a model and focus it to the assessment in hand.

The assessment model we have used for the HRA is a development of the model used in the 2017 HRA, which in itself was a development of the assessment model used in the 2011 ESC. These existing models have been through development and testing stages. Some of

the updates made post-2011, and for the 2026 ESC, have involved additional testing and development. For example, this includes improved representation of radionuclide transport through the geology underlying the repository in the groundwater pathway assessment model [24].

The 2026 ESC has included two phases of assessment model development. A comprehensive audit of biases in the assessment models was undertaken at the end of Phase 1. Improvements to reduce biases were implemented during Phase 2, contributing to robust assessment models.

Where possible, we use simpler supporting calculations to reproduce key assessment model results. This builds confidence in the assessment models. The simpler supporting calculations are not expected to exactly reproduce the results from the more complex assessment models. However, the results need to be similar, or the reasons for differences readily explainable. We discuss this further in Subsection 4.6.7.

Calculation of a range of intermediate outputs and assessment of a range of calculation cases enables the behaviour of models to be 'sense checked'. This includes checking individual model calculations and various outputs for a given case, and cross-comparison of the results for different cases to ensure changes in the results are as expected, or explainable and logical.

Software quality and implementation

When judging the suitability of software for use in a safety assessment, it must be demonstrated that the software is suitable for its purpose, has been used properly, the development process has followed appropriate procedures, and that the software produces accurate results.

Key assessment programs used in, or supporting, the 2026 HRA are:

- GoldSim, which has been used to model contaminant transport through the repository near field, geosphere and, for the radiological assessment, radionuclide transfer through the biosphere;
- ConnectFlow, which has been used to model groundwater flows;
- PFLOTRAN, which has been used to model near-field biogeochemistry.

In using the software, each assessment contractor has verified the suitability of the software and checked the implemented models for accuracy using appropriate procedures. Each assessment contractor has been required to produce a quality plan outlining their approach to ensuring quality, including checking of models. We have reviewed and agreed quality plans. As part of assurance activities, we have audited contractors to verify that they have followed their quality plans.

GoldSim

Over the last 30 years, GoldSim has been developed and used by, or for, a diverse set of customers and clients, including government agencies in over ten countries (such as the US

Department of Energy, NASA, the Nuclear Regulatory Commission), research laboratories (including Sandia National Laboratories, Los Alamos National Laboratory, the Paul Scherrer Institute, and Massachusetts Institute of Technology), and commercial organisations worldwide.

Released versions of GoldSim are verified by the software developers. During verification, many test problems are executed and the results compared to expected values. Extensive documentation and user guides are available for the software [101].

When developing our GoldSim assessment model, we have followed a rigorous process of specification of the assessment model, implementation and testing. GoldSim implementations are generally developed in stepwise fashion with testing of component elements, arithmetic and sense checks to verify that the model is behaving as expected. The final model implementation is subjected to comprehensive verification checks to systematically verify the implementation of the specified mathematical model.

ConnectFlow

ConnectFlow is a 3D finite element groundwater flow and transport program [89] which we consider to be an appropriate choice of software for the following reasons:

- It accurately handles flow through porous matrices—e.g., aquifers, soils, clays—via transient and steady-state modelling.
- It enables fine-scale representations of the engineered features within regional-scale models, ensuring consistency across scales.
- It uses the finite-element method which is ideal for complex geological geometries

ConnectFlow has been developed by Amentum (and predecessor organisations) over the last 30 years. It is developed under a rigorous quality system that conforms to the international standards ISO 9001 and TickIT.

The ConnectFlow model for the 2026 ESC and its implementation are described in reference [30].

PFLOTRAN

The 2011 ESC, and earlier assessments, used the Generalised Repository Model (GRM) for reactive-transport modelling, coupling biogeochemical processes with advective groundwater transport [102]. GRM was originally developed by British Nuclear Fuels Ltd for studies of the biogeochemical evolution of the LLWR trenches, and subsequently further developed to consider also the vaults and be applicable to other repositories.

While GRM provided a good representation of the reasonably well-mixed conditions in the trenches, it provided a less satisfactory representation of conditions in the vaults. Conditions in the vaults are expected to be more heterogeneous, due to the isolation provided by the containers and depending on the container contents.

For the 2026 ESC we have moved to using PFLOTRAN. The key advantage of using PFLOTRAN is that provides a more detailed representation of flow and transport than GRM. It is designed for parallel computing, so more detailed, complex models can be solved quickly.

PFLOTRAN is an open source, state-of-the-art massively parallel subsurface flow and reactive transport code. PFLOTRAN solves a system of generally nonlinear partial differential equations describing multiphase, multicomponent and multiscale reactive flow and transport in porous materials. The program is designed to run on massively parallel computing architectures as well as workstations and laptops. Many of the key biogeochemical processes represented in GRM have been carried over to the PFLOTRAN model.

4.6.7 Model Validation

The 3D groundwater flow model underpinning the assessment modelling reported in this section was initially developed for the 2011 ESC [103, 104] using the ConnectFlow software [89]. An in-depth discussion of the evidence supporting the underlying hydrogeological conceptual model for the 2011 ESC is presented in Section 4 of reference [104] and the flow model itself was calibrated against observations (see Section 6 of reference [104]) providing confidence in both.

Since the 2011 ESC, the 3D groundwater flow modelling and the underpinning conceptual model have evolved to match improved understanding of the regional and site-scale hydrogeology and to take advantage of new geological interpretations and modelling tools. Updates to the geological and hydrogeological modelling since 2011 are described in reference [7, 30].

The detailed 3D groundwater flow model is calibrated against observed groundwater heads and other observations. Achieving an acceptable match is used to build confidence in our hydrogeological understanding and provides confirmation that the model contains a good hydrological description of the upper groundwater system.

The extent of tritium migrating from the trenches provides evidence to support the hydrogeological conceptual model, although we note [105] that this evidence should be viewed qualitatively rather than as a quantitative calibration tool. Nevertheless, measured tritium concentrations in groundwater do provide evidence to support the groundwater pathway description given in Subsection 2.2.4. Furthermore, data provide evidence on the timescales for transport through the B2 and B3 lithofacies units. The role of tritium in this regard is discussed further in Subsection 5.4.2.

While the evidence presented above supports our view that the 3D flow modelling is well calibrated and validated against observed data, it must be remembered that the performance assessment model introduced in Subsection 4.6.1 does not use the 3D modelling results directly. Rather, it uses flows calculated in the Compartment Flow Model. The Compartment Flow Model is calibrated by checking the repository water levels in the Compartment Flow

Model against those in the ConnectFlow model and by comparing the B2 flow rates to ensure consistency between the models [30].

We do not consider that the performance assessment model can be validated. We do not use it as a predictive tool, rather as a means to place bounds on potential impacts, subject to appropriately cautious assumptions, and to enable investigation of the sensitivity of outputs to changes in the key parameters. As far as possible, outputs of the performance assessment model are checked to verify that they are broadly consistent with expectations. As an example, we consider whether strongly sorbing contaminants take longer to travel through the geosphere than those that sorb more weakly. Results are also checked for consistency (where expected) with previous assessments. An additional level of confidence in the outputs of the assessment model is provided by checking results against a much simpler 'scoping' model.

The scoping model and comparisons against full assessment outputs from the performance assessment model are presented in detail in reference [24]. The scoping model includes the key physical and chemical processes governing contaminant release and subsequent migration from a single trench or vault, while neglecting (or simplifying) other processes considered to be less important. Full model results for tritium, C-14, Cl-36, Sr-90, Tc-99 and I-129 are compared against results from the scoping model in reference [24]. The agreement is generally good, providing confidence in, and aiding understanding, of the performance assessment model results.

4.7 Calculated Releases of Non-radiological Contaminants to Groundwater

We present detailed results for our reference case in Subsection 4.7.1, which provides the basis of our numerical demonstration that we protect groundwater. We also provide detailed results for:

- the dual porosity model, which allows us to explore the effects of alternative assumptions about water flows and contaminant transport in the near field on assessed performance (see Subsection 4.7.2);
- a case in which the cap performs significantly worse than anticipated, which allows us to explore the sensitivity of assessed impacts to the performance of the engineered cap, which is the key control limiting releases from the system (see Subsection 4.7.3);
- a case in which we only assess non-radiological impacts arising from contaminants present in the grout used to fill voids within waste containers (see Subsection 4.7.4).

We have also explored other key uncertainties and assumptions in the assessment (see Table 4.1. The results of these cases are summarised in Subsection 4.7.5 and presented in detail in reference [25]. In Subsection 4.7.6 we discuss what the assessment case results imply for system performance.

In all cases presented in detail, concentrations of non-radiological hazardous substances are calculated at compliance points directly below each trench and vault within the B3 lithofacies units, accounting only for immediate dilution on entering the regional groundwater in B3. We present peak concentrations calculated at these compliance points. The year of peak concentration is also presented. The appropriate metric against which to compare these peak concentrations is the LLWRAS for the relevant substance.

The non-hazardous pollutants are assessed using their concentration at the site boundary. Results quoted in this subsection are the peak concentrations calculated at these compliance points. The appropriate metric against which to compare these peak concentrations is the LLWRAS for the relevant substance. The year of peak concentration is also presented.

During review of reference [24], we identified that the solubility limits for uranium in the trenches recommended in reference [92] for use in the non-radiological assessment significantly overestimate the solubility of uranium in the trench environment [106]. Therefore, calculations undertaken using those solubility limits overestimate the solubility of uranium and hence overestimate the concentration of uranium in groundwater.

In response, we have determined a revised uranium solubility parameter [106] and the reference case in the groundwater assessment has been re-ran. This results in a reduction in calculated concentrations of uranium of approximately four orders of magnitude. The revised results are presented in Subsection 4.7.1.

Even with an inappropriately high uranium solubility limit, the only exceedance during the PoA occurs in the case examining low sorption in the geosphere, GEO_LOW_SORPTION, where an exceedance of 380 times the LLWRAS is obtained (see Subsection 4.7.5). In the post-PoA exceedances of the LLWRAS for uranium are only found in the following cases:

- the poor cap performance case, where an exceedance of seven times the LLWRAS is obtained (see Subsection 4.7.3);
- the delayed coastal erosion case, DELAY-CE, where an exceedance of around eight times the LLWRAS is obtained (see Subsection 4.7.5).

With a revised uranium solubility, we expect that these exceedances would not occur. Given that all other cases result in uranium concentrations below the LLWRAS, we have considered it disproportionate to re-run all assessment cases with the revised trench solubility limit for uranium.

4.7.1 Reference Case

The reference case is based upon the reference emissions scenario, in which the northern edge of the repository is disrupted by coastal erosion 1250 years after present. The assessment runs from 1950 AD to 3275 AD. The uniform flow representation of the near field is used.

Hazardous substances

Calculated peak concentrations during the PoA, along with the year in which the peak calculated concentration occurs, are presented in Table 4.7. Both benzene and vinyl chloride have peak concentrations below the trenches and vaults at early times after disposal (arising from the trenches before the vault waste is disposed). This is due to the high flows out of the trenches during the PoA and the short half-lives of both contaminants.²⁵

In our assessment calculations we have adopted a half-life of 36 days for benzene [25]. Work undertaken in support of the Geological Disposal Facility (GDF) non-radiological contaminant Integrated Project Team (IPT) [107] suggests that the half-life of benzene under anaerobic conditions ranges from 0.27 years (around 98 days) to 1.97 years, with a best estimate of 1.5 years.

We have investigated the sensitivity of assessed impacts to the half-life of benzene. Using the best estimate value of 1.5 years, we obtain calculated concentrations that are between two to three orders of magnitude greater than the LLWRS from around 1961 to the present day.

The peak calculated concentration for vinyl chloride ($5.3 \times 10^{-4} \text{ mg l}^{-1}$) exceeds its LLWRAS ($5.0 \times 10^{-4} \text{ mg l}^{-1}$), but this is a historic exceedance (occurring below Trench 7 in 2005 AD). No exceedances are calculated at other locations. The concentrations of arsenic, BAP, chromium(VI), lead, mercury and TBP are all many orders of magnitude below their respective LLWRASs. For all these contaminants apart from lead, the largest concentrations are below the vaults and occur towards the end of the PoA.

²⁵ The half-lives are chemical half-lives which account for chemical degradation processes in the near field. They are not radioactive half-lives.

Table 4.7: Calculated peak concentrations for hazardous substances during the PoA for the reference case

Pollutant	LLWRAS (mg l ⁻¹)	Trenches			Vaults		
		Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)	Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Arsenic	1.00 10 ⁻³	9.40 10 ⁻²⁹	2240	9.40 10 ⁻²⁶	6.70 10 ⁻²⁰	2240	6.70 10 ⁻¹⁷
BAP	1.00 10 ⁻⁵	1.30 10 ⁻³⁰	2240	1.30 10 ⁻²⁵	3.40 10 ⁻²¹	2240	3.40 10 ⁻¹⁶
Benzene	1.00 10 ⁻³	1.30 10 ⁻⁵	1977	1.30 10 ⁻²	3.70 10 ⁻¹⁰	1977	3.70 10 ⁻⁷
Chromium(VI)	3.40 10 ⁻³	2.70 10 ⁻³⁷	2240	7.90 10 ⁻³⁵	7.40 10 ⁻²⁸	2240	2.20 10 ⁻²⁵
Lead	1.00 10 ⁻³	2.60 10 ⁻¹³	2150	2.60 10 ⁻¹⁰	5.90 10 ⁻¹⁷	2097	5.90 10 ⁻¹⁴
Mercury	1.00 10 ⁻⁵	8.70 10 ⁻³²	2240	8.70 10 ⁻²⁷	4.60 10 ⁻²²	2240	4.60 10 ⁻¹⁷
TBP	1.00 10 ⁻²	8.30 10 ⁻⁹	2240	8.30 10 ⁻⁷	3.20 10 ⁻⁸	2240	3.20 10 ⁻⁶
Vinyl chloride	5.00 10 ⁻⁴	5.30 10 ⁻⁴	2005	1.10	5.80 10 ⁻⁶	1998	1.20 10 ⁻²

Calculated peak concentrations after the PoA, along with the year in which the peak calculated concentration occurs, are presented in Table 4.8. The concentrations of all hazardous substances remain far below their relevant LLWRASs, even as the concentration increases due to increasing infiltration (for those that do not decay). Our variant calculation indicates that, even if we adopt a longer half-life, benzene concentrations remain less than the relevant LLWRAS after the PoA.

The high performance of the cap ensures that there is little water available for release to the geosphere. The evolution of the calculated concentrations over the assessment time frame at the Vault 8 compliance point is shown in Figure 4.8.

Table 4.8: Calculated peak concentrations for hazardous substances after the PoA for the reference case

Pollutant	LLWRAS (mg l ⁻¹)	Trenches			Vaults		
		Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)	Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Arsenic	1.00 10 ⁻³	3.50 10 ⁻²¹	3275	3.50 10 ⁻¹⁸	7.70 10 ⁻¹⁶	3275	7.70 10 ⁻¹³
BAP	1.00 10 ⁻⁵	1.30 10 ⁻²³	3275	1.30 10 ⁻¹⁸	3.30 10 ⁻¹⁷	3275	3.30 10 ⁻¹²
Benzene	1.00 10 ⁻³	0.00	-	0.00	1.30 10 ⁻³⁶	3275	1.30 10 ⁻³³
Chromium(VI)	3.40 10 ⁻³	2.30 10 ⁻³⁰	3275	6.70 10 ⁻²⁸	4.40 10 ⁻²³	3275	1.30 10 ⁻²⁰
Lead	1.00 10 ⁻³	5.90 10 ⁻¹²	3275	5.90 10 ⁻⁹	4.90 10 ⁻¹⁴	3275	4.90 10 ⁻¹¹
Mercury	1.00 10 ⁻⁵	1.80 10 ⁻²⁴	3275	1.80 10 ⁻¹⁹	3.10 10 ⁻¹⁸	3275	3.10 10 ⁻¹³
TBP	1.00 10 ⁻²	3.60 10 ⁻⁷	3275	3.60 10 ⁻⁵	5.00 10 ⁻⁷	3275	5.00 10 ⁻⁵
Vinyl chloride	5.00 10 ⁻⁴	2.90 10 ⁻¹⁶	2250	5.90 10 ⁻¹³	9.30 10 ⁻¹⁴	2250	1.90 10 ⁻¹⁰

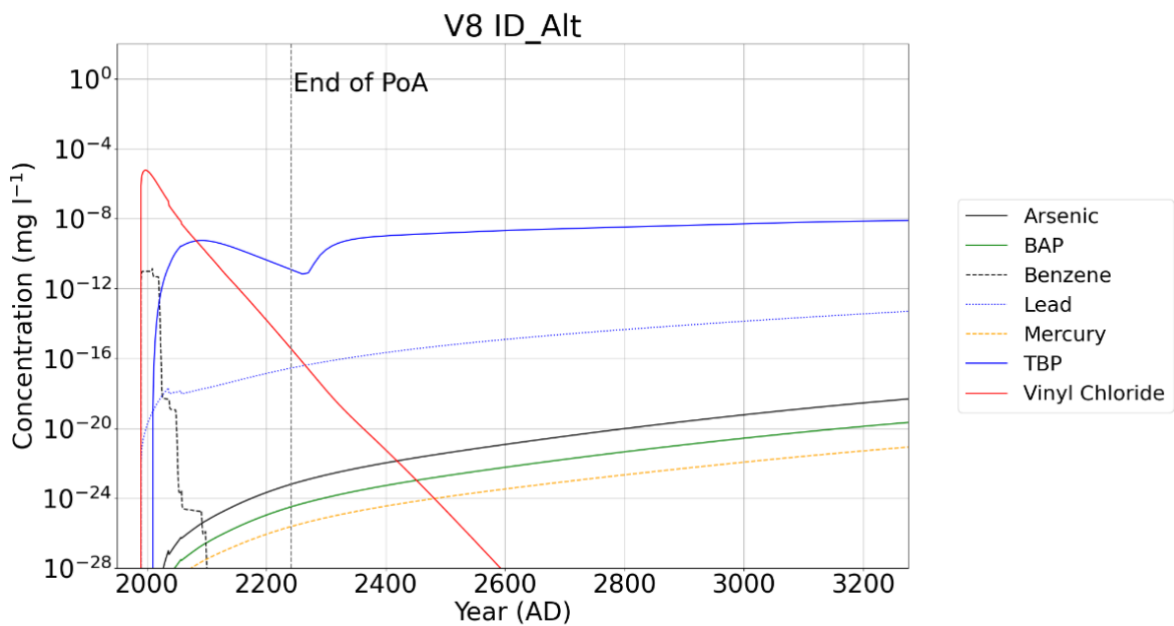


Figure 4.8: Time evolution of calculated concentrations of hazardous substances at the Vault 8 compliance point for the reference case

Non-hazardous pollutants

Calculated peak concentrations during the PoA, along with the year in which the peak calculated concentration occurs, are presented in Table 4.9. All peak concentrations are several orders of magnitude below the relevant LLWRAS.

For most of the non-hazardous pollutants assessed, the assessed concentrations at the site boundary increase throughout the entire assessment time frame, as shown in Figure 4.9. There is no significant variation in concentrations of non-hazardous pollutants at different locations along the site boundary. Calculated peak concentrations after the PoA are presented in Table 4.10. All peak concentrations are significantly below the relevant assessment criteria.

In our assessment calculations, phenol has been assigned a half-life of 2.8 days [25], so it is mostly degraded before reaching the site boundary. There is significant uncertainty in the half-life of phenol, and the value we have chosen is most appropriate for degradation in aerobic conditions, where microbial populations would be present.

We have explored the sensitivity of calculated phenol concentrations to degradation half-life. Work undertaken in support of the GDF non-radiological contaminant IPT suggests that, for anaerobic, neutral to mildly acidic conditions, the half-life of phenol would range from 0.02 years (around 7 days) to 0.82 years, with a best estimate of 0.4 years [107]. These values are appropriate for the trenches and we consider that they may be appropriate for the vaults, given that we anticipate that microbial populations will be able to exist in the lower pH conditions which we expect will be established in the gaps between container stacks.

Using the best estimate and upper bound half-life values, our calculations show the following.

- Using the best estimate of 0.4 years, all assessed phenol concentrations at the site boundary are lower than the assessment criterion for the entirety of the assessment time frame. In the PoA, the peak concentrations at the site boundary are nearly two orders of magnitude lower than the assessment criterion, and after the PoA the concentrations are negligible.
- Using the upper limit of 0.8 years, we obtain slight historic exceedances in the PoA (less than a factor of 2, and around 10 years duration between 1982 and 1992), but concentrations remain negligible after the PoA.

We consider therefore that our conclusions with respect to groundwater protection are robust with respect to uncertainty in the half-life of phenol.

Table 4.9: Calculated peak concentrations of non-hazardous pollutants during the PoA for the reference case

Pollutant	LLWRAS (mg l⁻¹)	Peak Concentration (mg l⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Aluminium	2.00 10 ⁻¹	1.40 10 ⁻²⁶	2240	7.10 10 ⁻²⁶
Ammonium	1.00	2.80 10 ⁻¹⁹	2240	2.80 10 ⁻¹⁹
Barium	7.00 10 ⁻¹	3.70 10 ⁻²²	2240	5.30 10 ⁻²²
Beryllium	4.00 10 ⁻³	6.30 10 ⁻²⁰	2240	1.60 10 ⁻¹⁷
Boron	2.00	3.00 10 ⁻²	2098	1.50 10 ⁻²
Cadmium	5.00 10 ⁻³	1.90 10 ⁻¹⁷	2240	3.80 10 ⁻¹⁵
Chromium	4.70 10 ⁻³	1.50 10 ⁻⁸	2240	3.30 10 ⁻⁶
Cobalt	3.00 10 ⁻³	1.90 10 ⁻¹¹	2240	6.50 10 ⁻⁹
Copper	1.00 10 ⁻³	3.60 10 ⁻¹³	2240	3.60 10 ⁻¹⁰
Fluoride	1.00	3.20 10 ⁻⁶	2058	3.20 10 ⁻⁶
Iron	1.00	9.40 10 ⁻¹⁷	2240	9.40 10 ⁻¹⁷
Magnesium	None	1.80 10 ⁻³	2170	-
Molybdenum	7.00 10 ⁻²	5.90 10 ⁻¹⁹	2240	8.50 10 ⁻¹⁸

Pollutant	LLWRAS (mg l ⁻¹)	Peak Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Nickel	2.00 10 ⁻²	4.00 10 ⁻⁸	2240	2.00 10 ⁻⁶
Phenol	7.70 10 ⁻³	1.50 10 ⁻¹⁸	1976	2.00 10 ⁻¹⁶
Phosphate	1.00 10 ⁻¹	8.60 10 ⁻¹²	2240	8.60 10 ⁻¹¹
Selenium	1.00 10 ⁻²	3.10 10 ⁻¹²	2240	3.10 10 ⁻¹⁰
Tin	2.50 10 ⁻²	4.10 10 ⁻³²	2240	1.60 10 ⁻³⁰
Titanium	-	5.80 10 ⁻³⁷	2240	-
Uranium	1.50 10 ⁻²	3.10 10 ⁻⁹	2240	2.1 10 ⁻⁷
Vanadium	2.00 10 ⁻²	6.70 10 ⁻²³	2240	3.30 10 ⁻²¹
Zinc	8.00 10 ⁻³	3.60 10 ⁻¹⁰	2240	4.40 10 ⁻⁸

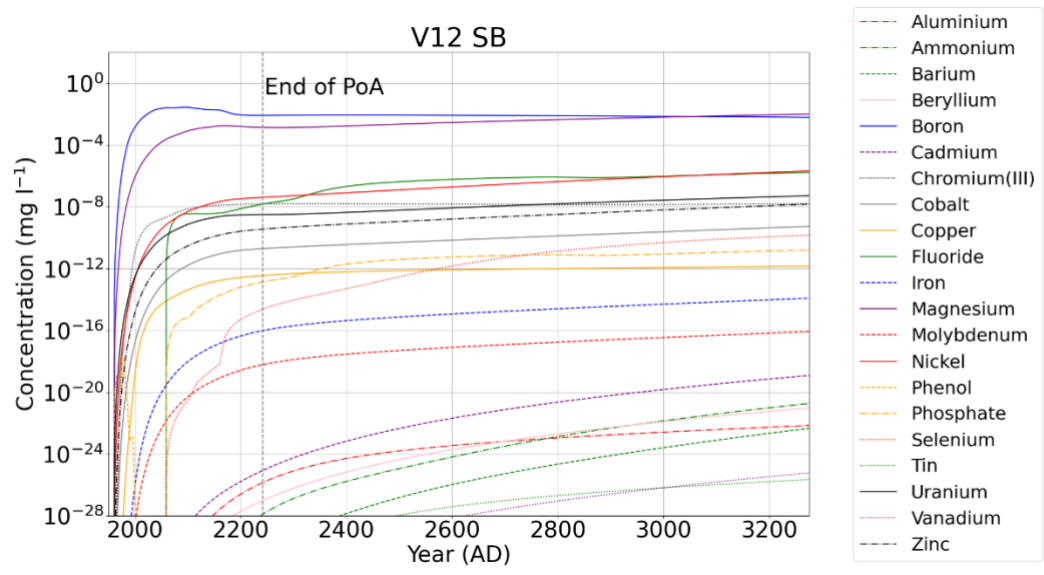


Figure 4.9: Calculated concentrations of non-hazardous pollutants at the site boundary close to Vault 12 for the reference case

Table 4.10: Calculated peak concentrations of non-hazardous pollutants after the PoA for the reference case

Pollutant	LLWRAS (mg l⁻¹)	Peak Concentration (mg l⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Aluminium	2.00 10 ⁻¹	6.40 10 ⁻²²	3275	3.20 10 ⁻²¹
Ammonium	1.00	2.00 10 ⁻¹⁴	3275	2.00 10 ⁻¹⁴
Barium	7.00 10 ⁻¹	7.10 10 ⁻¹⁷	3275	1.00 10 ⁻¹⁶
Beryllium	4.00 10 ⁻³	2.30 10 ⁻¹⁶	3275	5.70 10 ⁻¹⁴
Boron	2.00	9.20 10 ⁻³	2420	4.60 10 ⁻³
Cadmium	5.00 10 ⁻³	9.00 10 ⁻¹⁴	3275	1.80 10 ⁻¹¹
Chromium	4.70 10 ⁻³	1.70 10 ⁻⁸	2350	3.50 10 ⁻⁶
Cobalt	3.00 10 ⁻³	5.40 10 ⁻¹⁰	3275	1.80 10 ⁻⁷
Copper	1.00 10 ⁻³	1.40 10 ⁻¹²	3275	1.40 10 ⁻⁹
Fluoride	1.00	1.80 10 ⁻⁵	3275	1.80 10 ⁻⁵
Iron	1.00	1.20 10 ⁻¹⁴	3275	1.20 10 ⁻¹⁴
Magnesium	None	1.00 10 ⁻²	3275	-
Molybdenum	7.00 10 ⁻²	8.70 10 ⁻¹⁷	3275	1.20 10 ⁻¹⁵

Pollutant	LLWRAS (mg l ⁻¹)	Peak Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Nickel	2.00 10 ⁻²	2.10 10 ⁻⁶	3275	1.00 10 ⁻⁴
Phenol	7.70 10 ⁻³	1.70 10 ⁻²⁹	2940	2.20 10 ⁻²⁷
Phosphate	1.00 10 ⁻¹	1.70 10 ⁻¹⁰	3275	1.70 10 ⁻⁹
Selenium	1.00 10 ⁻²	2.00 10 ⁻⁸	3275	2.00 10 ⁻⁶
Tin	2.50 10 ⁻²	2.20 10 ⁻²⁶	3275	8.90 10 ⁻²⁵
Titanium	-	9.50 10 ⁻³⁰	3275	-
Uranium	1.50 10 ⁻²	5.40 10 ⁻⁸	3275	3.6 10 ⁻⁶
Vanadium	2.00 10 ⁻²	5.30 10 ⁻¹⁹	3275	2.60 10 ⁻¹⁷
Zinc	8.00 10 ⁻³	1.50 10 ⁻⁸	3275	1.90 10 ⁻⁶

4.7.2 Dual Porosity Model

The reference case is based upon the reference emissions scenario, in which the northern edge of the repository is disrupted by coastal erosion 1250 years after present. The assessment runs from 1950 AD to 3275 AD. The dual porosity representation of the near field is used. This case allows us to understand the effect of near-field representation on assessed impacts. This case uses the same infiltration rates and barrier properties as the reference case.

Hazardous substances

Calculated peak concentrations during the PoA, along with the year in which the peak calculated concentration occurs, are presented in Table 4.11. During the PoA, calculated peak concentrations of some contaminants (notably TBP) increase relative to the peaks calculated in the reference case. These increases occur because of increased releases from the vaults.²⁶

Calculated peak concentrations after the PoA, along with the year in which the peak calculated concentration occurs, are presented in Table 4.12. The results show increased peak concentrations of some contaminants relative to the reference case.

The increased concentrations relative to the reference case occur because the dual porosity model does not take into account diffusive coupling of contaminant concentrations between the wastes and the gap between the stacks. This is a cautious treatment because, if water flows rates through the gap are low, the contaminant concentrations in the gap can exceed the corresponding concentrations in the wastes. When this occurs, the dual porosity model can give rise to greater contaminant fluxes to the underlying B2 unit relative to the uniform flow model.

Even with this cautious treatment of contaminant release from the wastes, the only exceedance arises because of vinyl chloride. The exceedance, and the time of exceedance, are the same as for the reference case. All other calculated concentrations remain order of magnitude lower than the relevant LLWRASs.

²⁶ Lateral flows mean that increased releases from the vaults can result in increased concentrations under the trenches.

Table 4.11: Calculated peak concentrations of hazardous substances during the PoA for the dual porosity model

Pollutant	LLWRAS (mg l ⁻¹)	Trenches			Vaults		
		Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)	Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Arsenic	1.00 10 ⁻³	7.90 10 ⁻²⁵	2240	7.90 10 ⁻²²	2.50 10 ⁻¹⁷	2240	2.50 10 ⁻¹⁴
BAP	1.00 10 ⁻⁵	4.30 10 ⁻²⁶	2240	4.30 10 ⁻²¹	1.60 10 ⁻¹⁸	2240	1.60 10 ⁻¹³
Benzene	1.00 10 ⁻³	1.30 10 ⁻⁵	1977	1.30 10 ⁻²	3.70 10 ⁻¹⁰	1977	3.70 10 ⁻⁷
Chromium(VI)	3.40 10 ⁻³	1.60 10 ⁻³²	2240	4.70 10 ⁻³⁰	5.70 10 ⁻²³	2240	1.70 10 ⁻²⁰
Lead	1.00 10 ⁻³	2.60 10 ⁻¹³	2150	2.60 10 ⁻¹⁰	2.50 10 ⁻¹⁴	2240	2.50 10 ⁻¹¹
Mercury	1.00 10 ⁻⁵	1.50 10 ⁻²⁷	2240	1.50 10 ⁻²²	5.80 10 ⁻²⁰	2240	5.80 10 ⁻¹⁵
TBP	1.00 10 ⁻²	6.70 10 ⁻⁴	2240	6.70 10 ⁻²	2.20 10 ⁻³	2200	2.20 10 ⁻¹
Vinyl chloride	5.00 10 ⁻⁴	5.30 10 ⁻⁴	2005	1.1	3.40 10 ⁻⁵	2090	6.70 10 ⁻²

Table 4.12: Calculated peak concentrations of hazardous substances after the PoA for the dual porosity model

Pollutant	LLWRAS (mg l ⁻¹)	Trenches			Vaults		
		Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)	Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Arsenic	1.00 10 ⁻³	1.60 10 ⁻¹⁷	3275	1.60 10 ⁻¹⁴	2.20 10 ⁻¹²	3275	2.20 10 ⁻⁹
BAP	1.00 10 ⁻⁵	9.30 10 ⁻¹⁹	3275	9.30 10 ⁻¹⁴	1.00 10 ⁻¹²	3275	1.00 10 ⁻⁷
Benzene	1.00 10 ⁻³	0.00	-	0.00	0.00	-	0.00
Chromium(VI)	3.40 10 ⁻³	8.30 10 ⁻²⁶	3275	2.50 10 ⁻²³	1.20 10 ⁻¹⁸	3275	3.60 10 ⁻¹⁶
Lead	1.00 10 ⁻³	5.90 10 ⁻¹²	3275	5.90 10 ⁻⁹	1.00 10 ⁻⁹	3275	1.00 10 ⁻⁶
Mercury	1.00 10 ⁻⁵	5.40 10 ⁻²⁰	3275	5.40 10 ⁻¹⁵	4.50 10 ⁻¹⁴	3275	4.50 10 ⁻⁹
TBP	1.00 10 ⁻²	1.40 10 ⁻³	2440	1.40 10 ⁻¹	2.00 10 ⁻³	2250	2.00 10 ⁻¹
Vinyl chloride	5.00 10 ⁻⁴	6.10 10 ⁻¹²	2250	1.20 10 ⁻⁸	2.70 10 ⁻¹⁰	2250	5.40 10 ⁻⁷

Non-hazardous pollutants

Calculated peak concentrations of non-hazardous pollutants are presented for the PoA and after the PoA in Table 4.13 and Table 4.14, respectively. During the PoA, we calculate increases in concentrations at the compliance point for barium, chromium, copper, nickel, phosphate and selenium relative to the reference case. As with the hazardous substances, the increases occur because of the cautious neglect in the assessment model of diffusive coupling between the grouted wastes and the gap between the container stacks. Despite these increases, all calculated concentrations remain orders of magnitude lower than the assessment criteria during, and after, the PoA.

Table 4.13: Calculated peak concentrations of non-hazardous pollutants during the PoA for the dual porosity model

Pollutant	LLWRAS (mg l⁻¹)	Peak Concentration (mg l⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Aluminium	2.00 10 ⁻¹	1.90 10 ⁻²⁶	2240	9.50 10 ⁻²⁶
Ammonium	1.00	2.10 10 ⁻¹⁸	2240	2.10 10 ⁻¹⁸
Barium	7.00 10 ⁻¹	2.40 10 ⁻¹⁷	2240	3.40 10 ⁻¹⁷
Beryllium	4.00 10 ⁻³	8.80 10 ⁻¹⁸	2240	2.20 10 ⁻¹⁵
Boron	2.00	3.00 10 ⁻²	2098	1.50 10 ⁻²
Cadmium	5.00 10 ⁻³	1.80 10 ⁻¹⁵	2240	3.60 10 ⁻¹³
Chromium	4.70 10 ⁻³	4.20 10 ⁻⁴	2150	9.00 10 ⁻²
Cobalt	3.00 10 ⁻³	1.90 10 ⁻¹¹	2240	6.50 10 ⁻⁹
Copper	1.00 10 ⁻³	3.70 10 ⁻⁶	2160	3.70 10 ⁻³
Fluoride	1.00	2.30 10 ⁻³	2098	2.30 10 ⁻³
Iron	1.00	9.40 10 ⁻¹⁷	2240	9.40 10 ⁻¹⁷
Magnesium	None	1.80 10 ⁻³	2170	-
Molybdenum	7.00 10 ⁻²	5.90 10 ⁻¹⁹	2240	8.50 10 ⁻¹⁸

Pollutant	LLWRAS (mg l ⁻¹)	Peak Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Nickel	2.00 10 ⁻²	3.50 10 ⁻⁴	2240	1.70 10 ⁻²
Phenol	7.70 10 ⁻³	1.50 10 ⁻¹⁸	1976	2.00 10 ⁻¹⁶
Phosphate	1.00 10 ⁻¹	2.10 10 ⁻⁸	2098	2.10 10 ⁻⁷
Selenium	1.00 10 ⁻²	5.50 10 ⁻⁸	2240	5.50 10 ⁻⁶
Tin	2.50 10 ⁻²	4.10 10 ⁻³²	2240	1.60 10 ⁻³⁰
Titanium	-	6.70 10 ⁻³⁷	2240	-
Uranium	1.50 10 ⁻²	5.70 10 ⁻⁵	2210	3.80 10 ⁻³
Vanadium	2.00 10 ⁻²	6.90 10 ⁻²¹	2240	3.40 10 ⁻¹⁹
Zinc	8.00 10 ⁻³	3.60 10 ⁻¹⁰	2240	4.40 10 ⁻⁸

Table 4.14: Calculated peak concentrations of non-hazardous pollutants after the PoA for the dual porosity model

Pollutant	LLWRAS (mg l⁻¹)	Peak Concentration (mg l⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Aluminium	2.00 10 ⁻¹	4.80 10 ⁻¹⁹	3275	2.40 10 ⁻¹⁸
Ammonium	1.00	5.10 10 ⁻¹¹	3275	5.10 10 ⁻¹¹
Barium	7.00 10 ⁻¹	5.40 10 ⁻¹²	3275	7.80 10 ⁻¹²
Beryllium	4.00 10 ⁻³	1.60 10 ⁻¹²	3275	4.00 10 ⁻¹⁰
Boron	2.00	3.00 10 ⁻¹	3275	1.50 10 ⁻¹
Cadmium	5.00 10 ⁻³	1.20 10 ⁻¹⁰	3275	2.30 10 ⁻⁸
Chromium	4.70 10 ⁻³	5.50 10 ⁻⁴	2380	1.20 10 ⁻¹
Cobalt	3.00 10 ⁻³	1.20 10 ⁻⁹	3275	3.90 10 ⁻⁷
Copper	1.00 10 ⁻³	8.00 10 ⁻⁵	3275	8.00 10 ⁻²
Fluoride	1.00	5.60 10 ⁻³	3275	5.60 10 ⁻³
Iron	1.00	1.20 10 ⁻¹⁴	3275	1.20 10 ⁻¹⁴
Magnesium	None	1.00 10 ⁻²	3275	-
Molybdenum	7.00 10 ⁻²	9.30 10 ⁻¹⁵	3275	1.30 10 ⁻¹³

Pollutant	LLWRAS (mg l ⁻¹)	Peak Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Nickel	2.00 10 ⁻²	5.70 10 ⁻³	3275	2.80 10 ⁻¹
Phenol	7.70 10 ⁻³	1.70 10 ⁻²⁹	2940	2.20 10 ⁻²⁷
Phosphate	1.00 10 ⁻¹	5.30 10 ⁻⁸	3275	5.30 10 ⁻⁷
Selenium	1.00 10 ⁻²	1.10 10 ⁻⁴	3275	1.10 10 ⁻²
Tin	2.50 10 ⁻²	3.80 10 ⁻²⁶	3275	1.50 10 ⁻²⁴
Titanium	-	2.60 10 ⁻²⁵	3275	-
Uranium	1.50 10 ⁻²	7.50 10 ⁻⁴	3275	5.00 10 ⁻²
Vanadium	2.00 10 ⁻²	1.80 10 ⁻¹⁴	3275	9.00 10 ⁻¹³
Zinc	8.00 10 ⁻³	1.60 10 ⁻⁸	3275	2.00 10 ⁻⁶

4.7.3 Poor Cap Performance Case

This case explores the effect of poor cap performance on assessed impacts. It considers both:

- early onset of geomembrane failure (the 5th percentile from the PDF describing membrane failure time is chosen)
- high infiltration (the 95th percentile from the PDF describing cap infiltration is chosen).

This case is based upon the reference emissions scenario, in which the northern edge of the repository is disrupted by coastal erosion 1250 years after present. The assessment runs from 1950 AD to 3275 AD. The uniform flow representation of the near field is used.

Hazardous substances

Calculated peak concentrations during the PoA, along with the year in which the peak calculated concentration occurs, are presented in Table 4.15. During the PoA, the calculated peaks are very similar to those calculated for the reference case. The concentrations for the contaminants that peak at the end of the PoA have increased slightly as a result of the increased infiltration rate, but only by up to an order of magnitude. The only exceedance of the relevant LLWRAS is vinyl chloride. This is an historical exceedance.

Calculated peak concentrations after the PoA, along with the year in which the peak calculated concentration occurs, are presented in Table 4.16. In general, the peak concentrations under the vaults are larger than those in the trenches by a couple of orders of magnitude. The peaks occur at similar times in both locations.

With the exception of vinyl chloride and benzene, the calculated peak concentrations increase considerably relative to those calculated for the reference case. This is because the cap performs significantly worse in this case, resulting in significantly more water entering the system through the cap. Notwithstanding this, all calculated peak impacts are less than the relevant LLWRAS. This case demonstrates that the cap is a key control on the release of contaminants from the repository and provides confidence that the system would still provide protection of groundwater even if the cap were to perform substantially worse than we anticipate.

Table 4.15: Calculated peak concentrations of hazardous substances during the PoA for the poor cap performance case

Pollutant	LLWRAS (mg l ⁻¹)	Trenches			Vaults		
		Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)	Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Arsenic	1.00 10 ⁻³	1.60 10 ⁻²⁸	2240	1.60 10 ⁻²⁵	6.90 10 ⁻²⁰	2240	6.90 10 ⁻¹⁷
BAP	1.00 10 ⁻⁵	2.50 10 ⁻³⁰	2240	2.50 10 ⁻²⁵	3.40 10 ⁻²¹	2240	3.40 10 ⁻¹⁶
Benzene	1.00 10 ⁻³	1.30 10 ⁻⁵	1977	1.30 10 ⁻²	3.70 10 ⁻¹⁰	1977	3.70 10 ⁻⁷
Chromium(VI)	3.40 10 ⁻³	7.60 10 ⁻³⁷	2240	2.20 10 ⁻³⁴	2.70 10 ⁻²⁷	2240	7.90 10 ⁻²⁵
Lead	1.00 10 ⁻³	2.60 10 ⁻¹³	2150	2.60 10 ⁻¹⁰	5.90 10 ⁻¹⁷	2097	5.90 10 ⁻¹⁴
Mercury	1.00 10 ⁻⁵	1.30 10 ⁻³¹	2240	1.30 10 ⁻²⁶	4.60 10 ⁻²²	2240	4.60 10 ⁻¹⁷
TBP	1.00 10 ⁻²	4.60 10 ⁻⁸	2240	4.60 10 ⁻⁶	1.90 10 ⁻⁷	2240	1.90 10 ⁻⁵
Vinyl chloride	5.00 10 ⁻⁴	5.30 10 ⁻⁴	2005	1.10	5.80 10 ⁻⁶	1998	1.20 10 ⁻²

Table 4.16: Calculated peak concentrations of hazardous substances after the PoA for the poor cap performance case

Pollutant	LLWRAS (mg l ⁻¹)	Trenches			Vaults		
		Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)	Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Arsenic	1.00 10 ⁻³	1.90 10 ⁻⁷	3275	1.90 10 ⁻⁴	5.60 10 ⁻⁶	3275	5.60 10 ⁻³
BAP	1.00 10 ⁻⁵	4.10 10 ⁻¹⁰	3275	4.10 10 ⁻⁵	3.60 10 ⁻⁸	3275	3.60 10 ⁻³
Benzene	1.00 10 ⁻³	0.00	-	0.00	4.60 10 ⁻³⁶	2700	4.60 10 ⁻³³
Chromium(VI)	3.40 10 ⁻³	8.00 10 ⁻¹⁶	3275	2.40 10 ⁻¹³	2.20 10 ⁻¹²	3275	6.40 10 ⁻¹⁰
Lead	1.00 10 ⁻³	2.70 10 ⁻⁶	3275	2.70 10 ⁻³	6.50 10 ⁻⁴	3275	6.50 10 ⁻¹
Mercury	1.00 10 ⁻⁵	7.00 10 ⁻¹⁰	3275	7.00 10 ⁻⁵	3.40 10 ⁻⁸	3275	3.40 10 ⁻³
TBP	1.00 10 ⁻²	1.80 10 ⁻⁴	2910	1.80 10 ⁻²	2.10 10 ⁻³	2700	2.10 10 ⁻¹
Vinyl chloride	5.00 10 ⁻⁴	7.20 10 ⁻¹⁶	2250	1.40 10 ⁻¹²	9.30 10 ⁻¹⁴	2250	1.90 10 ⁻¹⁰

The time evolution of the calculated concentrations at the Vault 8 compliance point is shown in Figure 4.10. The abrupt increases in calculated concentrations around 2700 AD arise because this is when the geomembrane degrades, resulting in increased water input into the system.

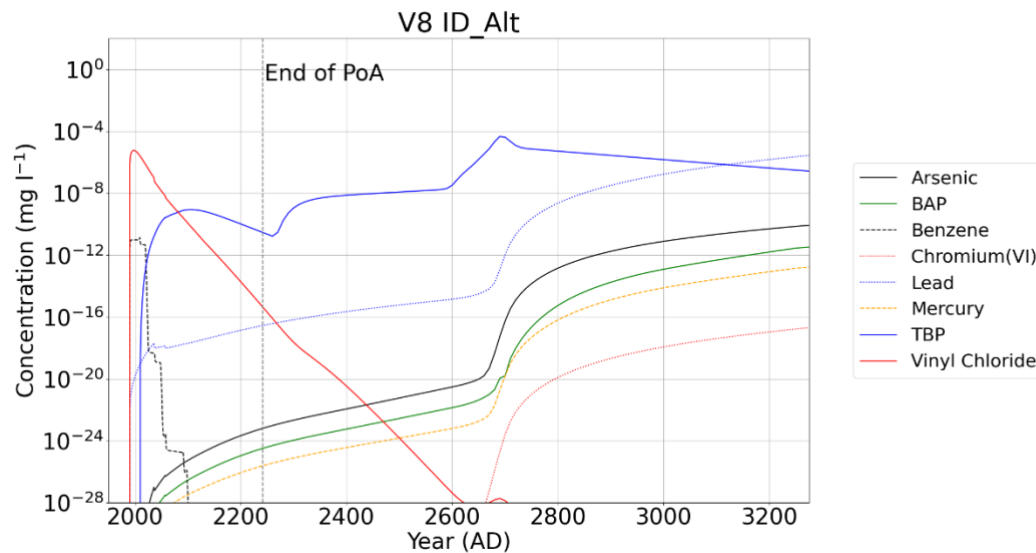


Figure 4.10: Calculated concentrations of hazardous substances at the Vault 8 compliance point for the poor cap performance case

Non-hazardous pollutants

Calculated peak concentrations during the PoA, along with the year in which the peak calculated concentration occurs, are presented in Table 4.17. All calculated peak concentrations are significantly lower than the relevant assessment criteria. The results are very similar to those of the RC, with only barium, fluoride, phosphate and selenium exhibiting slightly enhanced peak concentrations. We note that, because the uranium solubility in the trenches used in this case is different to that used in the reference case, it is inappropriate to compare the calculated uranium concentrations in the two cases.

Calculated peak concentrations after the PoA, along with the year in which the peak calculated concentration occurs, are presented in Table 4.18. As for the hazardous substances, there is a considerable increase in the concentrations of the non-hazardous pollutants relative to the concentrations calculated in the reference case. The peak concentrations of all contaminants except for boron and uranium remain below the relevant assessment criteria.

The peak concentration of boron (3.4 mg l^{-1} at 2760 AD) exceeds its LLWRAS (2.0 mg l^{-1}), occurring at the site boundary next to Vault 12. As shown in Figure 4.11 the duration of the exceedance is limited, with the calculated concentration reducing for the remaining duration of the assessment time frame. As there is no significant and sustained rising trend in the calculated concentration of boron in groundwater, the risk to groundwater quality posed by the boron exceedance is low.

The peak concentration of uranium (0.11 mg l^{-1} at 3275 AD) exceeds its LLWRAS (0.015 mg l^{-1}) by a factor of seven at all locations along the site boundary. The uniformity in the concentration along the site boundary, despite heterogeneity in the mass of uranium across the vaults, is because the disposals in the trenches contribute the most to the concentration at the site boundary. As discussed at the beginning of Subsection 4.7, this case uses an inappropriately large solubility limit for uranium in the trenches. As the trenches contribute most significantly to the assessed impacts, we expect that revising the solubility limit would result in calculated concentrations substantially below the LLWRAS.

Table 4.17: Peak calculated concentrations of non-hazardous pollutants during the PoA for the poor cap performance case

Pollutant	LLWRAS (mg l^{-1})	Peak Concentration (mg l^{-1})	Year of Peak (AD)	Ratio to LLWRAS (-)
Aluminium	2.00×10^{-1}	1.40×10^{-26}	2240	7.10×10^{-26}
Ammonium	1.00	2.80×10^{-19}	2240	2.80×10^{-19}
Barium	7.00×10^{-1}	9.60×10^{-22}	2240	1.40×10^{-21}
Beryllium	4.00×10^{-3}	6.30×10^{-20}	2240	1.60×10^{-17}
Boron	2.00	3.00×10^{-2}	2098	1.50×10^{-2}
Cadmium	5.00×10^{-3}	1.90×10^{-17}	2240	3.90×10^{-15}
Chromium	4.70×10^{-3}	1.50×10^{-8}	2240	3.30×10^{-6}
Cobalt	3.00×10^{-3}	1.90×10^{-11}	2240	6.50×10^{-9}
Copper	1.00×10^{-3}	3.60×10^{-13}	2240	3.60×10^{-10}
Fluoride	1.00	5.40×10^{-6}	2240	5.40×10^{-6}
Iron	1.00	9.40×10^{-17}	2240	9.40×10^{-17}
Magnesium	None	1.80×10^{-3}	2170	-
Molybdenum	7.00×10^{-2}	5.90×10^{-19}	2240	8.50×10^{-18}
Nickel	2.00×10^{-2}	4.00×10^{-8}	2240	2.00×10^{-6}
Phenol	7.70×10^{-3}	1.50×10^{-18}	1976	2.00×10^{-16}

Pollutant	LLWRAS (mg l⁻¹)	Peak Concentration (mg l⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Phosphate	1.00 10 ⁻¹	5.10 10 ⁻¹¹	2240	5.10 10 ⁻¹⁰
Selenium	1.00 10 ⁻²	6.20 10 ⁻¹²	2240	6.20 10 ⁻¹⁰
Tin	2.50 10 ⁻²	4.10 10 ⁻³²	2240	1.60 10 ⁻³⁰
Titanium	-	5.80 10 ⁻³⁷	2240	-
Uranium	1.50 10 ⁻²	5.70 10 ⁻⁵	2210	3.80 10 ⁻³
Vanadium	2.00 10 ⁻²	6.70 10 ⁻²³	2240	3.30 10 ⁻²¹
Zinc	8.00 10 ⁻³	3.60 10 ⁻¹⁰	2240	4.40 10 ⁻⁸

Table 4.18: Peak calculated concentrations of non-hazardous pollutants after the PoA for the poor cap performance case

Pollutant	LLWRAS (mg l ⁻¹)	Peak Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Aluminium	2.00 10 ⁻¹	5.00 10 ⁻¹⁶	3275	2.50 10 ⁻¹⁵
Ammonium	1.00	3.60 10 ⁻⁵	3275	3.60 10 ⁻⁵
Barium	7.00 10 ⁻¹	3.40 10 ⁻⁷	3275	4.80 10 ⁻⁷
Beryllium	4.00 10 ⁻³	2.00 10 ⁻⁸	3275	4.90 10 ⁻⁶
Boron	2.00	3.40	2760	1.70
Cadmium	5.00 10 ⁻³	3.30 10 ⁻⁶	3275	6.60 10 ⁻⁴
Chromium	4.70 10 ⁻³	2.20 10 ⁻³	2720	4.70 10 ⁻¹
Cobalt	3.00 10 ⁻³	8.00 10 ⁻⁶	3275	2.70 10 ⁻³
Copper	1.00 10 ⁻³	2.30 10 ⁻⁴	2730	2.30 10 ⁻¹
Fluoride	1.00	1.20 10 ⁻¹	2880	1.20 10 ⁻¹
Iron	1.00	1.90 10 ⁻¹⁰	3275	1.90 10 ⁻¹⁰
Magnesium	None	3.20 10 ⁻¹	3240	-
Molybdenum	7.00 10 ⁻²	1.50 10 ⁻⁸	3275	2.10 10 ⁻⁷

Pollutant	LLWRAS (mg l ⁻¹)	Peak Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Nickel	2.00 10 ⁻²	1.30 10 ⁻²	3275	6.30 10 ⁻¹
Phenol	7.70 10 ⁻³	1.70 10 ⁻²⁹	2940	2.20 10 ⁻²⁷
Phosphate	1.00 10 ⁻¹	1.10 10 ⁻⁶	2880	1.10 10 ⁻⁵
Selenium	1.00 10 ⁻²	3.20 10 ⁻⁴	2770	3.20 10 ⁻²
Tin	2.50 10 ⁻²	2.00 10 ⁻²⁴	3275	7.90 10 ⁻²³
Titanium	-	1.70 10 ⁻²²	3275	-
Uranium	1.50 10 ⁻²	1.10 10 ⁻¹	3275	7.00
Vanadium	2.00 10 ⁻²	9.20 10 ⁻¹⁰	3275	4.60 10 ⁻⁸
Zinc	8.00 10 ⁻³	2.80 10 ⁻⁵	3275	3.50 10 ⁻³

The time evolution of the concentrations of non-hazardous pollutants at the Vault 12 site boundary is shown in Figure 4.11.

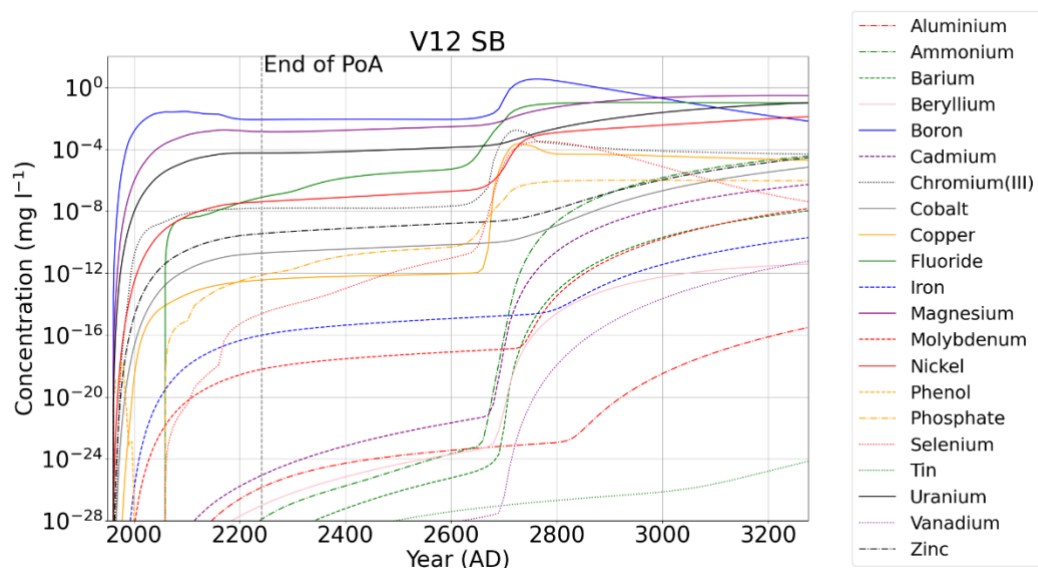


Figure 4.11: Time evolution of calculated concentrations of non-hazardous pollutants at the site boundary close to Vault 12 for the poor cap performance case

4.7.4 Grout Source Term Case

This case is based upon the reference emissions scenario, in which the northern edge of the repository is disrupted by coastal erosion 1250 years after present. The assessment runs from 1950 AD to 3275 AD. The uniform flow model of the near field is used.

Hazardous substances

The only hazardous substances present in the grout source term are arsenic and lead. Peak calculated concentrations during the PoA are presented in Table 4.19. This case considers a vault-only source term. This is because wastes in the trenches are not grouted. The calculated concentrations under the trenches therefore occur because of lateral flows of contaminants released from the vaults to the geosphere.

During the PoA the concentration of arsenic has increased relative to the reference case by five orders of magnitude below the trenches, and by over three orders of magnitude below the vaults. For lead, relative to the reference case, there is an increase in concentration by an order of magnitude below the vaults and a decrease by more than ten orders of magnitude below the trenches. The peak concentrations are several orders of magnitude lower than the relevant LLWRASs.

Calculated peak concentrations after the PoA are given in Table 4.20. Arsenic is calculated to have a peak concentration below the trenches almost three orders of magnitude higher than in the reference case, whereas below the vaults, the peak concentration is increased by

more than two orders of magnitude. For lead, relative to the reference case, there is a decrease in peak concentration below the trenches (of more than six orders of magnitude), and an increase of one order of magnitude in the peak concentration calculated below the vaults.

There are no calculated exceedances during or after the PoA.

Table 4.19: Peak calculated concentrations of hazardous substances during the PoA for the grout source term case

Pollutant	LLWRAS (mg l ⁻¹)	Trenches			Vaults		
		Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)	Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Arsenic	1.00 10 ⁻³	1.50 10 ⁻²³	2240	1.50 10 ⁻²⁰	2.70 10 ⁻¹⁶	2240	2.70 10 ⁻¹³
Lead	1.00 10 ⁻³	9.20 10 ⁻²⁴	2240	9.20 10 ⁻²¹	6.70 10 ⁻¹⁶	2240	6.70 10 ⁻¹³

Table 4.20: Peak calculated concentrations of hazardous substances after the PoA for the grout source term case

Pollutant	LLWRAS (mg l ⁻¹)	Trenches			Vaults		
		Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)	Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Arsenic	1.00 10 ⁻³	2.10 10 ⁻¹⁸	3275	2.10 10 ⁻¹⁵	2.40 10 ⁻¹³	3275	2.40 10 ⁻¹⁰
Lead	1.00 10 ⁻³	1.40 10 ⁻¹⁸	3275	1.40 10 ⁻¹⁵	6.10 10 ⁻¹³	3275	6.10 10 ⁻¹⁰

Non-hazardous pollutants

Calculated peak concentrations during and after the PoA are presented in Table 4.21 and Table 4.22, respectively. All calculated concentrations are several orders of magnitude lower than the relevant assessment criteria.

Table 4.21: Calculated peak concentrations of non-hazardous pollutants during the PoA for the grout source term case

Pollutant	LLWRAS (mg l ⁻¹)	Peak Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Aluminium	2.00 10 ⁻¹	7.10 10 ⁻²⁶	2240	3.50 10 ⁻²⁵
Antimony	5.00 10 ⁻³	2.80 10 ⁻⁹	2240	5.70 10 ⁻⁷
Boron	2.00	9.30 10 ⁻⁸	2240	4.70 10 ⁻⁸
Cadmium	5.00 10 ⁻³	6.50 10 ⁻¹⁶	2240	1.30 10 ⁻¹³
Chromium	4.70 10 ⁻³	2.60 10 ⁻¹⁵	2240	5.50 10 ⁻¹³
Iron	1.00	1.70 10 ⁻²⁴	2240	1.70 10 ⁻²⁴
Molybdenum	7.00 10 ⁻²	1.50 10 ⁻²²	2240	2.20 10 ⁻²¹
Nickel	2.00 10 ⁻²	9.50 10 ⁻¹⁵	2240	4.70 10 ⁻¹³
Selenium	1.00 10 ⁻²	5.80 10 ⁻⁷	2240	5.80 10 ⁻⁵
Vanadium	2.00 10 ⁻²	8.40 10 ⁻²⁰	2240	4.20 10 ⁻¹⁸
Zinc	8.00 10 ⁻³	2.30 10 ⁻¹⁵	2240	2.90 10 ⁻¹³

Table 4.22: Calculated peak concentrations of non-hazardous pollutants after the PoA for the grout source term case

Pollutant	LLWRAS (mg l ⁻¹)	Peak Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Aluminium	2.00 10 ⁻¹	2.10 10 ⁻²⁰	3275	1.10 10 ⁻¹⁹
Antimony	5.00 10 ⁻³	1.20 10 ⁻⁶	3275	2.30 10 ⁻⁴
Boron	2.00	7.20 10 ⁻⁶	3275	3.60 10 ⁻⁶
Cadmium	5.00 10 ⁻³	2.10 10 ⁻¹²	3275	4.30 10 ⁻¹⁰
Chromium	4.70 10 ⁻³	8.50 10 ⁻¹²	3275	1.80 10 ⁻⁹
Iron	1.00	2.00 10 ⁻²⁰	3275	2.00 10 ⁻²⁰
Molybdenum	7.00 10 ⁻²	2.00 10 ⁻¹⁸	3275	2.90 10 ⁻¹⁷
Nickel	2.00 10 ⁻²	2.00 10 ⁻¹¹	3275	1.00 10 ⁻⁹
Selenium	1.00 10 ⁻²	5.90 10 ⁻⁶	3020	5.90 10 ⁻⁴
Vanadium	2.00 10 ⁻²	4.50 10 ⁻¹⁶	3275	2.30 10 ⁻¹⁴
Zinc	8.00 10 ⁻³	7.60 10 ⁻¹²	3275	9.50 10 ⁻¹⁰

4.7.5 Summary of Other Assessment Cases

Results for other assessment cases are presented in reference [25]. Calculated concentrations of vinyl chloride exceed the LLWRAS during the PoA in every assessment case. For other contaminants, exceedances are only found for the following cases.

- DELAY-CE.
- GEO_LOW_SORPTION.

DELAY-CE

This case considers a hypothetical situation in which the LLWR is not disrupted by coastal erosion and remains 250,000 years after present. The case allows us to explore the evolution of assessed impacts over longer timescales. We stress that it does not represent an expected or plausible future evolution of the system.

Calculated peak concentrations of hazardous substances after the PoA are presented in Table 4.23. Exceedances are calculated for BAP (1.4 times greater than the LLWRAS)

and lead (970 times greater than the LLWRAS). These peak concentrations occur after many tens of thousands of years. All other calculated concentrations remain less than the relevant LLWRASs.

Table 4.23: Calculated peak concentrations of hazardous substances after the PoA for the delayed coastal erosion case

Pollutant	LLWRAS (mg l ⁻¹)	Trenches			Vaults		
		Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)	Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Arsenic	1.00 10 ⁻³	9.20 10 ⁻⁷	31950	9.20 10 ⁻⁴	1.30 10 ⁻⁵	22950	1.30 10 ⁻²
BAP	1.00 10 ⁻⁵	1.30 10 ⁻⁶	101950	1.30 10 ⁻¹	1.40 10 ⁻⁵	165950	1.40
Benzene	1.00 10 ⁻³	0.00	-	0.00	6.90 10 ⁻³⁶	9150	6.90 10 ⁻³³
Chromium(VI)	3.40 10 ⁻³	3.60 10 ⁻⁹	251950	1.10 10 ⁻⁶	1.60 10 ⁻⁷	188950	4.80 10 ⁻⁵
Lead	1.00 10 ⁻³	2.10 10 ⁻⁴	62950	2.10 10 ⁻¹	9.70 10 ⁻¹	48950	9.70 10 ²
Mercury	1.00 10 ⁻⁵	6.40 10 ⁻⁸	217950	6.40 10 ⁻³	5.00 10 ⁻⁷	251950	5.00 10 ⁻²
TBP	1.00 10 ⁻²	4.90 10 ⁻⁵	3730	5.00 10 ⁻³	6.10 10 ⁻⁴	3730	6.00 10 ⁻²
Vinyl chloride	5.00 10 ⁻⁴	2.70 10 ⁻¹⁶	2250	5.40 10 ⁻¹³	1.30 10 ⁻¹³	2250	2.60 10 ⁻¹⁰

Calculated peak concentrations of non-hazardous pollutants after the PoA are presented in Table 4.24. Exceedances are calculated for:

- chromium (a factor of 1.1 times greater than the LLWRAS);
- cobalt (almost ten times greater than the LLWRAS);
- copper (almost seven times greater than the LLWRAS);
- nickel (40 times greater than the LLWRAS);
- uranium (around eight times greater than the LLWRAS);
- zinc (17 times greater than the LLWRAS).

All exceedances are calculated to occur at least eight thousand years after the present day. The assessment case uses an inappropriately large solubility limit for uranium in the trenches. As the trenches contribute most significantly to the assessed impacts, we expect that revising the solubility limit would result in calculated concentrations substantially below the LLWRAS.

Table 4.24: Calculated peak concentrations of non-hazardous pollutants after the PoA for the delayed coastal erosion case

Pollutant	LLWRAS (mg l ⁻¹)	Peak Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Aluminium	2.00 10 ⁻¹	1.40 10 ⁻³	251950	6.90 10 ⁻³
Ammonium	1.00	2.10 10 ⁻²	78950	2.10 10 ⁻²
Barium	7.00 10 ⁻¹	9.20 10 ⁻⁵	23950	1.30 10 ⁻⁴
Beryllium	4.00 10 ⁻³	1.80 10 ⁻⁶	51950	4.50 10 ⁻⁴
Boron	2.00	1.80	4650	9.10 10 ⁻¹
Cadmium	5.00 10 ⁻³	1.30 10 ⁻³	27950	2.60 10 ⁻¹
Chromium	4.70 10 ⁻³	5.00 10 ⁻³	175950	1.10
Cobalt	3.00 10 ⁻³	3.00 10 ⁻²	50950	9.90
Copper	1.00 10 ⁻³	6.80 10 ⁻³	251950	6.80
Fluoride	1.00	4.90 10 ⁻²	11950	4.90 10 ⁻²

Pollutant	LLWRAS (mg l ⁻¹)	Peak Concentration (mg l ⁻¹)	Year of Peak (AD)	Ratio to LLWRAS (-)
Iron	1.00	1.50 10 ⁻¹	251950	1.50 10 ⁻¹
Magnesium	None	1.10 10 ⁻¹	6150	-
Molybdenum	7.00 10 ⁻²	4.50 10 ⁻²	210950	6.40 10 ⁻¹
Nickel	2.00 10 ⁻²	7.90 10 ⁻¹	28950	4.00 10 ¹
Phenol	7.70 10 ⁻³	1.70 10 ⁻²⁹	2940	2.20 10 ⁻²⁷
Phosphate	1.00 10 ⁻¹	4.60 10 ⁻⁷	11950	4.60 10 ⁻⁶
Selenium	1.00 10 ⁻²	1.20 10 ⁻⁴	4450	1.20 10 ⁻²
Tin	2.50 10 ⁻²	3.70 10 ⁻⁷	251950	1.50 10 ⁻⁵
Titanium	-	5.30 10 ⁻⁸	251950	-
Uranium	1.50 10 ⁻²	1.10 10 ⁻¹	8250	7.60
Vanadium	2.00 10 ⁻²	1.30 10 ⁻³	86950	6.60 10 ⁻²
Zinc	8.00 10 ⁻³	1.40 10 ⁻¹	51950	1.70 10 ¹

GEO_LOW_SORPTION

In this case sorption within the geosphere is neglected. This is to explore the role of the geosphere in retarding contaminant mobility.

For hazardous substances, the only exceedance calculated is for vinyl chloride, with the peak concentration calculated to be 2.3 times greater than the LLWRAS. For all other hazardous substances except TBP and benzene, calculated peak concentrations of hazardous substances are greater than in the reference case. Concentrations for these two contaminants are unchanged because they do not sorb in the geosphere in the reference case.

For non-hazardous pollutants, exceedances of the relevant LLWRAS are observed for cobalt (12 times greater than the LLWRAS), copper (1.5 times greater than the LLWRAS), nickel (24 times greater than the LLWRAS), uranium (380 times greater than the LLWRAS) and zinc (57 times greater than the LLWRAS) during the PoA. All of the peaks occur in the future except for uranium, where the peak is calculated to have occurred in 1986. This case uses

an inappropriately large solubility limit for uranium in the trenches. As the calculated concentration derives entirely from wastes in the trenches, revising the solubility limit would result in calculated concentrations substantially below the LLWRAS.

Several other non-hazardous pollutants (aluminium, boron, chromium, iron and molybdenum) have peak concentrations within an order of magnitude of their respective LLWRAS.

In the post-PoA period, most peak concentrations are also orders of magnitude higher than in the RC. Only zinc exceeds its LLWRAS (by a factor of 1.3 in 2250 AD), although cobalt, nickel and uranium peak concentrations are within an order of magnitude of their respective LLWRAS.

4.7.6 Discussion of Calculated Non-radiological Impacts

Hazardous substances

The reference case and dual porosity results indicate that, if the cap performs well, little water will enter the repository. Releases to the geosphere will therefore be very low and hence calculated concentrations at the compliance points will be low.

The only exceedances that occur in these cases are for vinyl chloride and, if a longer half-life were to be used, benzene. The vinyl chloride exceedances are calculated to have occurred around 2005, and those for benzene between 1961 and the present day, so derive largely from the trenches and pre-date installation of the final cap. The exceedances do not therefore provide any insight into final cap performance. The releases occur because the interim trench cap performs less well than expected. The performance of the interim trench cap is the same in all assessment cases considered in the assessment. These two cases differ only in their presentation of the vault near field. The calculated vinyl chloride exceedances are therefore the same in each case.

We obtain historical exceedances for vinyl chloride for all other assessment cases, with the exception of the HYDRO-HI case, which considers increased flows in the B3 unit. In this case, the increased flows result in lower calculated concentrations.

Only minor exceedances are calculated for vinyl chloride. The greatest exceedance occurs in the GEO_LOW_SORPTION case, where the peak concentration is a factor of 2.3 greater than the LLWRAS. The peak concentration increases because this case neglects sorption in the geosphere. It is a case designed to explore the significance of the geosphere in controlling contaminant concentrations - it does not represent realistic behaviour.

Vinyl chloride is assumed to arise from the degradation of halogenated plastics and rubbers and it has been detected in leachate. Our assessment is cautious as the vinyl chloride molecules are modelled as being immediately available for dissolution in infiltrating water. Our treatment of benzene is similarly cautious: we assume that a large range of organic materials in the inventory are comprised entirely of benzene which is immediately available for dissolution in infiltrating water.

Despite these cautions, we only calculate vinyl chloride minor exceedances. The duration of the vinyl chloride and benzene exceedances is limited (see Figure 4.8 for vinyl chloride). This is because both contaminants will be subject to biodegradation [25] and also because our optimised site development plan involves the installation of an engineered final cap which will reduce future releases. We argue, therefore, that we are taking all necessary and reasonable measures to avoid the entry of hazardous substances into groundwater and that these measures ensure that there is no significant risk to groundwater in the future. Given these considerations, the risk to groundwater posed by vinyl chloride and benzene in the LLWR wastes is low.

Aside from vinyl chloride and benzene, the only case for which we obtain exceedances for hazardous substances is the delayed coastal erosion case. In this case, we obtain exceedances for BAP (1.4 times greater than the LLWRAS at 165950 AD) and lead (970 times greater than the LLWRAS at 48950 AD). We emphasise that the repository is expected to be disrupted by coastal erosion on a timescale of a few hundred to a few thousand years and it is not plausible that the repository would remain intact over the timescales considered in this assessment case. The environmental significance of the exceedances is therefore low.

In summary, the assessment cases considered indicate that, even if key barriers perform significantly less well than anticipated, there is no significant future risk of hazardous substances entering the environment as a result of disposals at the LLWR.

Non-hazardous pollutants

The reference case and dual porosity assessment cases indicate that, if the final cap performs as expected, assessed concentrations of non-hazardous pollutants at the compliance points will be several orders of magnitude lower than the relevant LLWRAS. This is because the quantities of water entering the repository will be very small.

Exceedances would only occur if we used a half-life of 0.8 years for phenol, which would result in very slight historical exceedances (less than a factor of 2) between 1982 and 1992. These exceedances derive entirely from the trenches and pre-date installation of the interim trench cap and final cap. The exceedances do not therefore provide any insight into final cap performance. Our treatment of phenol is cautious as we assume that organic ion exchange resins are comprised entirely of phenol which is immediately available for dissolution and transport in infiltrating water.

Even if the cap performs significantly worse than expected, exceedances are only obtained for boron (peak concentration is a factor of 1.7 times greater than the LLWRAS) and uranium (peak concentration is a factor of seven times greater than the LLWRAS). This demonstrates the robustness of the system with respect to uncertainties in cap performance.

The only other cases in which exceedances are calculated to occur are:

- the GEO_LOW_SORPTION case, in which sorption in the geosphere is neglected;
- the delayed coastal erosion case.

The low sorption case is intended to explore the role of the geosphere in controlling contaminant concentrations. It does not represent a situation which is physically reasonable.

In this case exceedances occur in the PoA for cobalt, copper, nickel, uranium and zinc. No exceedances are calculated for these contaminants in the reference case, which illustrates the role of geosphere sorption in limiting mobility of these contaminants. After the PoA, the only exceedance calculated is for zinc, for which the peak concentration is 1.3 times greater than the LLWRAS. The calculated concentration reduces for the remainder of the assessment. The calculated reductions occur because installation of the final cap reduces the quantities of water entering the repository.

This result demonstrates that, even if geosphere sorption were to be significantly less than anticipated, our optimised site development plan will ensure that the risks of groundwater pollution by non-hazardous pollutants are low.

In the delayed coastal erosion case, exceedances only occur after at least eight thousand years. It is not credible that the repository would remain intact on these timescales. The significance of the impacts is therefore low.

Overall, the assessment cases considered show that, even if key barriers perform significantly less well than anticipated, there is no significant future risk of non-hazardous pollutants entering the environment as a result of disposals at the LLWR.

4.8 Releases of Radiological Contaminants to Groundwater

4.8.1 Impacts during the PoA

We have assessed radiological impacts via the groundwater pathway in reference [14]. There, we demonstrate that assessed annual doses to receptors at the relevant compliance points calculated using our groundwater assessment model are less than 10 μSv for the entirety of the PoA. Dose rates to non-human biota are calculated to be very low.

Monitoring data reveal that there are localised elevated concentrations of tritium and C-14 between the LLWR and the coast. These arise from wastes disposed of in the trenches. The C-14 has migrated via low dilution pathways.

We have therefore also considered, for the early part of the PoA, a 'what if' in which a hypothetical well is sunk between the LLWR site and the coast. It is not credible, as discussed in reference [14], that a member of the public could interact with this groundwater. Similarly, non-human biota would principally interact with water resources recharged by the Upper Groundwater, rather than interact with the elevated concentrations in the Medium Groundwater.

Even if we were to consider our 'what if' calculation in which a well is sunk in the near term, assessed annual doses would be less than 0.3 mSv and our optimised Site Development Plan would ensure that these doses would be as low as reasonably achievable.

We conclude that we prevent the input of radionuclides to groundwater during the PoA.

4.8.2 Impacts after the PoA

We report in reference [15] radiological impacts via the groundwater pathway in the period after the PoA. There, we show that for a range of assessment cases assessed impacts are less than 10^{-6} y^{-1} and we therefore prevent the input of radionuclides into groundwater.

4.9 Environmental Permit Surrender

The GRA [19] sets out the permitting process that applies to a near-surface repository for radioactive waste. Once disposal operations cease and the closure engineering has been installed, it is envisaged that there will be an appropriate period of management control and monitoring after which regulatory control will cease. Our current assumption is that this will be for a period of 100 years.

The surrender of an environmental permit is a regulated process that typically occurs after a facility has been fully closed and demonstrated compliance with its permit conditions and the Site-wide Environmental Safety Case (SWESC). The process involves submitting a formal application to the regulator, supported by a final SWESC that reflects the as-built and closed state of the facility and includes accurate waste records. The regulator will review the application, consult as necessary, and assess compliance against relevant guidance.

5 Monitoring

The environmental monitoring programme has developed over many years with a large historical dataset held. The majority of the data is only available from late 1980s and therefore the data are not available to establish baseline conditions prior to disposals commencing or prior to the use of the site as a ROF. Up-gradient monitoring is therefore used to establish background conditions. More details on the monitoring programme are provided in the '*Monitoring*' report [9].

We define the following objectives for the environmental monitoring programme.

- To confirm that the repository system is not giving rise to unacceptable environmental hazards by direct measurement of its impacts.
- To assess whether the repository system is compliant with the relevant environmental standards.
- To develop and build confidence in the models of the repository system by collecting data that may be used to refine conceptual models or in model parameterisation, calibration or validation.
- To define baseline conditions before specific engineering developments or activities, such as the construction of a new repository component (for example, a vault) or the disposal of wastes at a particular location.
- To confirm that construction activities are not giving rise to unacceptable environmental hazards by direct measurement of their impacts.
- To provide reassurance to stakeholders that the system is safe and is evolving in a manner consistent with the models and assumptions in the ESC.
- To monitor and identify any impact of land contaminated by historical operations on groundwater and surface water.
- To provide information needed to support the development of a WMP and SWESC.
- To work towards a programme that provides technical confirmation of towards the site reference state (e.g. include validation monitoring outline plans)

The monitoring programme is carried out on an annual cycle based on the financial year of April to March. Every third year, enhanced monitoring is performed to capture a wider spectrum of sample points and a broader analysis suite for certain locations. A wider suite of analyses is also undertaken on bulk trench and vault leachates on a triennial basis to understand if the composition of non-radioactive chemical contaminants is changing through degradation of the waste. Findings from these analyses are used to inform changes to the environmental monitoring programme in subsequent years

The full monitoring programme is extensive, in terms of both sampled locations/media and target contaminants/parameters. With regard to non-radiological environmental impacts, our

primary concern is monitoring for non-radiological contaminants in leachate, groundwater and surface water at, near or beneath the site. Specific elements of the programme relating to non-radiological impacts are discussed in detail in Subsections 5.4 and 5.5.

5.1 Management of the Monitoring Programme

An effective management system is an important requirement in ensuring effective outcomes from the environmental monitoring programme. The management system for the monitoring programme forms part of a wider management framework that is outlined in the '*Management and Dialogue*' report [2]. Each stage of the implementation of the environmental monitoring programme is controlled by procedures to ensure that all monitoring data are stored, verified and managed appropriately. The full procedures relating to environmental monitoring are provided in reference [108].

As part of the integrated monitoring programme, an in-house database management system has been developed using a Structured Query Language (SQL) server-based database administered through MonitorPro software. Data import, verification export, report production and review are managed through a series of repository procedures [109]. Data are reviewed, assessed and reported on a quarterly basis throughout the year. The findings are summarised in an annual Monitoring Summary Report produced to meet the requirements outlined in the Compilation of Environment Agency Requirements (CEAR) [110]. Specific details regarding the most recent programme review are given in the 2024-25 Environmental Monitoring Summary Report [111].

The environmental monitoring programme is revised and developed each year, following the annual review of data, to ensure that it continues to provide us with a comprehensive dataset for assessment and interpretation. The review and revision of the environmental monitoring programme also takes into account physical changes (e.g., removal, replacement or suspension of monitoring infrastructure) relating to upgrading of the interim trench cap and work to install the final cap.

5.2 Methodology for Analysis of Monitoring Data

In view of the large volumes of data collected in the environmental monitoring programme, a semi-automated approach is adopted for analysis of these data. There are three main steps in this semi-automated approach. These steps, and the frequency with which they are carried out, are:

- 'unusual' data are automatically identified by comparison to predefined levels (monthly assessment);
- data are trended for visual identification of potential gradual adverse changes (quarterly assessment);

- 'unusual' data and adverse trends are assessed against performance measures, where required, in order to assess the significance of the levels observed (following other summary assessment).

Each of these steps is described in the following subsections.

Where data are identified as exceeding relevant screening criteria, or following an unusual trend, this does not necessarily mean that operations at the LLWR site are leading to an unacceptable impact on the environment. It does, however, allow the significance of the results to be assessed and further investigation instigated where appropriate.

5.2.1 Identification of 'Unusual' Data

A key component of the semi-automated analysis of monitoring data is the automatic identification of 'unusual' data. This is accomplished using two separate Unusual Data Identification Levels (UDILs):

- Mean + 4 Standard Deviations (Mean +4SD);
- Data Quality Flag (DQF).

These UDILs (described below) flag data that are either unusual for a single location, unusual for a water system or both. The comparison of data against UDILs is carried out automatically when laboratory data are imported into the Monitor Pro database system. Data that exceed the relevant UDILs are flagged by the database system.

Mean + 4 standard deviations

The Mean +4SD level is used, as defined in the CEAR, to flag analytical laboratory data that are 'unusual' for an individual location. Mean +4SD are moving values calculated at four standard deviations above the mean of the previous twelve results in the database. These are automatically calculated within the database for all analytical data where at least five previous data values exist. Mean +4SD results are investigated and reported to the Environment Agency on a monthly frequency.

Data Quality Flag

DQFs are thresholds used to identify environmental monitoring data that could fall outside the expected or 'usual' range. The early identification of data that are unusual allows the reasons behind the elevated result to be investigated in a timely fashion

At LLWR, DQFs are applied to both field measurements and laboratory data to:

- detect adverse trends or anomalies;
- trigger investigations when values exceed predefined thresholds;
- support regulatory reporting and internal performance reviews

DQFs for field measurements are defined for an individual sampling location, whilst DQFs for laboratory data have been defined for each environmental system. DQFs have been statistically calculated from baseline datasets, unless overarching regulatory levels are

applicable. Prior to calculating DQF values, the baseline data are sampled to exclude the highest and lowest 5% of the data for each determinand. This step is undertaken to remove outlying data values. The DQF for each determinand is then set at three standard deviations from the mean of the remaining data set. As the DQF value lies within the usual range for each determinand, it is expected that some of the DQF exceedances are actually within the usual range for the system. This ensures that all data that are close to the maximum value observed are flagged for further investigation.

In some cases, not related to laboratory data, DQFs are defined to identify data that are outside the physical minimum or maximum of the data range for an individual location, e.g. stream levels below a weir v-notch level. Data identified as exceeding a DQF can then be reviewed and further investigation undertaken. The DQF levels are regularly reviewed and revised levels are implemented at the start of the monitoring year [112].

5.2.2 Data Trending

Data trending charts are produced on a quarterly basis for all numerical data collected during the previous quarter. These charts allow the visual identification of adverse trends at a location, which may not be captured as a UDIL, i.e. if a trend is progressively increasing at a rate low enough not to trigger a Mean +4SD exceedance and continues to be below the DQF value. Internal trending reports are produced quarterly for review [113, 114].

5.2.3 Assessment Against Performance Measures

Performance measures have been defined, against which the environmental impact of the repository on river water, surface water and groundwater can be assessed. Comparison of monitoring results with performance measures does not necessarily indicate the presence of pollution, as background levels may already exceed the assessment criteria in some cases.

LLWRAS

For radiological groundwater quality and all river water and surface water quality laboratory results, measured concentrations are compared with LLWRAS.

The current groundwater radiological LLWRAS were introduced during 2019 [115]. They correspond to the activity concentration in water consistent with an annual radiation dose of 20 μSv from each specific radionuclide. The calculation addresses all exposure pathways potentially associated with use of a water abstraction well i.e. including consumption of water and contaminated animal products. The assessment of the groundwater pathway during the PoA [14] only considers dose from consumption of abstracted well water. These LLWRAS are also utilised to assess the impact of the LLWR on river and surface water quality as the predominant exposure pathway aligns with that for groundwater (water for livestock) [15]. We recognise that although the annual dose level of 20 μSv level is consistent with the previous guidance [116] it is higher than the 10 μSv level provided in the new guidance issued in 2021 [29] and we will review the radiological LLWRAS accordingly.

Monitoring data relative to the current radiological LLWRAS are presented in references [9] and reference [14].

Non-radiological LLWRAS are discussed in Subsection 2.1.2 and are presented in Table 2.1.

Monitoring Assessment Levels (MALs)

Monitoring data are compared to MALs to assess the non-radiological impact of the LLWR site operations, including disposals, on groundwater. In general, MALs are set at the LLWRAS or the level discernible above the background, whichever is higher. The discernible level has been calculated at the 95th percentile of the baseline dataset. The use of MALs will result in a reduction in the number of false positives [117].

Baseline

The baseline is the naturally occurring level of a contaminant measured prior to construction of a facility, or in the case of an existing facility, the level measured upstream of the site. For some contaminants, the baseline level may exceed the assessment standard.

Baseline values have been derived for rainwater, surface water and groundwater, as discussed in Subsection 5.3.

Baseline and background are closely related but are not the same thing, even though they are sometimes derived from the same datasets. Baseline water quality represents a defined reference condition established at a specific time and location for regulatory assessment, whereas background water quality describes the broader ambient range of concentrations arising from natural and diffuse sources; the terms are therefore related but not synonymous.

Using a baseline provides a clear, fixed reference for assessing change, whereas relying on background alone would be ambiguous and variable, making results harder to interpret and defend when naturally high concentrations are present.

5.3 Baseline Water Quality

Baseline quality was derived for rainwater, surface water and groundwater entering the LLWR site in a review of historical and more contemporary data conducted in 2026 [118]. The baseline values are summarised in the following subsections.

5.3.1 Rainwater Baseline

Baseline rainwater quality was derived from a limited number of samples taken between 2008 and 2024. The resulting concentrations are presented in Table 5.1. Local rainwater is characterised by the presence of low concentrations of dissolved solids, principally chloride, sulphate, calcium, magnesium and sodium. The elevated concentration of Total Organic Carbon (TOC) was noted as unusual and was attributed to likely contamination by windblown organic material. It is unknown whether the trace concentrations of heavy metals, arsenic, copper, lead, mercury, nickel and zinc observed were actually present in the rainwater or were due to the windblown organic material or residual impurities arising from the sampling equipment.

Table 5.1: Rainwater baseline quality

Determinand	No. tests	No. >LoD	Minimum ($\mu\text{g l}^{-1}$)	Maximum ($\mu\text{g l}^{-1}$)	95th percentile
Arsenic	11	6	0.43	6.34	5.62
Barium	10	7	1	20	20
Beryllium	9	0	0.09	1	1
Boron	7	7	3.7	13.5	12.2
Cadmium	11	0	0.08	1	1
Chloride	22	22	4530	129000	102995
Chromium	11	9	0.4	10	6.5
Cobalt	6	0	0.07	1	1
Copper	11	9	0.9	40	29.5
Cyanide	2	1	20	20	20
Iron	6	6	10.5	37.1	35.9
Lead	9	4	0.36	6.25	4.95
Mercury	9	1	0.05	0.1	0.1
Nickel	11	8	0.3	31.4	24.8
Nitrate as NO ₃	9	8	50	10990	10275
Nitrite	9	2	7	900	740
Phenol	4	1	0.1	1.8	1.6
Selenium	7	5	0.16	1.96	1.67
Sulphate as SO ₄	9	9	1400	6600	6244
Vanadium	9	5	0.05	10.95	10.57
Zinc	11	11	28	250	221

A comparison was drawn in reference [58] between the local LLWR rainwater quality presented above and that measured in samples taken at Devoke Water, some 10 km to the southeast of the LLWR and at a higher elevation. The samples from Devoke Water were found to be higher in sulphate and nitrate, but lower in chloride, sodium and potassium than the local LLWR samples. This was attributed to the closer proximity of the LLWR to the coast.

It was noted in reference [58] that recharge to groundwater will show an increase in concentrations over rainwater as a result of evapotranspiration. Thus, the baseline for recharge will be higher than the rainwater baseline by a factor related to the proportion of rainfall that recharges groundwater.

5.3.2 Stream Water Baseline

Stream water baseline quality was derived from a limited number of samples of water collected from the East-West Stream, immediately upstream of the LLWR site, between 2008 and 2024 inclusive. This baseline allows data from onsite streams to be compared against concentrations prior to the stream entering the site. The baseline has been calculated as the 95th percentile of the baseline dataset. In a small number of samples the historical LoD was elevated in comparison to the rest of the dataset. These values were removed from the baseline dataset in order to minimise the potential for overestimation of the baseline [118]. The results are presented in Table 5.2.

Table 5.2: Stream water baseline quality

Determinand	No. tests	No. >LoD	Minimum ($\mu\text{g l}^{-1}$)	Maximum ($\mu\text{g l}^{-1}$)	95th percentile
Arsenic	9	9	0.84	11.43	8.50
Barium	12	12	20	272	191
Beryllium	7	3	0.2	1	1
Boron	12	12	18	50	39
Cadmium	10	3	0.2	1	1
Chromium	10	9	0.90	10	9.55
Cobalt	5	4	1	7.69	6.69
Copper	12	12	3.35	15	14.92
Cyanide	2	1	20	20	20
Iron	5	5	1720	15600	13584
Lead	12	7	0.13	34	26.80
Mercury	11	4	0.05	2	1.88
Molybdenum	6	3	0.15	1	0.87
Nickel	12	10	3.08	20	15.05
Nitrate as NO ₃	10	10	4200	24300	23805
Nitrite	11	9	0.24	274	264
Phenol	6	1	0.1	0.52	0.52
Phenol Index	2	1	50	50	50
Selenium	10	6	0.6	4	4
Sulphate as SO ₄	10	10	4700	23900	23495
Vanadium	11	7	1.1	23.1	16.9

Determinand	No. tests	No. >LoD	Minimum ($\mu\text{g l}^{-1}$)	Maximum ($\mu\text{g l}^{-1}$)	95th percentile
Zinc	12	12	4.5	129.5	107.8

It is evident that the upstream baseline exceeds the LLWRAS for arsenic, copper, iron, lead, mercury, nickel and zinc.

5.3.3 Groundwater Baseline

The following groundwater baselines were derived [118]:

- a regional baseline for the sandstone aquifer, from Environment Agency data;
- a local baseline for the shallow drift deposits and sandstone aquifer, from site data.

The resulting groundwater quality baselines are presented in Table 5.3. The regional baseline was derived from data provided by the Environment Agency's regional water quality monitoring network. Data from a single borehole at Bank House Park, Gosforth, believed to be screened in the Triassic Sandstone, were used. The calculation used all available data from the period 2008 to 2025 inclusive. The regional groundwater is of generally good quality, although aluminium, iron, manganese and lead are above drinking water standards.

Local groundwater baselines were derived from samples extracted at twenty wells in seventeen boreholes predominantly to the east of the site. Boreholes lying within the range of tritium migrating from the trenches were not considered as these would not be indicative of true baseline conditions.

Table 5.3: Groundwater baseline quality

Determinand	Local groundwater (from site data) ($\mu\text{g l}^{-1}$)	Regional groundwater (from Environment Agency data) ($\mu\text{g l}^{-1}$)
Aluminium		892
Ammoniacal nitrogen	817	30
Ammonium as NH ₄	2761	
Antimony		1
Arsenic	19.2	1.1
Barium	501.8	63.7
Benzene		0.1

Determinand	Local groundwater (from site data) ($\mu\text{g l}^{-1}$)	Regional groundwater (from Environment Agency data) ($\mu\text{g l}^{-1}$)
Beryllium	10	1
Boron	67	100
Cadmium	1.77	0.11
Chloride	206800	23650
Chromium	5.61	0.92
Chromium (III)	5.61	
Chromium (VI)	5.61	
Cobalt	2.61	1
Copper	11.3	69.8
Fluoride	503.5	76.5
Iron	9770	496
Lead	4	10.7
Magnesium	23020	2865
Mercury	1	0.01
Molybdenum	11	3
Nickel	10	4.20
Nitrate as NO ₃	9502	
Nitrite	100	4
Phenol	1.50	0.14
Phenol index	50	-
Selenium	5.9	1

Determinand	Local groundwater (from site data) ($\mu\text{g l}^{-1}$)	Regional groundwater (from Environment Agency data) ($\mu\text{g l}^{-1}$)
Tin (inorganic)		2.5
Vanadium	14.5	2
Vinyl chloride		0.1
Uranium		0.5
Zinc	42.49	74.98

Compared with the regional baseline, the local groundwater baseline is of poorer quality.

5.4 Non-radiological Monitoring Plan

5.4.1 Objectives and Approach

With regard to the HRA documented in this report, our primary concern is monitoring for non-radiological contaminants in leachate, groundwater and surface water near or beneath the site. Monitoring for radiological contaminants is reported in reference [14].

Objectives for the non-radiological monitoring correspond with those of the wider monitoring programme as stated earlier in this section. Specific objectives for monitoring of leachate, groundwater and surface water are listed in the following subsections.

As with the wider monitoring programme, non-radiological monitoring predominantly runs on a triennial cycle with additional non-radiological analysis also carried out on an ad-hoc basis as driven by review of the data collected. Thus, monitoring for key analytes is carried out annually, with an extended programme being undertaken during the third year of the cycle to capture a wider spectrum of sample points and a broader analysis suite for certain locations. The non-radiological analytes that are included within the triennial extended programme are informed by the chemical composition of leachate, identified through bulk leachate analysis carried out on leachate from Vault 8, Vault 9 and the trenches during the second year of the cycle. Where an analyte is found in leachate at a concentration considered to be of concern if released to the environment it is added to the standard suite of analysis for leachate and other water systems to assess if it is having an impact on the surrounding environment. A crosscheck against key contaminants identified from waste inventory records and via ESC studies is also carried out against these analysis suites to identify if further specific analysis is required for these analytes to assess their presence.

Monitoring for potential radiological contaminants in groundwater is extensive, with frequent analysis conducted on samples taken at numerous locations within and outside the site boundary, up-gradient and down-gradient of the repository and at a range of depths. The full extent of this radiological groundwater monitoring is described in the most recent monitoring review report [111] and is considerably more comprehensive than groundwater monitoring undertaken for landfill sites.

By comparison with the radiological groundwater monitoring, analysis for potential non-radiological contaminants is more targeted. While non-radiological analysis is conducted regularly on samples taken at a range of up-gradient and down-gradient locations, in Upper, Medium and Deep groundwaters, the spatial and temporal density of this analysis is less than that for the radiological component of the monitoring programme (see Subsection 5.4.4). Nevertheless, the spatial extent of non-radiological monitoring is still more comprehensive than the equivalent monitoring mandated for landfills and is considered sufficient to capture any potential contamination issues or adverse trends.

As an extra safeguard, additional targeted non-radiological analysis is undertaken as considered appropriate. Targeting of this additional non-radiological monitoring is facilitated by knowledge of actual pollution pathways indicated by tritium migrating from the trenches, as described in Subsection 5.4.2.

We consider our approach to non-radiological monitoring to be both risk-based and proportionate to the non-radiological hazard presented by the waste.

5.4.2 The Role of Tritium in Site Characterisation

Tritium plays a key role in site characterisation and understanding contaminant migration at the LLWR because it behaves conservatively and is transported with groundwater at the linear flow velocity. This is in contrast to other contaminants that experience (sometimes very considerable) retardation relative to the groundwater due to sorption. While the monitoring data in this HRA focus on the impact from non-radioactive contaminants, it is helpful to consider what can be inferred from the distribution of tritium as more extensive monitoring data are available for tritium than for other contaminants.

Tritium has been disposed to the trenches and release of tritium has occurred to groundwater beneath the site and westwards towards the coast [9]. Tritium has been detected across most of the central and north-western parts of the site and between the site and the coast, the highest concentrations being clustered around the trenches. It is evident that tritium migration has occurred in both the Upper and the deeper Regional Groundwaters, with significant downward migration. The monitoring data indicate low levels of tritium (less than 100 Bq l^{-1}) in the Upper Groundwater in the area to the south of the trenches and at two points in the surface water drainage system which discharge to the Drigg Stream [58]. There is uncertainty as to whether these elevated surface water concentrations derive from a leak in the leachate line at the southern end of the trenches or are evidence of a groundwater pathway between the disposal facility and the Drigg Stream.

Extension of the cut-off wall as part of installation of closure engineering will remove this pathway.

A general declining trend in tritium in the Upper and Intermediate Groundwater, is evident over the longer term with a slow decline in tritium in the Medium Regional Groundwater and an increase in tritium in the Deep Regional Groundwater, as shown in Figure 5.1 to Figure 5.4. This declining trend is attributed to radioactive decay, progressive leaching and the combined effect of capping of the trenches and cut-off wall installation. The interim cap has decreased infiltration into the waste and installation of the cut-off wall restricting lateral migration.

The slight increase in tritium in Deep Groundwater since 2008 is due to the integration of site monitoring into a single programme, which allowed targeted monitoring in areas of concern and is not generally related to an increase in tritium activity concentrations.

Figure 5.1 shows that the number of off-site Upper Groundwater monitoring points included for analysis was reduced in 2021 as a good baseline dataset had been collected and the majority were too shallow to show any impact from the site. Some have been retained for groundwater level monitoring and remain available for future analysis if required.

The distribution of tritium has been and will continue to be well-characterised in both space and time by a large number of measurements. Such information supports development of the hydrogeological conceptual model of the site. It also provides valuable evidence that the engineered barriers are performing as expected. For example, construction of the cut-off wall to the east of the trenches led to a significant reduction in tritium concentrations in the railway drain and further east [85].

Monitoring of tritium concentrations can provide an early indication of potentially new migration pathways from the LLWR, for example in the event of disruption to an engineered barrier, some other failure of the disposal system or evolution in hydrogeological conditions. Thus, detailed data on the distribution of tritium allow monitoring of non-radioactive contaminants to be targeted at particular areas of interest. When deemed appropriate, for example following an unexpected measurement or some observed change in the disposal system or environment, more detailed non-radiological investigations can be carried out at specific locations suggested by the tritium data. This approach is considered to be proportionate to the risk posed by the non-radioactive component of the waste inventory.

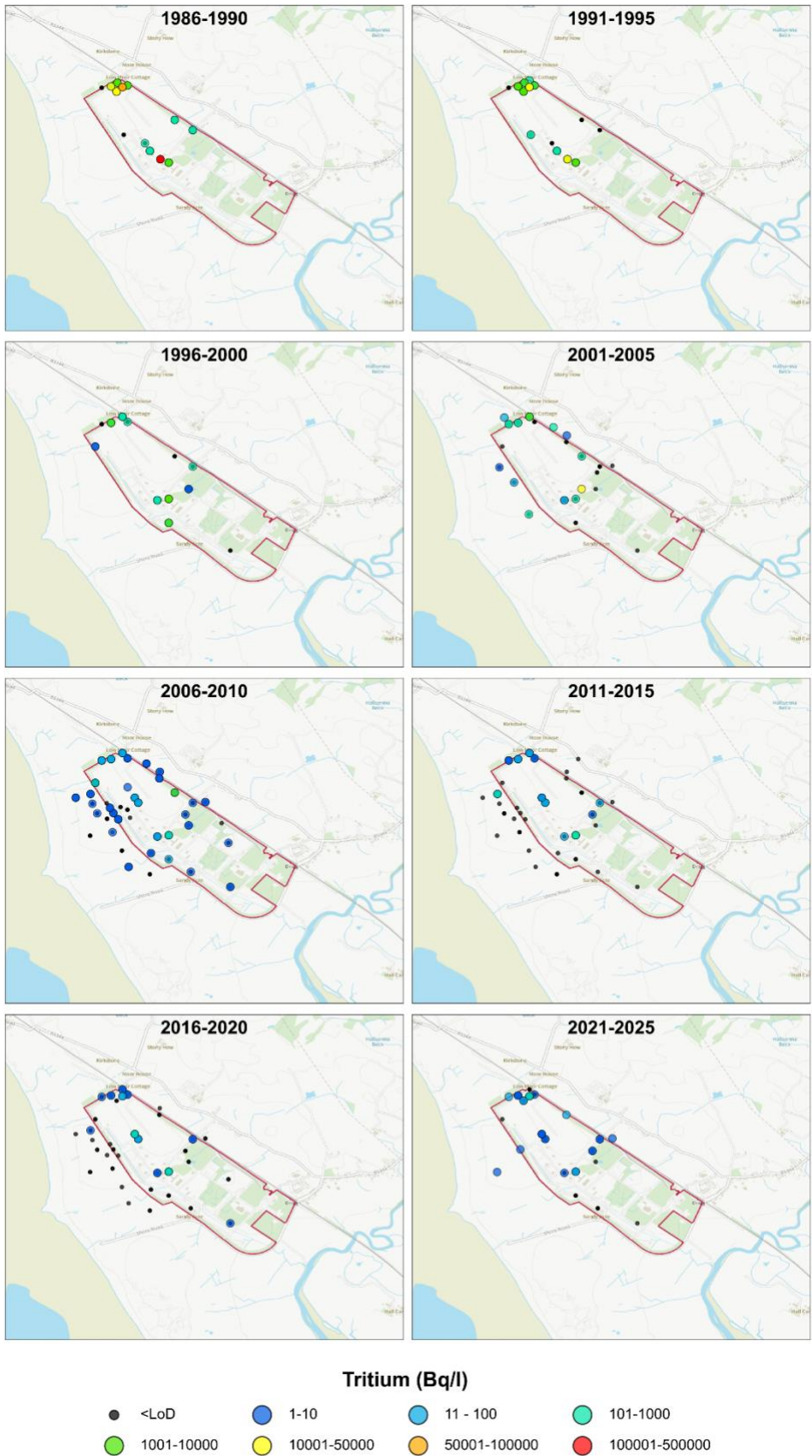


Figure 5.1: Maximum tritium concentrations in Upper Groundwater 2008-2025

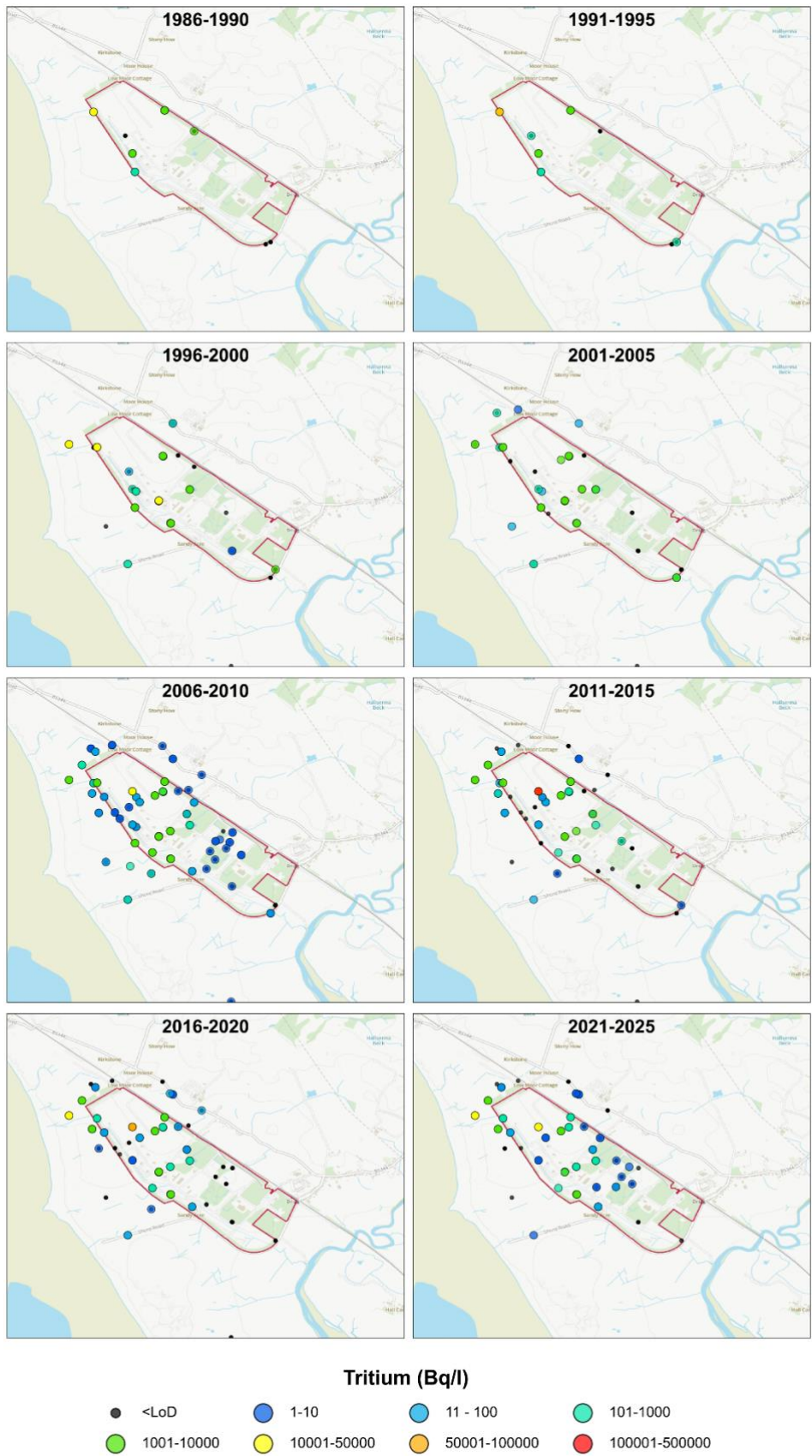


Figure 5.2: 2024 Maximum tritium activity within the Medium Regional Groundwater

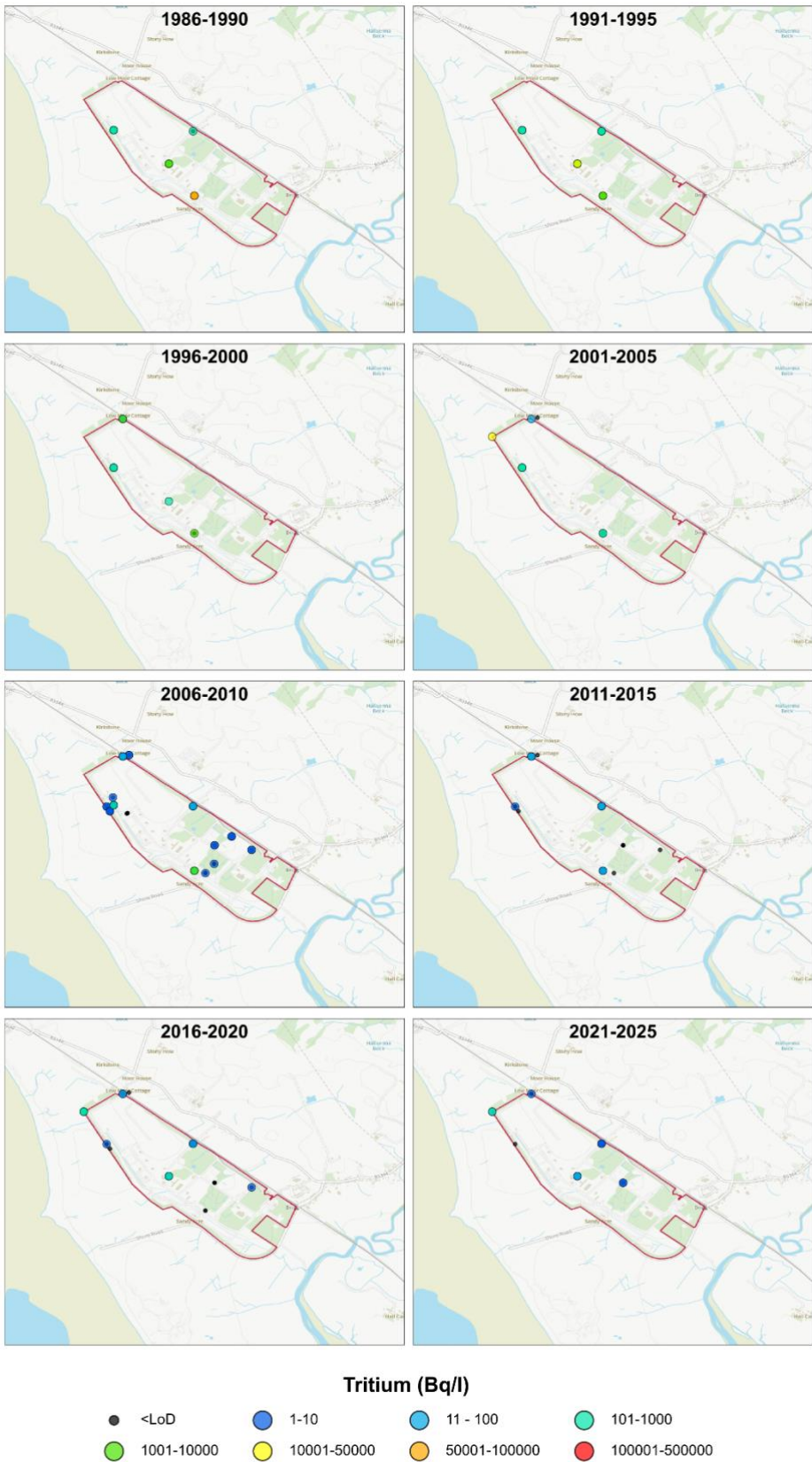


Figure 5.3: 2024 Maximum tritium activity within the Intermediate Regional Groundwater

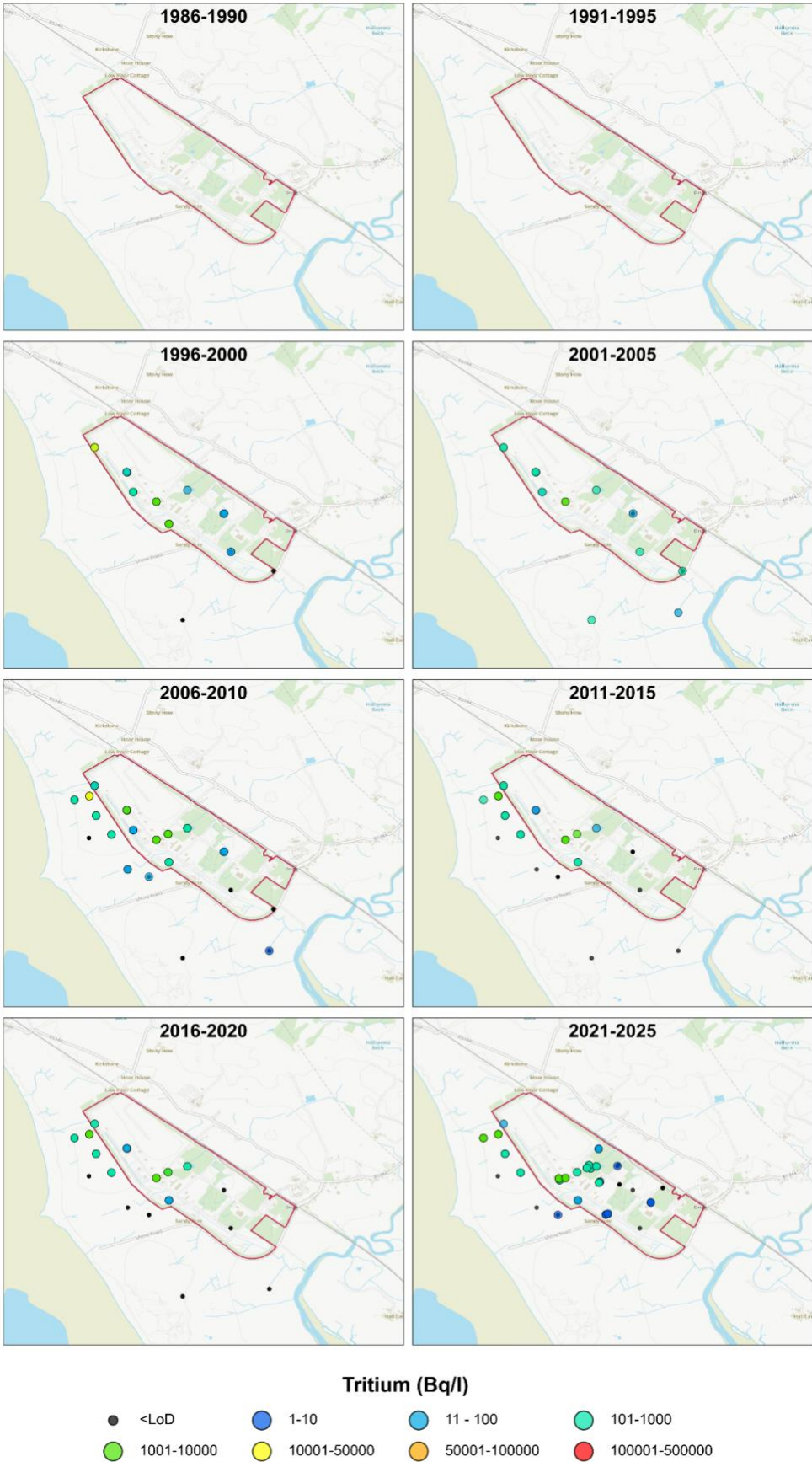


Figure 5.4: 2024 Maximum tritium activity within the Deep Groundwater

5.4.3 Leachate Monitoring Plan

The principal specific objectives of the leachate monitoring plan are to:

- provide data on the hydrology of the trenches;
- provide data on the volume of leachate discharged from each individual trench;
- provide data on the total flow from each of Vault 8 and Vault 9;
- provide data for the assessment of trench cap integrity;
- provide data relating to source composition;
- provide data on changes to the source term;
- record discharges to the Irish Sea via the marine pipeline;
- record other authorised discharges to the environment.

Leachate levels

Leachate level measurements are made by automatic loggers installed in trench cap probes, as shown in Figure 5.5. This dataset is further supplemented with measurements carried out by manual dipping on a quarterly basis in each of the remaining probe holes across the trench cap.

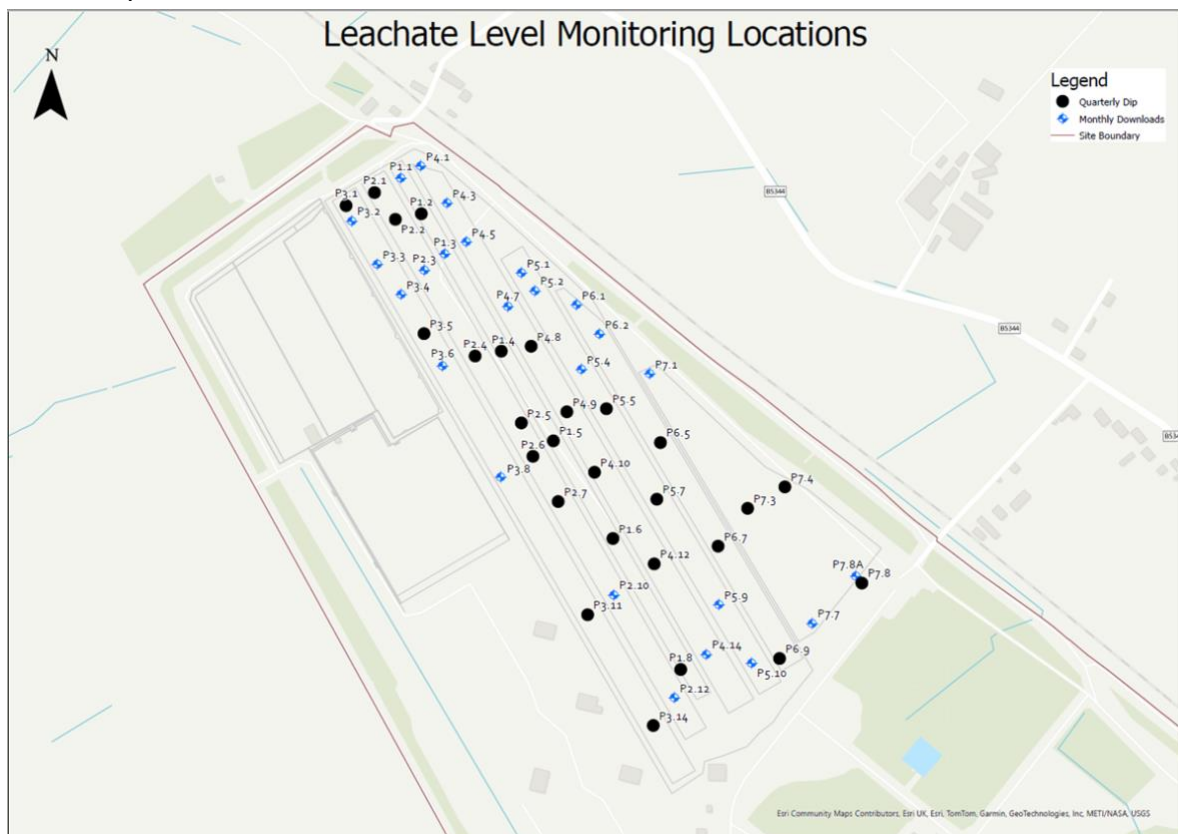


Figure 5.5: Leachate level monitoring locations (2025/26)

Flow measurements have been previously taken in monitoring chambers located at the end of each trench and each of the pumping chambers in Vault 8 and 9. The data logging

equipment in each of the end-of-trench monitoring chambers has reached its operational battery life expectancy. Measurements from these data-loggers has been suspended and a replacement system is expected to be installed in 2027.

It will take a couple of years to design and replace the installed equipment and therefore a temporary system to measure the daily number of pump activations at each end of trench monitoring pit is to be installed in 2025-26. This system will monitor trends in leachate production at the end of trench monitoring locations whilst the new system is being designed and installed.

The data collected are analysed in several ways in order to ascertain cap performance and changes within the system. Leachate levels are also compared with historically measured maxima and minima at each probe hole in order to assess whether data remain within the bounds of that collected previously.

Details of each discharge to the Irish Sea via the marine pipeline are recorded on an automatic logging system.

Leachate quality

Leachate quality monitoring locations are shown in Figure 5.6. Bulk leachate quality samples are collected in the second year of the triennial monitoring cycle. Analysis of these samples informs the water quality monitoring suite across all water systems. Where an analyte is found in bulked leachate samples, at a concentration considered to be of concern if released to the environment, it is added to the standard suite of analysis for leachate and other water systems. Enhanced monitoring of all waters (leachate, groundwater and surface water) is performed in the third year of the triennial cycle to capture any changes in bulk leachate quality noted the previous year.

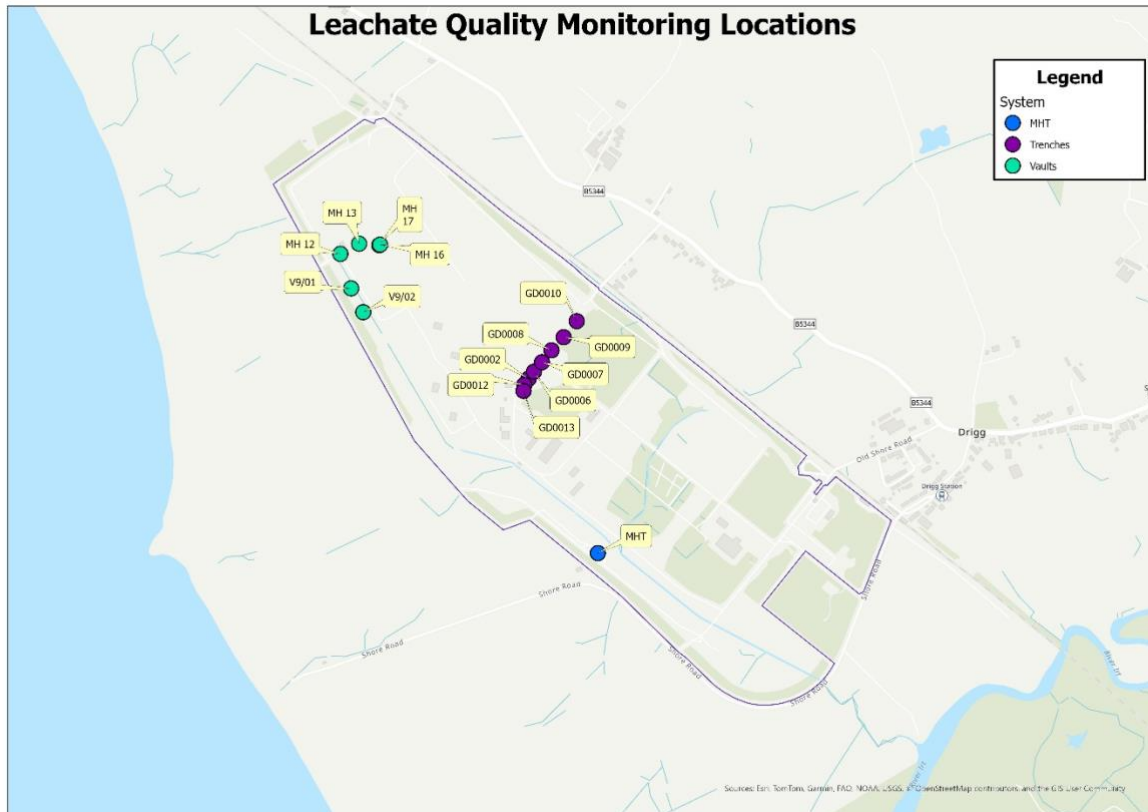


Figure 5.6: Leachate quality monitoring locations (2025/26)

Triennial bulked leachate analyses were carried out as part of the 2024/25 environmental monitoring programme. The reported results from the bulked leachate analysis to date have not identified any additional contaminants of concern that needed to be added to the environmental monitoring programme.

Discharges from Vault 8 and Vault 9 and the leachate chambers at the southern ends of the trenches are sampled and analysed on a quarterly basis to provide information on the source term.

During the third year of the triennial monitoring cycle, these Vault 8, Vault 9 and end-of-trench samples are analysed for a non-radiological suite to indicate if there have been any changes to the source term. In all years, measurements are made in the field for specific conductance, pH and temperature.

Additional analyses may be performed on leachate samples as required to provide further insight into the source term.

Leachate from the trenches and vaults is collected in the MHT. Non-radiological analysis for cadmium, chromium, copper, lead, nickel, phosphorus, zinc, total nitrogen (including nitrate, nitrite and total ammonium) and visible solvent is carried out on a bulked sample from the MHT on a monthly basis to meet the Environmental Permit requirement for analysis of non-consented parameters.

A monthly 'spot' sample is also taken to confirm adherence to the Environmental Permit requirements (NPSWQD002191) for TSS (300 mg l⁻¹), COD (400 mg l⁻¹), Fe (100 mg l⁻¹) and pH (5.00-9.00) and an absence of visible signs of oil or grease.

5.4.4 Groundwater Monitoring Plan

The principal objectives of the groundwater monitoring plan are to:

- provide data on groundwater levels;
- provide data on the performance of site engineering;
- provide confirmation or facilitate improvement of the hydrogeological conceptual model;
- characterise groundwater chemistry upgradient of the LLWR;
- identify any issues relating to contaminant emissions from the LLWR (e.g. exceedances, unusual data, adverse trends);
- monitor long-term variations in groundwater chemistry.

Groundwater levels

Groundwater level monitoring data are collected at key locations across the LLWR site and within the surrounding area both upstream and downstream of the site. The locations were chosen to provide extensive coverage, allowing a comprehensive assessment of the groundwater environment within and surrounding the LLWR site.

The annual review of monitoring data considers whether additional groundwater level monitoring is required in the following year.

For the 2024/25 reporting period, groundwater level data were collected at the following locations and frequencies (see Figure 5.7).

- Sixty-two locations with automatic groundwater level data loggers installed, recording measurements at six-hourly intervals with data downloaded quarterly. Loggers are installed in more locations than is required to provide sufficient redundancy to mitigate against data loss.
- One hundred and forty-five locations were manually dipped on a quarterly basis.

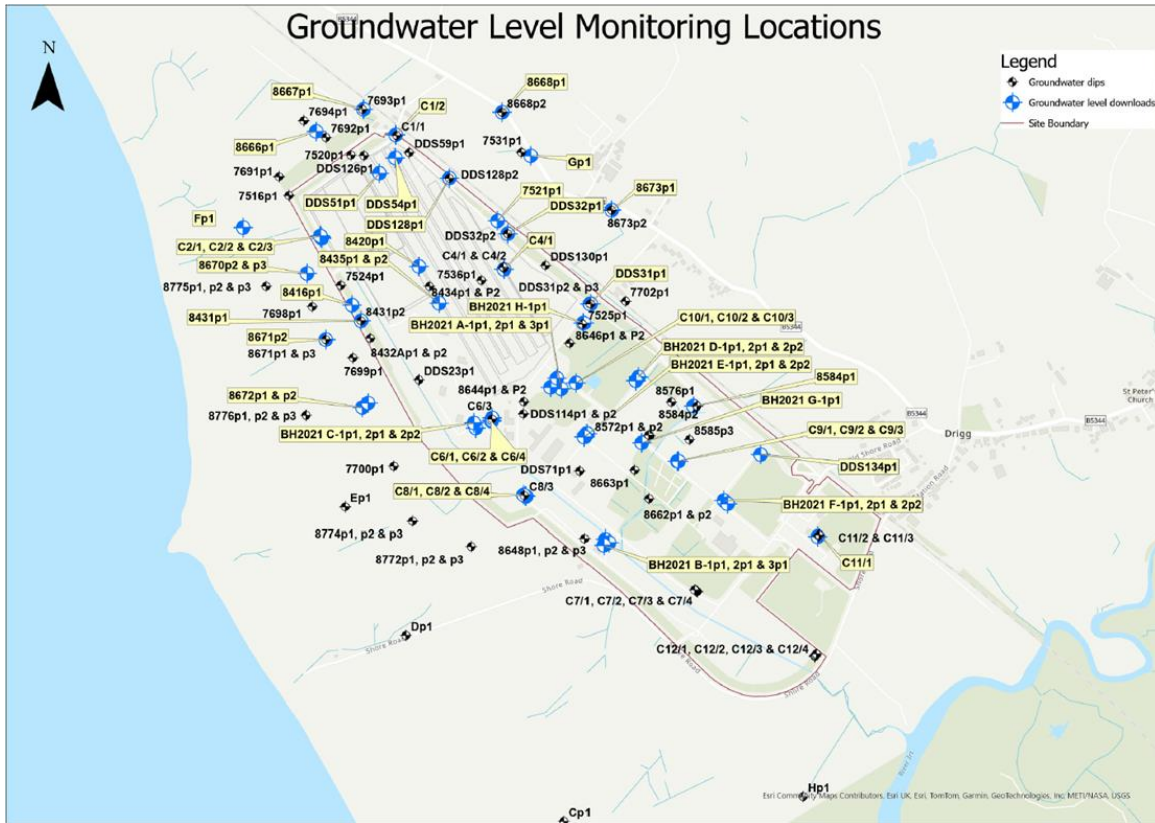


Figure 5.7: Groundwater level monitoring locations

Groundwater quality

Groundwater samples were regularly collected for chemical analysis at eighty locations in 2024/25, both within and outside the LLWR site licensed boundary. These locations are chosen to:

- assess the impact of the LLWR site on the groundwater beneath and down gradient of the site;
- monitor the performance of the existing cut-off wall;
- monitor groundwater upstream of the LLWR;
- monitor down-hydraulic gradient of potential areas of land contamination.

Groundwater quality sampling locations for radionuclides and non-radiological contaminants are shown in Figure 5.8 and Figure 5.9, respectively. These figures show sampling locations, type of analysis and frequency of sampling.

As discussed in Subsection 5.4.2 the results of tritium monitoring are used to focus the non-radiological monitoring strategy.

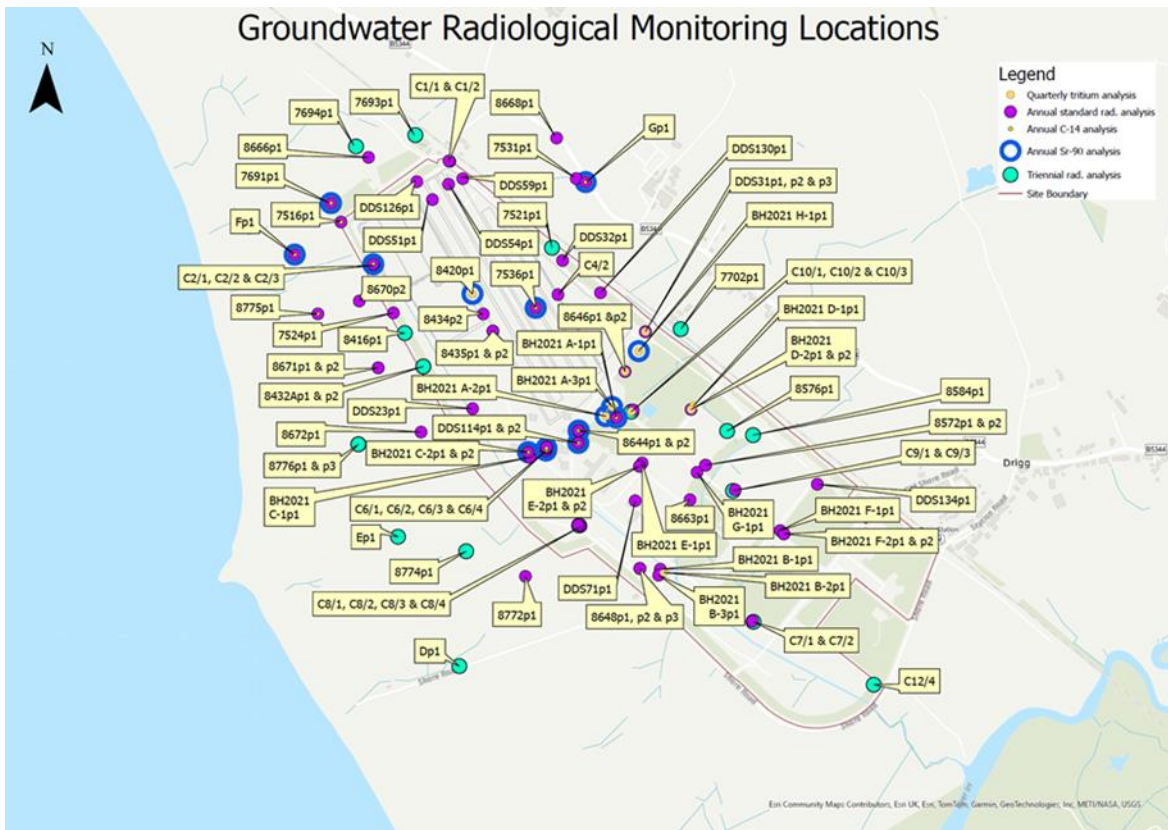


Figure 5.8: Groundwater monitoring locations for radionuclides

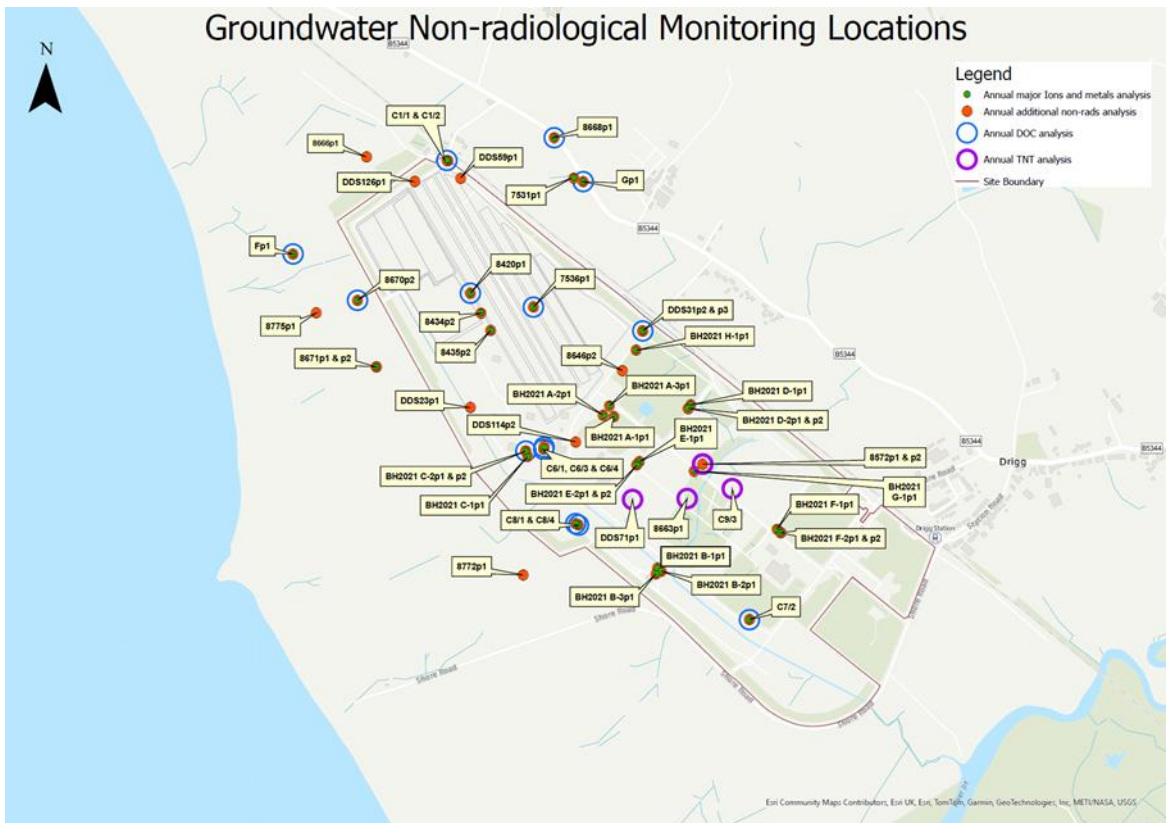


Figure 5.9: Groundwater monitoring locations for non-radiological contaminants

Between 2016 and 2025, the triennial groundwater analysis suite consisted of the following determinands.

- Metals: barium, beryllium, cadmium, chromium, cobalt, copper, lead, mercury, molybdenum, nickel, vanadium, zinc.
- Non-metals: arsenic, boron, selenium.
- Water chemistry indicators: alkalinity (HCO₃), ammoniacal nitrogen, calcium, iron, manganese, potassium, sodium.
- Others: total cyanide, total phenols, nitrate, nitrite, TOC.

Between 2017 and 2023, analysis for metals and non-metals has been undertaken annually on all quarterly samples and selected annual samples. Metals analysis frequency was reduced to triennial in 2024. Antimony is no longer included in the annual analysis suite, while fluoride analysis is undertaken only for selected samples (from locations with a high tritium activity).

Following input into the Monitor Pro database, analysis of the non-radioactive groundwater concentrations is performed according to the methodology set out in Subsection 5.2. Further investigations are undertaken upon exceedance of UDILs, exceedance of LLWRASs or identification of adverse trends.

Where data are identified as exceeding relevant screening criteria, or following an unusual trend, this does not necessarily mean that there is an unacceptable impact on the environment. It does, however, allow unexpected data to be identified and the significance of the results observed to be assessed.

In addition to laboratory analysis of groundwater for specific contaminants, several field parameters (physical and chemical measurements made in the field) are recorded during sampling of the groundwater: Dissolved Oxygen (DO), specific conductance, pH, redox potential and temperature. Such measurements are taken prior to the collection of each groundwater sample. These field parameter readings are used to identify changes in the overall groundwater chemistry.

When deemed appropriate, the regular non-radiological monitoring programme is supplemented by additional targeted sampling and analysis as described in Subsection 5.4.1.

5.4.5 Surface Water Monitoring Plan

The principal objectives of the surface water monitoring plan are to provide data:

- for the assessment of engineered barrier performance;
- for developing understanding of site hydrology;
- to identify any potential repository-derived contamination of surface waters or other changes in surface water chemistry.

Surface water levels and flows

Surface water levels and flows within the Drigg Stream and the East-West Stream provide baseline information used in the assessment of engineered barrier performance and are used to constrain water balance calculations and models of site hydrology. This includes the measurement of levels of stream water as well as drainage water.

Five locations along the two streams are monitored at 15-minute intervals using automatic data loggers. The locations are as follows:

- Frog Weir on the East-West Stream;
- Lonesome Pine Weir and GD0011 Weir, both of which are on the Drigg Stream;
- two on-site land drain weirs discharging into the Drigg Stream (OF1 Weir and OF2 Weir).

These locations are shown in Figure 5.10.

Surface water land drains on-site are also monitored at 15-minute intervals using automatic data loggers. One off-site land drain (GF0004) discharges into the East-West Stream upgradient of Frog Weir.

Stream water and drainage water are dealt with separately as stream water is influenced by upstream concentrations. If non-radiological contamination in stream water were to be assessed against EQS concentrations the assessment would not portray the impact that the site operations have on stream water. Drainage water does not have the elevated baseline issue and therefore can be assessed against EQS concentrations.

Stream water from nine locations, including two locations off-site along the River Irt, was sampled between 2016 and 2025 inclusive. Six of these locations were sampled on a quarterly basis with an annual sample collected from the three remaining locations. Data collected from stream water samples are compared against the stream water baseline as described in Subsection 5.3.2.

Five surface water land drains and six off-site surface water accumulations were also sampled between 2016 and 2025. Three of these drainage water locations were sampled on a quarterly basis, one was sampled on a biennial basis and one was sample annually.

A standard non-radiological suite, biological oxygen demand (BOD) and total suspended solids (TSS) analysis, were undertaken on all of the river water and surface water samples. BOD is a measure of the amount of oxygen required by aerobic biological organisms to breakdown the organic matter that is present and, therefore, elevated results could indicate the presence of an increased concentration of pollutants. TSS analysis is undertaken to measure the sediment load in the water samples.

In the third year of the triennial cycle, an additional enhanced suite of non-radiological analysis was undertaken, consisting of the following determinands.

- Metals: barium, beryllium, cadmium, chromium, copper, lead, mercury, nickel, zinc.
- Non-metals: arsenic, boron, selenium.
- Others: total cyanide, total phenols, nitrate, nitrite, TOC.

Additional metals and non-metals analysis was carried out on an annual basis at all river water and surface water locations between 2019 and 2023 inclusive.

All samples are also characterised in the field for specific conductance, pH and temperature.

5.5 Review of Monitoring Data

5.5.1 Leachate Data Review

Leachate levels

Dipped levels in each trench are shown in Figure 5.11 to Figure 5.18. These figures show the average leachate levels during 2024 alongside the average levels in the previous 10 years (2014 to 2023). In general, probe holes show a reduced leachate level compared to their historical average levels, indicating a general reduction in leachate levels over time due to the placement of the interim cap. No leachate levels were observed above the membrane level at any probe on the trench cap.

Trending of the leachate level and temperature results is undertaken on a quarterly basis to ensure continuity of the data against previous results. Where anomalous results are reported, the location is investigated to ascertain the cause of the result and, where necessary, remedial action is undertaken.

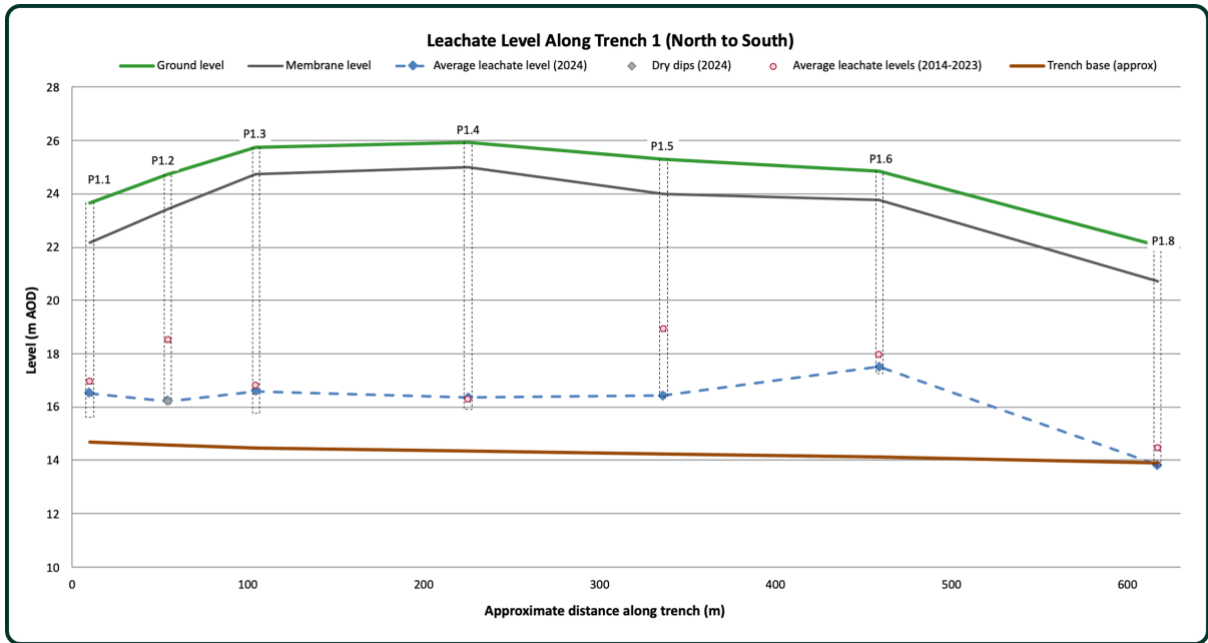


Figure 5.11: Leachate level along Trench 1

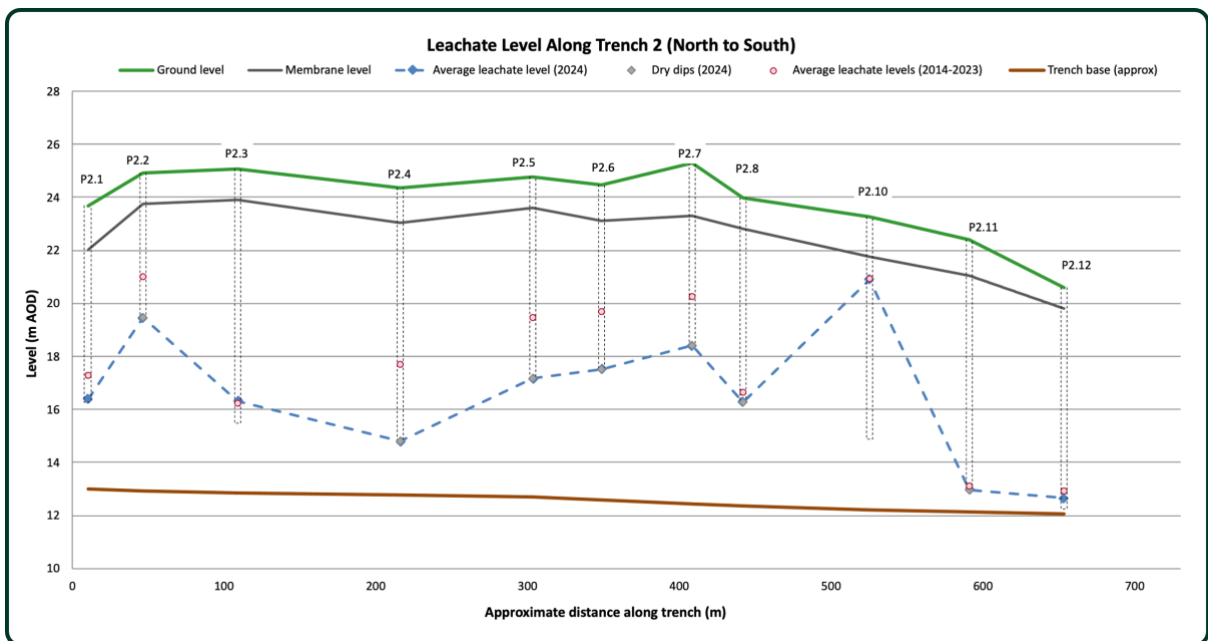


Figure 5.12: Leachate level along Trench 2

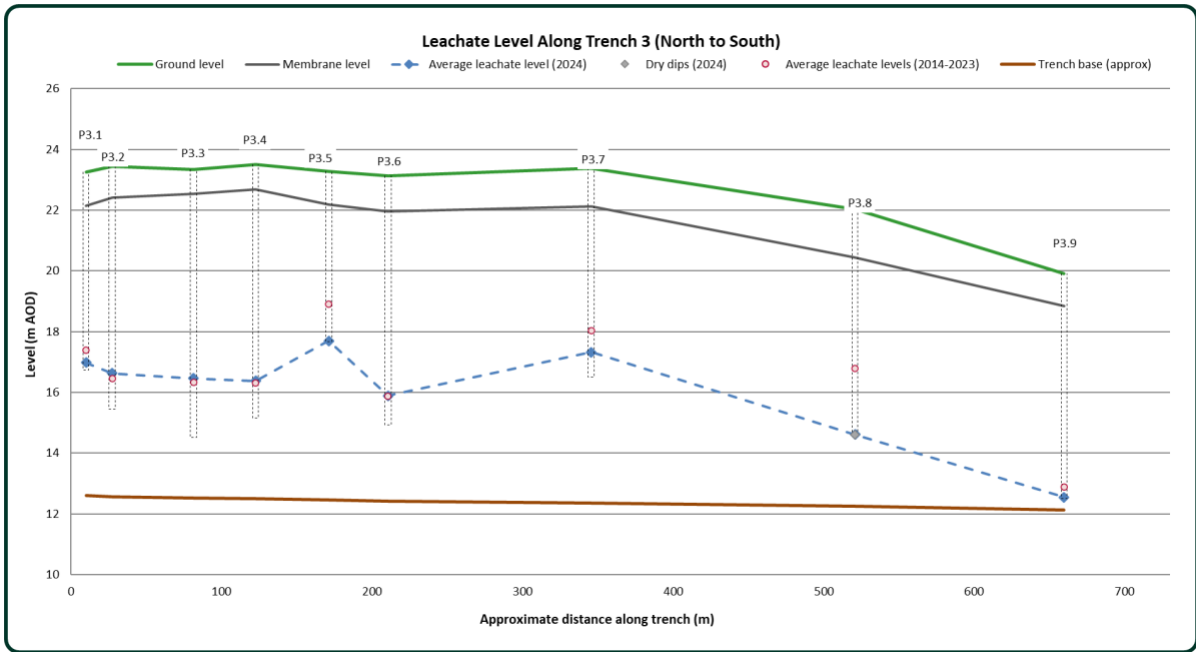


Figure 5.13: Leachate level along Trench 3

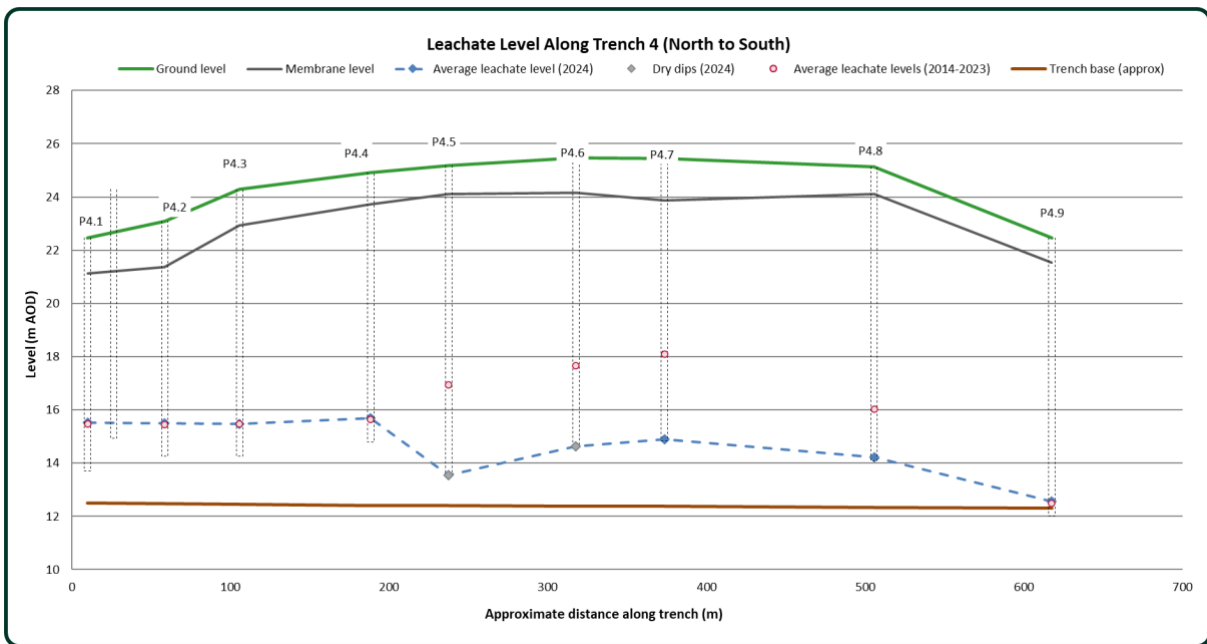


Figure 5.14: Leachate level along Trench 4

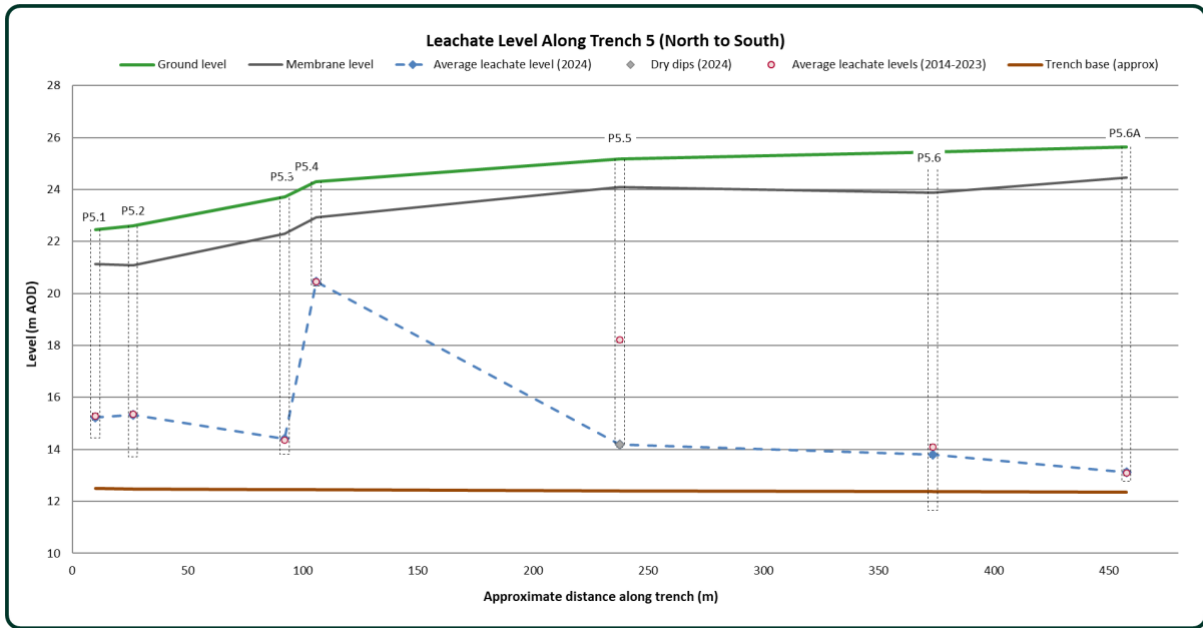


Figure 5.15: Leachate level along Trench 5

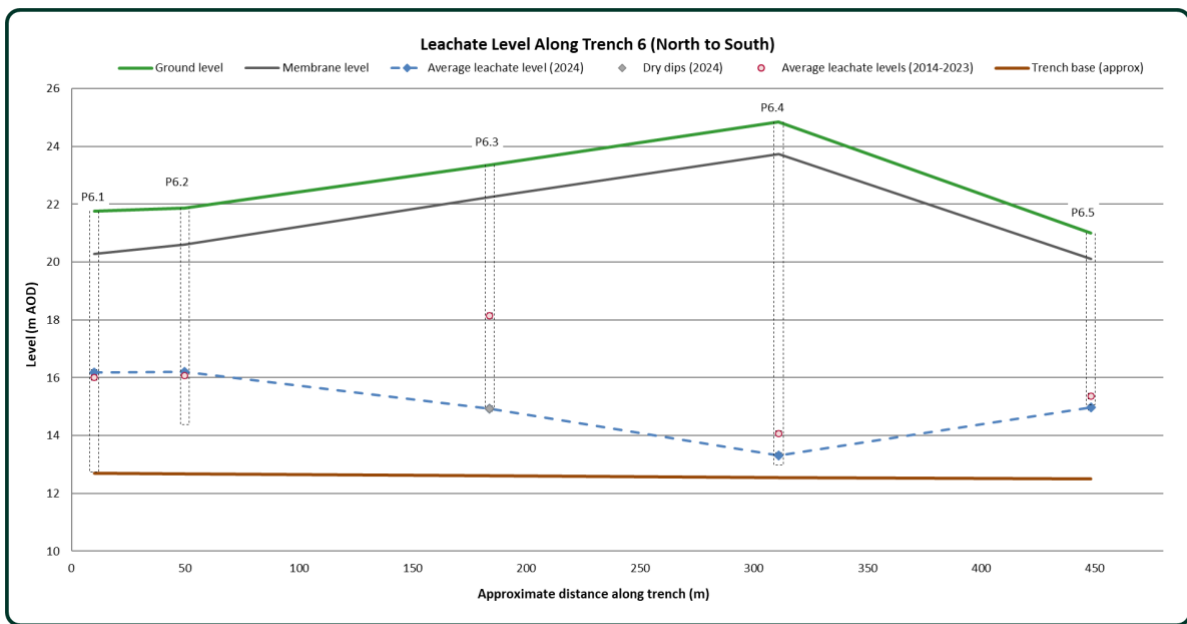


Figure 5.16: Leachate level along Trench 6

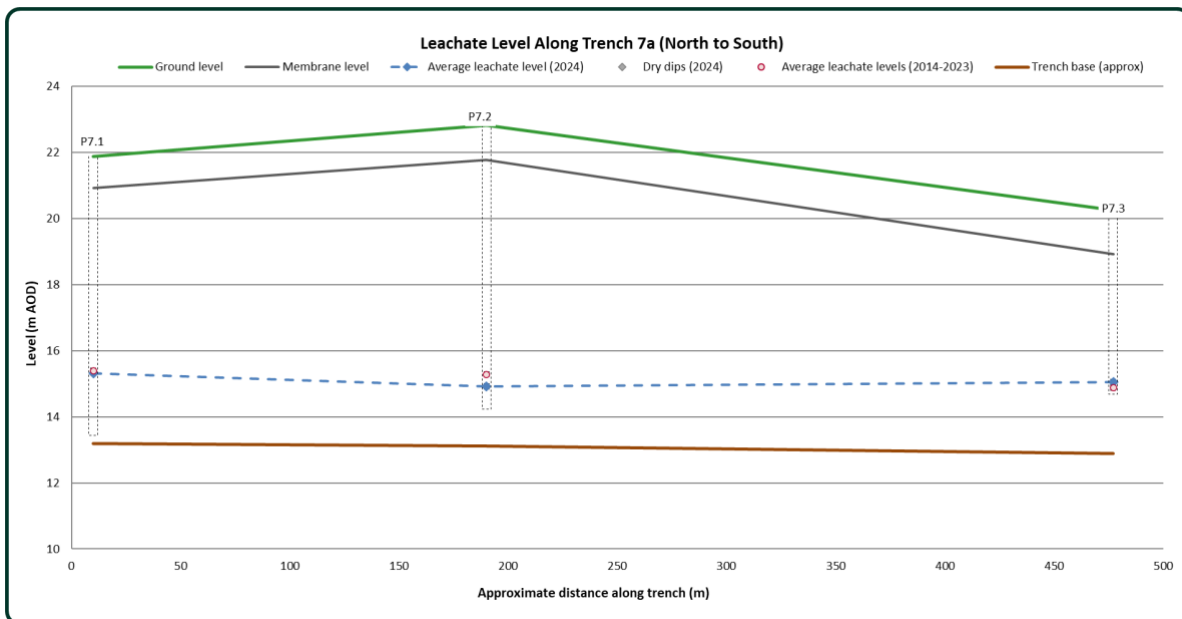


Figure 5.17: Leachate level along Trench 7a

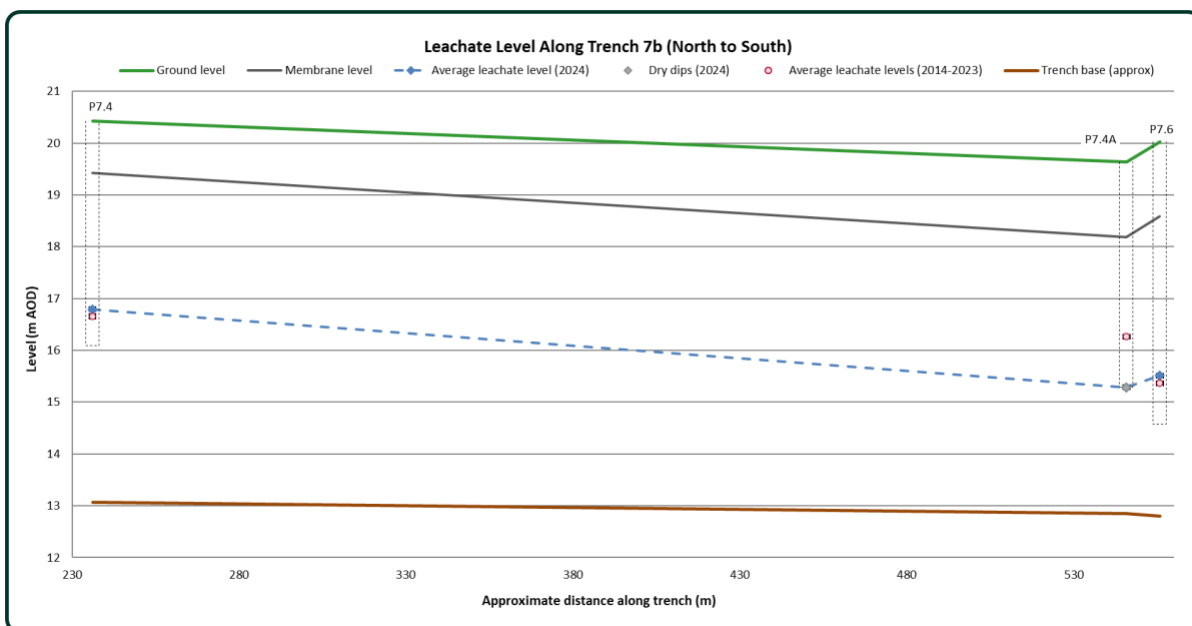


Figure 5.18: Leachate level along Trench 7b

Leachate quality

The majority of the hazardous and non-hazardous determinands in the bulk leachate samples in 2024/25 were found to be below the LoD. Determinands that were found to be above the LoD in 2024/25 are presented in Table 5.4. The results are consistent with those from previous years [44].

Of the determinands detected in 2024/25 above the LoD, six were found to be present in at least one sample at levels exceeding the LLWRAS. With the exception of arsenic and mercury, each of these six determinands is part of the non-radioactive suite that is currently analysed at the MHT.

Table 5.4: Bulk leachate quality results (above LoD) in 2024/25

Determinand	No. of analyses	No. > LoD	Max (mg l ⁻¹)	LLWRAS ^[1] (mg l ⁻¹)	No. of exceedances
Ammoniacal Nitrogen	3	3	0.73	n/a	-
Chloride	3	3	83	n/a	-
Aluminium	3	3	0.7	n/a	-
Arsenic	3	3	0.062	0.025	1
Boron	3	3	0.2	7	0
Cadmium	3	2	0.00056	0.0002	1
Cobalt	3	1	0.002	0.003	0
Copper	3	2	0.002	0.00376	0
Lead	3	3	0.01	0.0013	3
Mercury	3	1	0.00033	0.00007	1
Molybdenum	3	2	0.007	n/a	-
Nickel	3	3	0.17	0.0086	1
Sodium	3	3	47	n/a	-
Sulphate as SO ₄	3	3	65	n/a	0
Zinc	3	3	1.02	0.0068	3
Total TPH >C8-C40	3	2	0.02	n/a	-
Monobutyltin ^[2]	3	2	0.00024	0.016	0

[1] These standards are based on an EQS for salt water and therefore differ from those set out in Table 2.1.

[2] For monobutyltin, the units are mg Sn per litre.		
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Lead and zinc were consistently above the relevant LLWRAS for leachate discharge to salt water. The determinands exceeding the LLWRAS less frequently were arsenic, cadmium, mercury and nickel. These results are considered to be broadly consistent with leachate quality reported in the 2017 HRA [44], the 2011 ESC [119] and with results reported in a major review of hydrogeochemistry at the LLWR reported in 2010 [120]. There is thus no indication of significant changes to the non-radioactive source term.

Analysis of samples taken from MHT leachate for non-radiological contaminants is carried out on a monthly basis. The results of this analysis for the most recent reporting period (2024) are shown in Table 5.5. These results are broadly consistent with previous years.

Table 5.5: Summary of non-radiological contaminants in discharge samples (2025)

Determinand	Number of samples	Number > LoD ²⁷	Minimum (µg l ⁻¹)	Mean (µg l ⁻¹)	Maximum (µg l ⁻¹)
Ammonium as NH ₄	12	5	3.8	16.3	32.3
Cadmium	12	6	0.0	0.04	0.2
Chromium	12	12	0.7	1.1	1.8
Copper	4	4	1.8	2.3	3.1
Iron	12	12	316.0	998.0	3430.0
Lead	12	12	0.6	1.7	3.3
Nickel	12	12	2.1	3.6	8.5
Nitrate as NO ₃	12	12	3010.0	5446.7	9490.0
Nitrite	12	3	7.0	30.3	76.8
Phosphorus	12	12	32.9	50.0	81.0
Zinc	12	12	17.8	31.8	52.9

²⁷ LoDs were appropriate at the time of the issue of the relevant monitoring report and may differ from those in Table 2.1.

For the purposes of identifying unusual data, UDILs are calculated as described in Subsection 5.2.1. Separate UDILs are calculated for trench leachate, vault leachate and MHT leachate in order to allow assessment of results that are unusual for the relevant system.

The number of UDIL exceedances is low compared with the total number of measurements made. This provides reassurance that the current understanding of trench and vault leachates is accurate. The low number of DQF exceedances for the trench and vault leachates indicate a lack of significant changes within the leachate system.

Overall, the results of leachate monitoring are considered to be generally consistent over time. The trenches rather than the vaults are the predominant contributors to measured concentrations of non-radiological contaminants at the MHT. While some UDIL exceedances are expected, they are investigated to identify if a rising trend in contaminant concentrations is occurring at the relevant location. No adverse trends have been observed in either trench or vault leachate concentrations.

A risk assessment is carried out annually to assess the impact of discharges of non-radiological contaminants on receiving waters in accordance with Environment Agency guidance [121]. The assessment for releases to the marine environment for the most recent reporting period (2024) is documented in reference [122]. All of the non-radiological analytes were classified as insignificant. Discharges do not, therefore, present an unacceptable risk to the environment.

5.5.2 Groundwater Data Review

Groundwater levels

Groundwater level data are assessed in order to interpret changes in groundwater systems and establish their significance. Typically, groundwater elevation data are therefore considered in two different ways:

- changes to the overall system – indicated by changes in groundwater levels in multiple wells in a groundwater system;
- localised impact – indicated by groundwater level change in the environment around an individual well, which may indicate an effect of engineered structures or site operations on groundwater flow in that immediate area.

Further investigations are undertaken if significant changes are identified in either of the above. Diurnal (tidal), seasonal and annual fluctuations in elevation are apparent in the data collected, but the changes are not deemed to be significant, with groundwater elevation in the relative groundwater systems remaining consistent over the longer term.

We show the interpolated regional contour plot in Figure 5.19. The head in the Regional Groundwater generally has a weak horizontal gradient roughly perpendicular to the coast, in line with the hydrogeological conceptual model (Subsection 2.2.4).

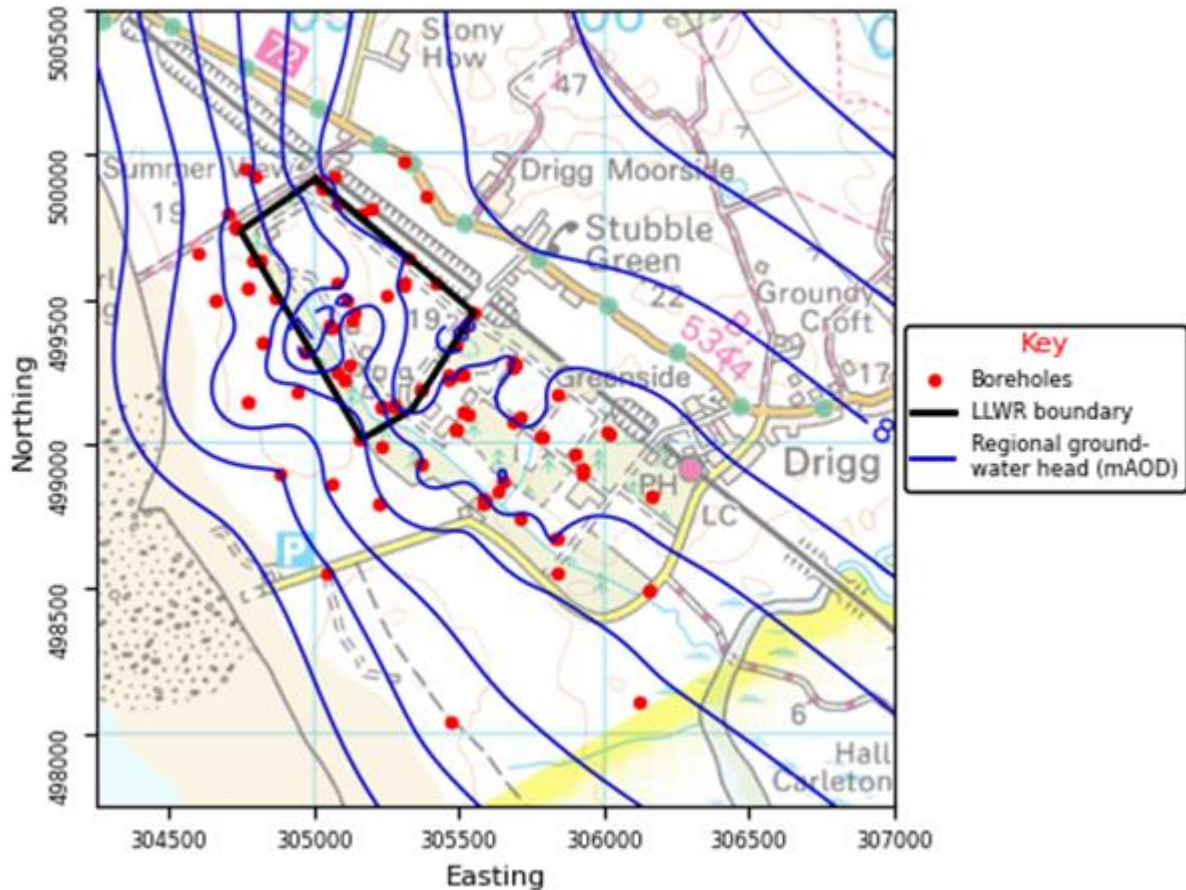


Figure 5.19: Interpolated contours of the groundwater heads in the regional groundwater around the LLWR site (with outliers removed). The contours are at 1 m intervals.

Figure 5.19 also shows an apparent ‘groundwater mound’ on the western boundary of the LLWR where the head consistently rises several metres above the value associated with the overall gradient. The most recent geological interpretation [51] and associated hydrogeological modelling [30] show slightly improved agreement between modelled and observed heads within the mound, but significant differences remain. While the origins of the mound are not fully understood at this point, it is not considered to have a significant impact either on the overall hydrogeological interpretation or on contaminant migration from the facility.

With the exception of the groundwater mound, measured groundwater heads are generally in line with groundwater flow modelling [30] providing confidence both in the conceptual hydrogeological interpretation and in the parameterisations and data used in the modelling.

The potential groundwater mound immediately to the west of the disposal area is currently being investigated further via geophysics.

Groundwater quality

Groundwater quality data for periods 2016 to 2020 and 2021 to 2025 are presented in Table 5.6. In the most recent period, concentrations of contaminants were either similar to, or less

than, the concentrations measured in 2016 to 2020. A longer-term analysis of the data shows that concentrations have fluctuated over time but with no significant trends observed. These fluctuations are attributed to natural variation.

While there are no historical groundwater quality data from the period before disposal operations began in 1959, the 2016 to 2020 and 2021 to 2025 groundwater quality data in Table 5.6 may be compared with baseline values derived during an extensive review of non-radiological water quality entering the site reported in 2026 [118], presented here in Table 5.3. Such comparison reveals that groundwater quality in the periods 2016 to 2020 and 2021 to 2025 is broadly consistent with that reported in the 2017 HRA [44]. This suggests no evolving risk posed to groundwaters due to release of non-radioactive contaminants from the site. It also suggests that there is no need to increase the frequency of the non-radiological aspect of the triennial monitoring programme.

As discussed in Subsection 5.2.3, MALs are the appropriate measure to compare groundwater monitoring data against. We present this comparison for the periods 2016 to 2020 and 2021 to 2025 in Table 5.7. These data indicate that there were MAL exceedances for ammoniacal nitrogen, arsenic, barium, cobalt, copper, cyanide, iron, lead, magnesium, nickel, phenol index, vanadium and zinc. Ammoniacal nitrogen, arsenic and barium were the contaminants with the greatest number of MAL exceedances. Ammoniacal nitrogen is present in the groundwater because of local agricultural practices. Figure 5.20 shows the large spatial distribution of each contaminant across and around the LLWR site which is indicative of a large natural variance in groundwater. There is no evidence that the exceedances occur because of wastes disposed of at the LLWR.

Table 5.6: Summary of non-radiological contaminants in groundwater (2016 to 2025)

Determinand	2016-2020				2021-2025			
	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)
Ammoniacal nitrogen	73	62	287	3420	147	119	337	3260
Arsenic	167	141	8.75	67.20	178	178	9.44	117.60
Barium	167	167	255	2830	178	178	298	3330
Beryllium	167	11	0.80	1.00	178	24	0.32	1.43
Boron	167	167	55	505	178	178	48	449
Cadmium	167	14	0.80	1.00	178	22	0.29	1.20
Chloride	73	73	60568	171000	121	121	55848	170700
Chromium	167	135	3.36	20.10	178	165	1.56	15.44
Cobalt	150	94	1.85	10.50	178	157	1.38	54.20
Copper	167	95	1.70	19.70	178	167	2.48	75.60
Cyanide	57	1	64	860	138	31	20	60.00

Determinand	2016-2020				2021-2025			
	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)
Fluoride	34	1	202	266	147	123	153	710
Iron	233	231	4089	75800	356	346	3580	237000
Lead	167	38	0.97	7.21	178	98	0.92	37.40
Magnesium	73	73	17588	45200	116	116	16291	48300
Mercury	167	21	0.17	1.40	178	36	0.21	1.50
Molybdenum	150	77	1.66	13.20	178	163	1.18	7.71
Nickel	167	159	5.38	58.40	178	178	4.34	67.30
Nitrate as NO ₃	111	46	1923	53000	121	67	1539	40800
Nitrite	111	8	109	409	121	22	40	280.00
Phenol	57	21	0.87	4.30				
Phenol index					138	43	54	150.00
Selenium	167	115	1.65	11.60	178	177	1.03	6.89

Determinand	2016-2020				2021-2025			
	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)
Vanadium	150	138	7	185	178	178	6	25
Zinc	167	158	21	633	178	176	22	644

Table 5.7: MAL exceedances 2016 to 2025

Determinand	MAL ($\mu\text{g l}^{-1}$)	2016-2020			2021-2025		
		Number of samples	Number > LoD	Percentage > MAL	Number of samples	Number > LoD	Percentage > MAL
Ammoniacal nitrogen	817.2	73	62	9.6	147	119	13.6
Arsenic	19	167	141	11.4	178	178	11.2
Barium	501.8	167	167	12	178	178	16.3
Beryllium	10	167	11	0	178	24	0
Boron	2000	167	167	0	178	178	0
Cadmium	1.77	167	14	0	178	22	0
Chloride	206800	73	73	0	121	121	0
Chromium	50	167	135	0	178	165	0
Cobalt	3	150	94	20	178	157	10.1
Copper	11.3	167	95	1.2	178	167	5.1

Determinand	MAL ($\mu\text{g l}^{-1}$)	2016-2020			2021-2025		
		Number of samples	Number > LoD	Percentage > MAL	Number of samples	Number > LoD	Percentage > MAL
Cyanide	50	57	1	1.8	138	31	0.7
Fluoride	1000	34	1	0	147	123	0
Iron	9770	233	231	11.2	356	346	9.8
Lead	4	167	38	2.4	178	98	3.4
Magnesium	23020	73	73	19.2	116	116	12.9
Mercury	1	167	21	0	178	36	0
Molybdenum	10.99	150	77	1.3	178	163	0
Nickel	10	167	159	9.6	178	178	5.6
Nitrate as NO ₃	50000	111	46	0.9	121	67	0
Nitrite	500	111	8	0	121	22	0
Phenol	7.7	57	21	0			
Phenol index					138	43	8.7

Determinand	MAL ($\mu\text{g l}^{-1}$)	2016-2020			2021-2025		
		Number of samples	Number > LoD	Percentage > MAL	Number of samples	Number > LoD	Percentage > MAL
Selenium	40	167	115	0	178	177	0
Vanadium	20	150	138	2.7	178	178	2.2
Zinc	42.49	167	158	9	178	176	10.7

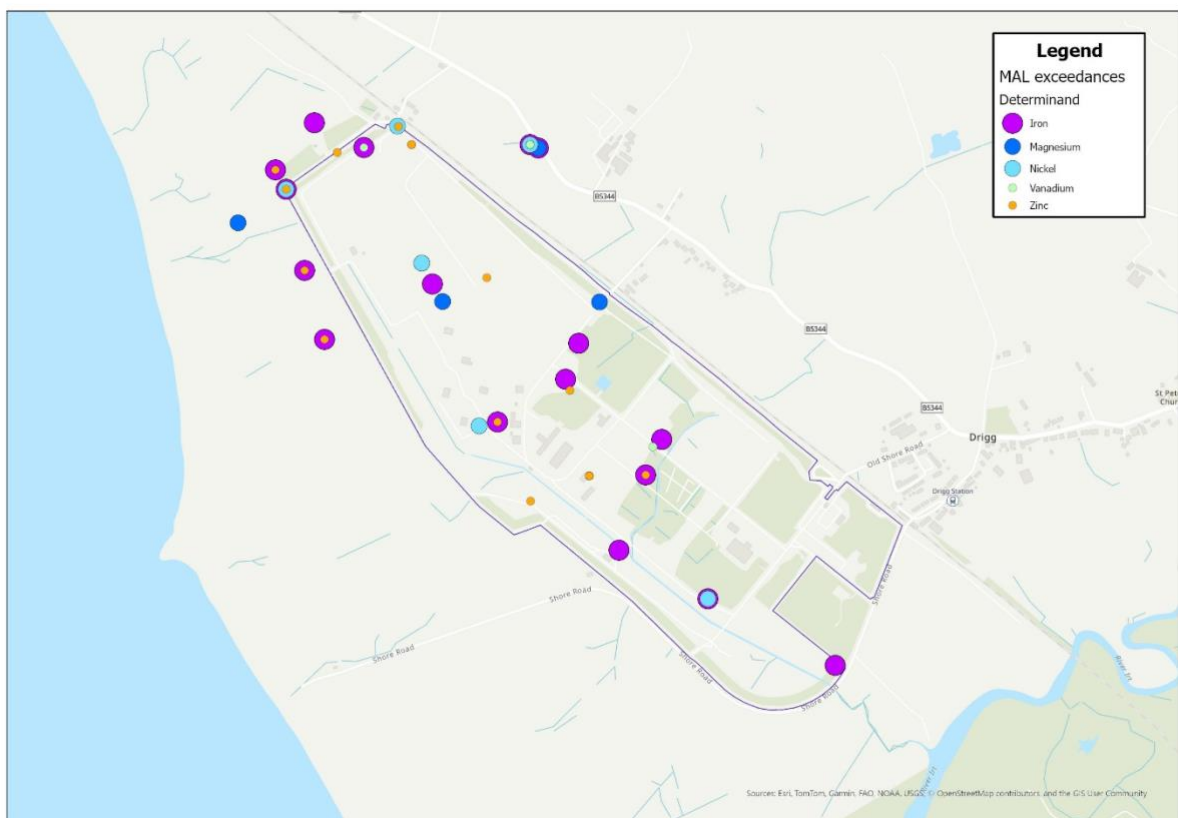
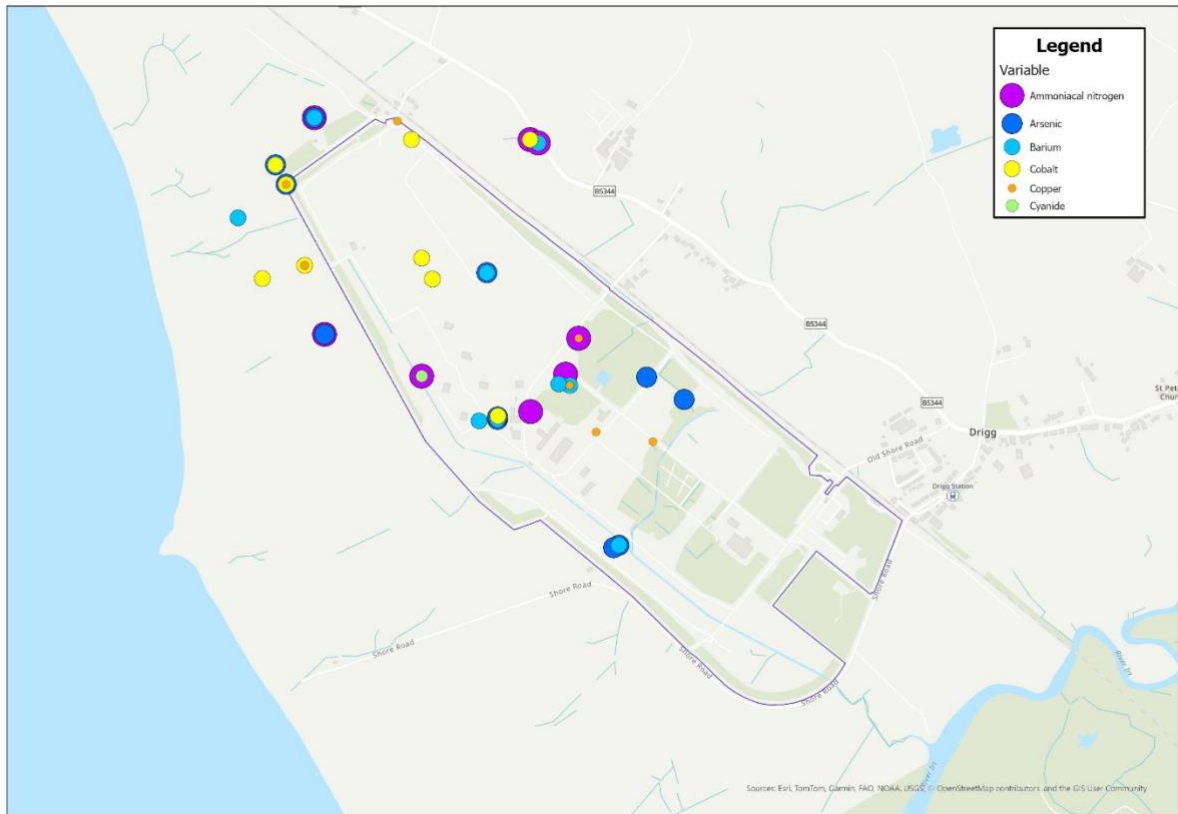


Figure 5.20: Spatial distribution of MAL exceedances

As discussed in Subsection 5.4.4, a series of field parameter measurements are made prior to sampling. These measurements are generally taken in every year of the monitoring cycle and are used to indicate potential changes in groundwater chemistry. The results do not indicate a significant change in groundwater chemistry resulting from operations at the LLWR. The vast majority of monitoring wells show a consistently near neutral pH.

The Regional Groundwater has been observed to contain higher concentrations of tritium than the equivalent Upper Groundwater [111]. This is in contrast with the non-radiological contaminants, which demonstrate no significant difference between the groundwater depth intervals. This suggests that potential sources of non-radiological contaminants have not impacted groundwater quality beneath the LLWR site.

Overall, non-radiological monitoring results show no significant changes in groundwater systems and are consistent with the conceptual models and ESC assumptions. Non-radiological groundwater quality remains consistent with results observed in previous years.

On the basis of evidence presented here, we conclude that discharges to groundwater from the LLWR pose no significant risk to the environment.

5.5.3 Surface Water Data Review

Surface water levels and flows

Data collected from the surface water monitoring points discussed in Subsection 5.4.5 have shown a general trend of falling discharge volumes from March to September with higher discharge volumes during the winter months. However, in 2024/25 above average rainfall was recorded in August, which resulted in higher discharge volumes being recorded in the late summer months of 2024. Surface water levels are considered to be generally consistent with the site conceptual model and assumptions embedded within the ESC.

Surface water balance modelling provides a means to assess the performance of the interim trench cap. Such assessments are carried out and reported annually in the '*Disposal Area Performance*' report [85]. The effectiveness of the cap is estimated by comparing the run-off from the cap into the cap perimeter drains with the HER. If the cap is 100% efficient then all HER will become run-off. Efficiency values greater than 100% can arise due to uncertainties in water balance modelling, especially when:

- run-off measurements are overestimated due to temporary drainage enhancements;
- HER is underestimated, particularly during dry spells or when rainfall data are interpolated;
- probe hole drainage or surface water diversion temporarily increases measured run-off beyond expected HER

It is known that the current interim trench cap is suboptimal in performance and therefore a programme of work to replace the interim cap commenced in 2025. The suboptimal performance was primarily identified from the cap efficiency calculations, which indicated significant variation between years, as illustrated by Figure 5.21. This was initially attributed

to potential inflow around probe holes [123] but intrusive investigations demonstrated that the interim membrane had been damaged during the construction of the interim cap

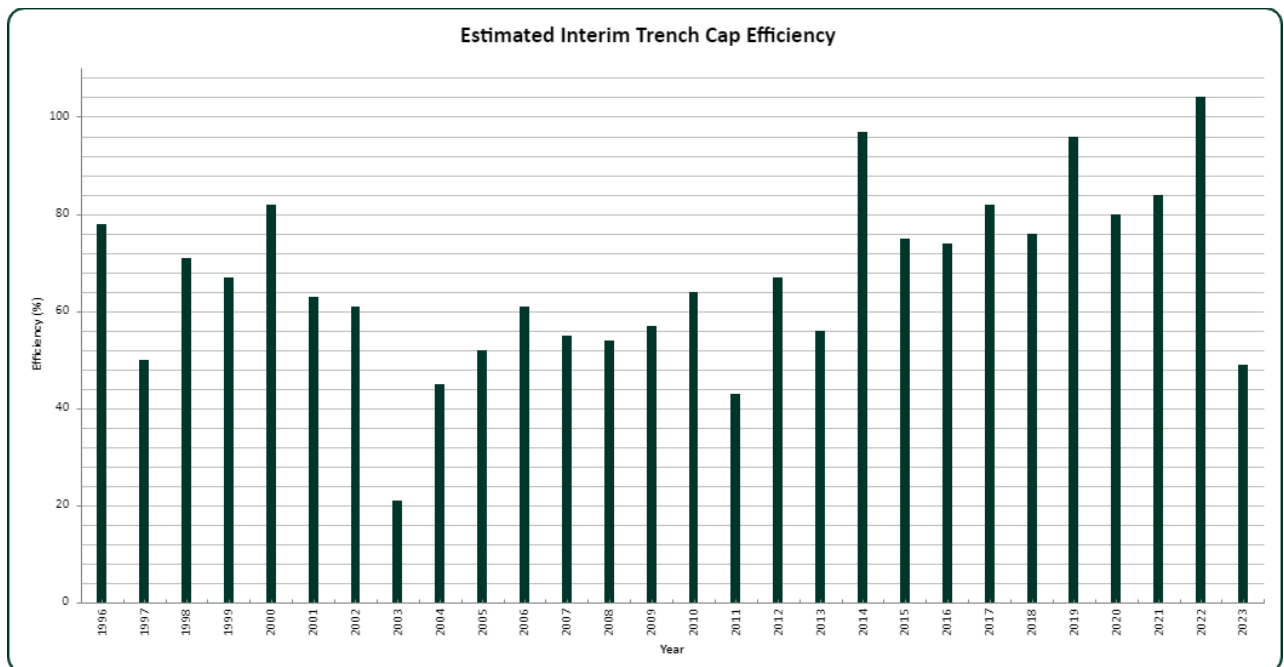


Figure 5.21: Estimated interim trench cap efficiency [85]

Stream water quality

Stream water quality samples are collected at various locations as described in Subsection 5.4.5. In the third year of the triennial cycle, samples are analysed for an enhanced suite of non-radiological contaminants. 2025 was the most recent such enhanced sampling year, the previous being 2022. Metals and non-metals were monitored on an annual basis between 2019 and 2023 in all stream water samples. Summarised analytical results for 2016-2020 and 2021-2025 are presented in Table 5.8.

Stream water samples are assessed against the stream water baseline. Comparison with Table 5.2 indicates that stream water quality in the periods 2016 to 2020 and 2021 to 2025 is broadly consistent with the baseline stream water quality values derived. This suggests no evolving risk posed to stream waters due to release of non-radioactive contaminants from the site. It also supports the view that there is no need to increase the frequency of the non-radiological aspect of the triennial monitoring programme.

There were six (1.9%) stream water baseline exceedances for non-radiological determinands in the period 2016 to 2020 and sixteen (3.2%) in 2021-2025. Six of the exceedances observed in the 2021-2025 period are for sulphate as SO_4 , which was not included in the analysis suite between 2016-2020. Examination of trended data did not reveal any adverse trends at the relevant locations.

No contaminants consistently exceeded the stream water baseline concentrations and the concentrations observed do not indicate an impact of site operations on the quality of on-site streams.

Table 5.8: Summary of non-radiological contaminants in stream water between 2016 and 2025

Determinand	Baseline concentration ($\mu\text{g l}^{-1}$)	2016-2020				2021-2025			
		Number of samples	Number > LoD	Max. ($\mu\text{g l}^{-1}$)	No. > baseline	Number of samples	Number > LoD	Max. ($\mu\text{g l}^{-1}$)	No. > baseline
Arsenic	8.50	19	15	7.39	0	26	26	8.29	0
Barium	191	19	19	88	0	27	27	127.5	0
Beryllium	1	19	0	1	0	26	1	0.75	0
Boron	39	19	19	66.4	1	26	26	65.7	1
Cadmium	1	19	1	1	0	26	3	1.3	0
Chromium	9.55	19	16	2.15	0	26	26	2.39	0
Cobalt	6.69	13	6	1	0	26	24	0.75	0
Copper	14.92	19	17	12.3	0	26	26	26	1
Cyanide	20	11	1	50	0	17	6	20	0
Iron	13584	13	13	2820	0	26	26	25700	2
Lead	26.80	19	8	3.52	0	26	25	4.18	0

Determinand	Baseline concentration ($\mu\text{g l}^{-1}$)	2016-2020				2021-2025			
		Number of samples	Number > LoD	Max. ($\mu\text{g l}^{-1}$)	No. > baseline	Number of samples	Number > LoD	Max. ($\mu\text{g l}^{-1}$)	No. > baseline
Mercury	1.88	19	1	0.5	0	26	3	0.26	0
Molybdenum	0.87	13	5	1	0	26	21	0.97	1
Nickel	15.05	19	18	3.63	0	26	26	3.6	0
Nitrate as NO_3	23805	10	10	22000	0	16	16	26300	1
Nitrite	264	10	6	637	2	16	12	428	2
Phenol	0.52	11	2	5	2	-	-	-	0
Phenol Index	50	-	-	-	0	16	7	60	2
Selenium	4	19	8	4.35	1	26	26	1.22	0
Sulphate as SO_4	23495	-	-	-	0	16	16	29400	6
Vanadium	16.94	13	13	6.84	0	26	26	11.29	0
Zinc	107.775	19	19	11	0	26	26	85.7	0

Drainage water quality

Drainage water quality samples are collected at various locations as described in Subsection 5.4.5. Non-radiological monitoring of drainage water samples is limited to monitoring of on-site drainage. In the third year of the triennial cycle, samples are analysed for an enhanced suite of non-radiological contaminants. 2025 was the most recent such enhanced sampling year, the previous being 2022. Summarised analytical results for 2016 to 2020 and 2021 to 2025 are presented in Table 5.9.

Drainage water samples are assessed against relevant LLWRAS. Comparison with Table 5.2 indicates that drainage water quality in the periods 2016 to 2020 and 2021 to 2025 is broadly consistent with LLWRAS. This suggests no evolving risk posed to drainage waters due to release of non-radioactive contaminants from the site. It also supports the view that there is no need to increase the frequency of the non-radiological aspect of the triennial monitoring programme.

There were 41 LLWRAS exceedances for non-radiological determinands in the period 2016 to 2020 and 69 in the period 2021 to 2025. These are identified in Table 5.10. Examination of trended data did not reveal any adverse trends at the relevant locations.

Concentrations of the following contaminants consistently exceed the LLWRAS.

- Arsenic concentrations were elevated above the LLWRAS for almost all the drainage water samples tested between 2016 and 2025. The widespread nature of the elevated concentrations represents an elevated arsenic background rather than an impact of site operations. The concentration range in drainage water samples analysed for arsenic between 2021 and 2025 is 0.58 to 5.62 $\mu\text{g l}^{-1}$. The surface water baseline for stream water entering the site in the Drigg Stream is 8.5 $\mu\text{g l}^{-1}$.
- Copper concentrations were elevated above the LLWRAS for almost all the drainage water samples tested between 2016 and 2025. The widespread nature of the elevated concentrations represents an elevated copper background rather than an impact of site operations. The concentration range in drainage water samples analysed for copper between 2021 and 2025 is 0.9 to 2.34 $\mu\text{g l}^{-1}$. The surface water baseline for stream water entering the site in the Drigg Stream is 14.9 $\mu\text{g l}^{-1}$. One copper result from a sample taken from P29 swale has been removed from the dataset due to its erroneously high value (67.5 $\mu\text{g l}^{-1}$). Whilst the sample analysis was checked, and shown to be correct, it is believed that there has been some cross contamination of the sample during sampling.

The following contaminants have fewer exceedances of the relevant LLWRAS.

- Iron concentrations were elevated about the LLWRAS in several samples taken from drainage within the historical ROF manufacturing area. It is unknown whether the levels of iron observed are due to natural processes in the area or historical ROF activities. The proportion of iron exceedances in the period 2021 to 2025 has decreased when compared against the exceedances in the period 2016 to 2020.

- Lead concentrations consistently exceed the LLWRAS in samples taken from OF2. OF2 is a drainage line that drains part of the ROF TNT manufacturing area. The historical use of sacrificial lead in troughs, used to transport acids around the area, has caused ground contamination in the area. It is possible that the slightly elevated lead levels observed are due to migration of this contamination; however, it may also be due to an elevated background in the area. The concentrations observed are lower than the stream water baseline, calculated from the concentration observed in Drigg Stream as it enters site. The concentrations observed at OF2 are usually in the range 1.3 to 2.83 $\mu\text{g l}^{-1}$ with occasional peaks in concentration observed whilst the stream water baseline is 26.8 $\mu\text{g l}^{-1}$. The levels of lead observed in samples taken for the OF1 drainage system are lower than those taken from OF2 but could also be impacted in the same ways.
- Mercury concentrations exceed the LLWRAS in a small number of samples taken from the eastern trench cap catchment area, OF1 and OF2. We note that the LoD for mercury is above the current LLWRAS and an order of magnitude above the LLWRAS prior to 2024. As a result, any positive detects of mercury lead to an LLWRAS exceedance. The OF2 catchment is not influenced by disposals. This, together with the widespread exceedances observed across site, suggests that the elevated results are not due to repository operations, although the source of the mercury is not known. The positive detects observed range from 0.011 to 0.122 $\mu\text{g l}^{-1}$ which is significantly below the 1.88 $\mu\text{g l}^{-1}$ stream water baseline concentration. We note that the increase in the number of mercury LLWRAS exceedances between the 2016 to 2020 and 2021 to 2025 periods is due to a reduction in the LoD in 2021-2025 rather than an increase in concentrations observed.
- Nickel concentrations exceed the LLWRAS in a small number of samples taken from the historical ROF TNT manufacturing area (OF2 and P29 swale). The levels observed range from 4.3 to 5.3 $\mu\text{g l}^{-1}$ exceeding the LLWRAS of 4 $\mu\text{g l}^{-1}$. It is possible that the slightly elevated nickel concentrations observed are due to migration of ground contamination from the previous industrial land use. It may, however, also be due to an elevated background in the area. We note that the concentrations observed are lower than the stream water baseline of 15.1 $\mu\text{g l}^{-1}$, calculated from the concentration observed in Drigg Stream as it enters site.
- The vanadium concentration exceeded the LLWRAS in one sample taken from the historical ROF TNT manufacturing area (P29 swale). The concentration observed was 20.4 $\mu\text{g l}^{-1}$ compared to an LLWRAS of 20 $\mu\text{g l}^{-1}$. The range for all other surface water vanadium analysis was 0.53 to 11 $\mu\text{g l}^{-1}$.
- The zinc concentration exceeded the LLWRAS in approximately half of the samples analysed. All the exceedances observed drain part of the historical ROF TNT manufacturing area. No zinc contamination has been observed in soils in the area and it is thought that zinc drainage water LLWRAS exceedances may be natural.

As a general observation, the drainage waters around the LLWR can be influenced by significant periods of dry or wet weather and the sample collection method that is used.

BOD and TSS analysis are routinely undertaken on each of the river water and surface water samples collected on a quarterly or annual basis. Elevated TSS results were recorded at a number of sample points throughout 2024/25, but this has principally occurred during periods of low rainfall when the levels at the respective water bodies have been low, and sediment has been mobilised during sampling. Occasional spikes in BOD have been recorded, but no sustained elevated concentrations have been recorded that would indicate a change in river or surface water chemistry on and surrounding the site and the results observed remain consistent with previous years.

A series of field parameters (physical and chemical measurements) are regularly determined during sampling of the surface water. During the most recent reporting period, these parameters included: pH, specific conductance and temperature. The field parameter readings provide data to identify changes in the overall surface water chemistry.

The field parameters did not identify any changes in the overall river water or surface water chemistry [114].

Overall, we consider that non-radiological drainage water quality is consistent with the conceptual model and ESC assumptions. There is also consistency with non-radiological results recorded in previous years. There is no evidence of impact on drainage water from the non-radiological component of site operations at the LLWR.

Table 5.9: Summary of non-radiological contaminants in drainage water (2016 to 2025)

Determinand	2016-2020				2021-2025			
	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)
Ammoniacal nitrogen	2	2	181.50	197.00	1	1	585.00	585.00
Arsenic	12	10	3.63	7.63	20	19	2.43	5.62
Barium	12	12	89.14	200.00	20	20	110.71	224.00
Beryllium	12	0	0.75	1.00	20	1	0.26	0.53
Boron	12	12	24.37	32.30	20	20	32.63	78.10
Cadmium	12	0	0.74	1.00	20	0	0.36	1.30
Chromium	12	12	1.29	2.68	20	20	1.01	1.95
Cobalt	10	4	0.68	1.00	20	16	0.37	0.68
Copper	12	12	1.95	3.27	20	20	4.98	67.50
Cyanide	4	0	29.50	50.00	4	1	27.50	>50.00

Determinand	2016-2020				2021-2025			
	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)
Iron	10	10	1057.20	1940.00	20	20	865.75	2650.00
Lead	12	8	2.85	14.62	20	16	1.66	19.00
Mercury	12	0	0.14	0.32	20	7	0.10	0.26
Molybdenum	10	10	1.34	2.53	20	20	1.42	3.88
Nickel	12	12	2.96	4.86	20	20	2.98	5.32
Nitrate as NO ₃	4	4	2757.50	4140.00	5	5	2238.00	2650.00
Nitrite	4	0	100.00	100.00	5	0	18.80	37.00
Phenol	4	2	2.80	5.00				
Selenium	12	8	1.02	1.39	20	20	0.80	1.24
Sulphide	2	0	50.00	50.00	5	3	27.50	50.00
Total organic carbon	4	4	6775.00	8000.00	4	4	5477.50	6110.00

Determinand	2016-2020				2021-2025			
	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)	Number of samples	Number > LoD	Mean ($\mu\text{g l}^{-1}$)	Max. ($\mu\text{g l}^{-1}$)
Vanadium	10	10	3.20	7.12	20	20	6.50	20.40
Zinc	12	12	16.26	43.20	20	20	17.60	77.00

Notes: LoD used as a basis for this table were appropriate at the time of issue of the relevant monitoring report and may differ from those in Table 2.1.

It is cautiously assumed that Cr is present in surface waters as Cr(VI)

Table 5.10: Summary of non-radiological LLWRAS exceedances in drainage water (2016 to 2025)

Determinand	LLWRAS ($\mu\text{g l}^{-1}$)	2016-2020			2021-2025		
		Number of samples	Number > LoD	Number > LLWRAS	Number of samples	Number > LoD	Number > LLWRAS
Arsenic	1.00	12	10	10	20	19	18
Barium	1300.00	12	12	0	20	20	0
Beryllium	4.00	12	0	0	20	1	0
Boron	2000.00	12	12	0	20	20	0
Cadmium	5.00	12	0	0	20	0	0
Chromium	50.00	12	12	0	20	20	0
Cobalt	3.00	10	4	0	20	16	0
Copper	1.00	12	12	12	20	20	19
Cyanide	1.00	4	0	0	4	0	0
Iron	1000.00	10	10	5	20	20	7
Lead	1.00	12	8	6	20	16	6

Determinand	LLWRAS ($\mu\text{g l}^{-1}$)	2016-2020			2021-2025		
		Number of samples	Number > LoD	Number > LLWRAS	Number of samples	Number > LoD	Number > LLWRAS
Mercury	0.01	12	0	0	20	7	7
Nickel	4.00	12	12	2	20	20	3
Nitrate as NO ₃	50000.00	4	4	0	5	5	0
Nitrite	500.00	4	0	0	5	0	0
Phenol	7.70	4	2	0			
Selenium	10.00	12	8	0	20	20	0
Vanadium	20.00	10	10	0	20	20	1
Zinc	10.90	12	12	6	20	20	8

Notes: (a) the reported LoD for cyanide is $20 \mu\text{g l}^{-1}$, which exceeds the LLWRAS.

LoDs used as a basis for this table were appropriate at the time of issue of the relevant monitoring report and may differ from those in Table 2.1.

5.5.4 Summary of Monitoring Data and Implications

The review of monitoring data presented in the above subsections shows that the majority of non-radiological contaminant concentrations are consistent with natural background levels. There is no evidence that activities at the LLWR have resulted in significant non-radiological contamination of groundwater beneath the site or stream and surface water passing through the site

A review of non-radiological monitoring data was reported in 2010 [120], which concluded that there was limited evidence of contamination of groundwater, surface water and leachate as a result of past or current activities at the site. Monitoring results from recent years are broadly consistent with this review and with results reviewed as part of the 2011 ESC and the 2017 HRA. This provides evidence that:

- there has been no significant change to the non-radiological source term;
- the repository continues to have no impact on non-radiological groundwater quality in and around the repository site;
- there is no evolving risk to the environment resulting from disposal operations at the LLWR.

MAL exceedances in groundwater are relatively rare, occurring in 8% of all non-radiological samples analysed in both of the 2016 to 2020 and 2021 to 2025 periods. Ammoniacal nitrogen levels are consistently higher than the relevant MAL but derive from off-site agricultural practices. Over one third of iron concentrations exceed the relevant MAL. Iron exceeds LLWRAS in groundwater samples taken from a range of depths, both up-gradient and down-gradient of the LLWR, and the elevated concentrations are therefore not considered to derive from the waste. More generally, the distribution of the LLWRAS exceedances for non-radiological determinands does not indicate a clear zone of influence resulting from release of contaminants from the disposal area.

A small percentage of stream water samples exceeded the calculated river water baseline for non-radiological determinands in the period 2016 to 2025. Examination of trended data did not reveal any adverse trends at the relevant locations. No contaminants consistently exceeded the river water baseline concentrations and the concentrations observed do not indicate an impact of site operations on the quality of on-site streams.

LLWRAS exceedances occurred in around 22% of all non-radiological drainage water analyses in both of the 2016 to 2020 and 2021 to 2025 periods. The majority of these exceedances occurred because of the elevated arsenic and copper background. Removing the arsenic and copper results from the dataset results in an overall decrease in exceedances to 10% for both time periods. It is possible that the slightly elevated nickel and lead concentrations observed in surface waters could be due to migration of ground

contamination from the previous industrial land use; however, it may also be caused by an elevated background in the local area.

There is no evidence of significant LLWR-derived contamination of drainage waters.

6 Conclusions

Throughout this report, we have provided evidence to support the conclusion that non-radiological contaminants present in the wastes at the LLWR do not pose an unacceptable environmental risk to groundwater or surface water. We summarise the evidence and arguments supporting regulatory compliance in this regard in Subsection 6.1. Then in Subsection 6.2 we consider future related work, including formal review of the HRA presented in this report, ongoing work to provide reassurance that the site is evolving as expected and in a manner that does not pose an increasing risk to the environment, and studies aimed at better characterising both the waste and the hydrogeology around the site.

6.1 Regulatory Compliance

The LLWR is regulated by the Environment Agency under Schedule 23 of EPR16. Guidance on the specific regulatory requirements relevant to near-surface disposal of LLW is set out in the GRA. Requirement R10 of the GRA relates specifically to protection against non-radiological hazards and requires that the LLWR ‘provides adequate protection’ against such hazards.

The GRA provides further guidance on how this adequate protection may be demonstrated, noting that ‘nationally acceptable standards’ for disposing of hazardous waste ‘need not necessarily be applied’ in the case of waste that presents both a radiological and a non-radiological hazard. Nevertheless, there is a requirement that the level of protection be ‘no less stringent’ than if such standards were applied.

The relevant standards are the Water Framework Directive and the Groundwater Daughter Directive, as implemented through WER 2017 and Schedules 22 and 23 of EPR 2016. WER 2017 places a prohibition on direct discharges of pollutants into groundwater, while EPR 2016 places requirements to:

- prevent the input of hazardous substances to groundwater;
- limit the input of non-hazardous pollutants to groundwater

Supplementary guidance to the GRA related to implementation of the Groundwater Directive accepts that the means of complying with groundwater activity regulations for non-radiological hazards may be ‘proportionate’ to the non-radiological hazard presented by the waste.

In the following subsections, we summarise evidence presented throughout this report and present arguments to support our case for compliance with Requirement R10:

- Subsection 6.1.1 considers aspects relating to the design of the LLWR disposal system;

- Subsection 6.1.2 considers results of the quantitative HRA;
- Subsection 6.1.3 we consider aspects relating to the environmental monitoring programme;
- Subsection 6.1.4 summarises how we comply with regulatory requirements with respect to groundwater protection.

6.1.1 Design of the Disposal System

In the case of vault disposals, a key element of our argument for compliance appeals to guidance in Paragraph 6.4.2 of the GRA that an acceptable level of environmental protection:

‘could be achieved by ensuring that materials posing non-radiological hazards are contained within the facility over timescales at least as long as those provided for by the barriers at a site for disposal of non-radiological hazardous wastes.’

We presented evidence and arguments in Subsection 3.6 that the highly engineered vault disposal system will ensure that the vault waste is contained over longer timescales than would be expected for landfill disposal. We note that all disposals subsequent to 1995 have been and will continue to be to highly engineered vaults.

As explained in Section 3, the following are key components of our engineered design that have been identified as part of an ongoing process of design optimisation:

- a single domed final cap consistent with best practice to reduce infiltration;
- a cut-off wall to reduce lateral inflows into, and outflows from, the facility and which will also help to direct discharges to the underlying geology, rather than the near-surface environment;
- disposal of the wastes in steel containers, which reduces releases at early times;
- the use of cementitious grout, which provides a chemical barrier to the release of contaminants and provides a substrate for sorption, as well as reducing voidage in the waste stacks;
- removal of leachate from the vaults during the PoA to prevent the release of contaminants to the geosphere;
- design of a low sidewall in Vault 9 and subsequent vaults to ensure that unsaturated conditions continue to be promoted after the end of the PoA.

6.1.2 Quantitative HRA

We have undertaken a quantitative HRA, as presented in Section 4, to assess the future impacts of non-radiological (trench and vault) disposals. We consider this HRA to be no less robust than an equivalent HRA required for a hazardous waste landfill.

We assessed environmental impacts resulting from the whole waste inventory. We used an approach in which the source term was derived mechanistically from the disposed inventory, approximating the effects of various physical and chemical processes in the near field.

In our reference assessment calculation, including known inventories for both the trenches and the vaults, we found no future LLWRAS exceedances for non-radiological hazardous substances, although there were past exceedances (by a factor of 1.1) relating to vinyl chloride disposals in the trenches. Past exceedances were also found for benzene if we used a long degradation half-life.

Vinyl chloride is assumed to arise from the degradation of halogenated plastics and rubbers. It has been detected in trench leachate [6]. Our assessment is cautious as the vinyl chloride molecules are modelled as being immediately available for dissolution in infiltrating water. Our treatment of benzene is similarly cautious: we assume that a large range of organic materials in the inventory are comprised entirely of benzene which is immediately available for dissolution in infiltrating water. The exceedances are likely to be artefacts of these cautions, rather than indicating that exceedances occurred in reality.

The duration of the vinyl chloride and benzene exceedances is limited. This is because these contaminants will be subject to biodegradation and also because our optimised site development plan involves the installation of an engineered final cap which will reduce future releases.

In the reference case, calculated concentrations of non-hazardous pollutants were many orders of magnitude lower than the relevant assessment criteria for the whole of the assessment time period.

For this case, exceedances would only occur if we used a half-life of 0.8 years for phenol, which would result in very slight historical exceedances (less than a factor of 2) between 1982 and 1992. These exceedances derive entirely from the trenches and pre-date installation of the interim trench cap and final cap. The exceedances do not therefore provide any insight into final cap performance. Our treatment of phenol is cautious as we assume that organic ion exchange resins are comprised entirely of phenol which is immediately available for dissolution and transport in infiltrating water.

Our assessment calculations indicate that, even if key barriers perform significantly less well than anticipated, there is no significant risk of hazardous substances or non-hazardous pollutants entering the environment as a result of disposals at the LLWR.

Similarly, our radiological assessment calculations demonstrate that we will prevent the input of radionuclides to groundwater during, and after, the PoA.

We have also investigated the potential for contaminants in the grout used in vault disposals to present an environmental risk. All calculated concentrations are several orders of magnitude lower than the relevant assessment criteria.

Overall, calculated concentrations are significantly lower than those calculated in the 2011 ESC and in the 2017 HRA. This is predominantly because we have a revised understanding of cap performance [13] and we expect a much better level of performance than we did in the previous HRA and in the 2011 ESC.

6.1.3 Monitoring

In addition to Requirement R10, relating to overall protection against non-radiological hazards, the GRA also contains a specific requirement (Requirement R14) relating to monitoring. In particular, Paragraph 6.4.34 advises that:

'... during the construction stage and the period of authorisation, the developer/operator will need to monitor non-radiological parameters to confirm understanding of the effects that construction, operation and closure of the facility have on the characteristics of the site.'

We presented evidence and arguments in Section 5 that demonstrate compliance with this requirement. Our extensive environmental monitoring programme includes regular monitoring of leachate, groundwater and surface water for potential non-radiological contaminants.

Leachate monitoring provides data on the hydrology of the trenches and leachate volumes from both trenches and vaults, allowing assessment of the integrity of the interim cap. Monitoring of leachate quality tracks composition of the source term, identifying any changes and ultimately informing the non-radiological analysis suite across all three water systems. Authorised discharges to the environment, including the Irish Sea, are recorded.

Non-radiological groundwater monitoring is more spatially extensive than the equivalent monitoring required for a hazardous waste landfill. Groundwater levels are used to assess the performance of site engineering and confirm or facilitate improvement of hydrogeological understanding. Groundwater quality monitoring identifies any potential issues relating to contaminant emissions from the LLWR and tracks potential long-term variations in groundwater chemistry. The regular non-radiological groundwater monitoring programme can be supplemented as deemed appropriate by more targeted investigations based on actual groundwater pathways identified from the extent of migrating tritium.

Surface water monitoring enables assessment of engineered barrier performance and facilitates understanding of site hydrology. Any potential repository-derived contamination of surface waters or other changes in surface water chemistry can also be identified.

Monitoring results are analysed using well-defined semi-automated procedures. Upon input into the database, results are automatically analysed to identify 'unusual' data. Data are additionally trended on a quarterly basis for visual identification of potential gradual adverse changes. Together, these analyses constitute an 'early warning system', similar to control levels for landfill developments. Where unusual data or adverse trends are identified, additional investigations are undertaken. Finally, observed concentrations are compared against assessment standards (LLWRASs and MALs) to assess whether they conform to applicable water quality standards (EQS or DWD). Where LLWRASs or MALs are breached, additional comparison is performed against background concentrations to ascertain whether water quality is being adversely affected by the LLWR.

The review of monitoring results presented in Subsection 5.5 demonstrates that the LLWR is not having an adverse impact either on groundwater or on surface water. Most substances are present in groundwater, surface water and leachate at concentrations below their LLWRAS. Where LLWRAS standards are breached, this is generally due to elevated background concentrations relative to the LLWRAS or MAL or off-site activities. Groundwater and surface water quality remain consistent both with levels quoted in the 2017 HRA, the 2011 ESC and with the findings of an independent review of the hydrogeochemistry of non-radioactive contaminants reported in 2010. This found that groundwater contamination is not observed to be localised around the LLWR.

Where the concentrations of non-radiological contaminants in leachate exceed LLWRAS, these concentrations are broadly consistent with the groundwater baseline. It is therefore considered unlikely that the impacts from these contaminants entering groundwater would be discernible from background concentrations. Overall, there is no evidence of an evolving risk to the environment from the non-radiological component of the waste.

Non-radiological monitoring is performed on a triennial cycle, with analysis for an enhanced suite of potential contaminants (informed by leachate analysis) in the third year of the cycle. The triennial non-radiological monitoring cycle is supported by more frequent and spatially dense monitoring for tritium, enabling targeted additional non-radiological monitoring if deemed appropriate. We consider this to be proportionate to the non-radiological hazard posed by the waste.

6.1.4 Summary

Our key arguments that the LLWR disposal system provides appropriate protection against non-radiological hazards, thus complying with Requirement R10 of the GRA are as follows.

- The closure engineering at the LLWR will ensure that waste is contained for longer timescales than would be expected for landfill disposal.
- All vaults at the LLWR either have been, or will be, constructed from the surface and will not extend below the regional groundwater table. Over the lifetime of the site's Permit, the Regional Groundwater will remain below the level of the vaults, as

demonstrated in our hydrogeological studies. The 'prohibition' requirement of WER 2017 will therefore be satisfied by the LLWR as any discharges of pollutants must percolate through the underlying soil before entering the Regional Groundwater

- Our assessment calculations and monitoring data demonstrate that we prevent the input of hazardous substances to groundwater and limit the input of non-hazardous pollutants to groundwater during and after the PoA, thus complying with the requirements of EPR 2016.

6.2 Forward Plan

The extensive monitoring programme will continue throughout the PoA, continuing to provide confidence that the repository is performing as expected and not giving rise to unacceptable levels of environmental pollution. Annual monitoring review reports will continue to be written to summarise the findings of the previous year and these will be submitted to the Environment Agency for review.

No major changes to the monitoring programme are envisaged at present, although this situation is reviewed annually and changes will be introduced as deemed necessary or appropriate. On the basis of evidence presented here, we consider the triennial monitoring cycle to be working well and it will therefore be continued. LLWRAS values are reviewed annually against current guidance and updated if necessary.

Improving the inventory is a key underpinning task. We have been working to support the development of an improved treatment of non-radiological contaminants in future national inventories. We will also work to derive inventories of disposed substances (for which inventory estimates are not yet available), on the basis of newly-derived fingerprints where this is practicable.

Our quantitative HRA considered the potential impacts of contaminants in the grout used in vault disposals. Our source term for contaminants in grout was derived based on Buildings Research Establishment data. We will undertake work to develop a source term for grout used at the LLWR.

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