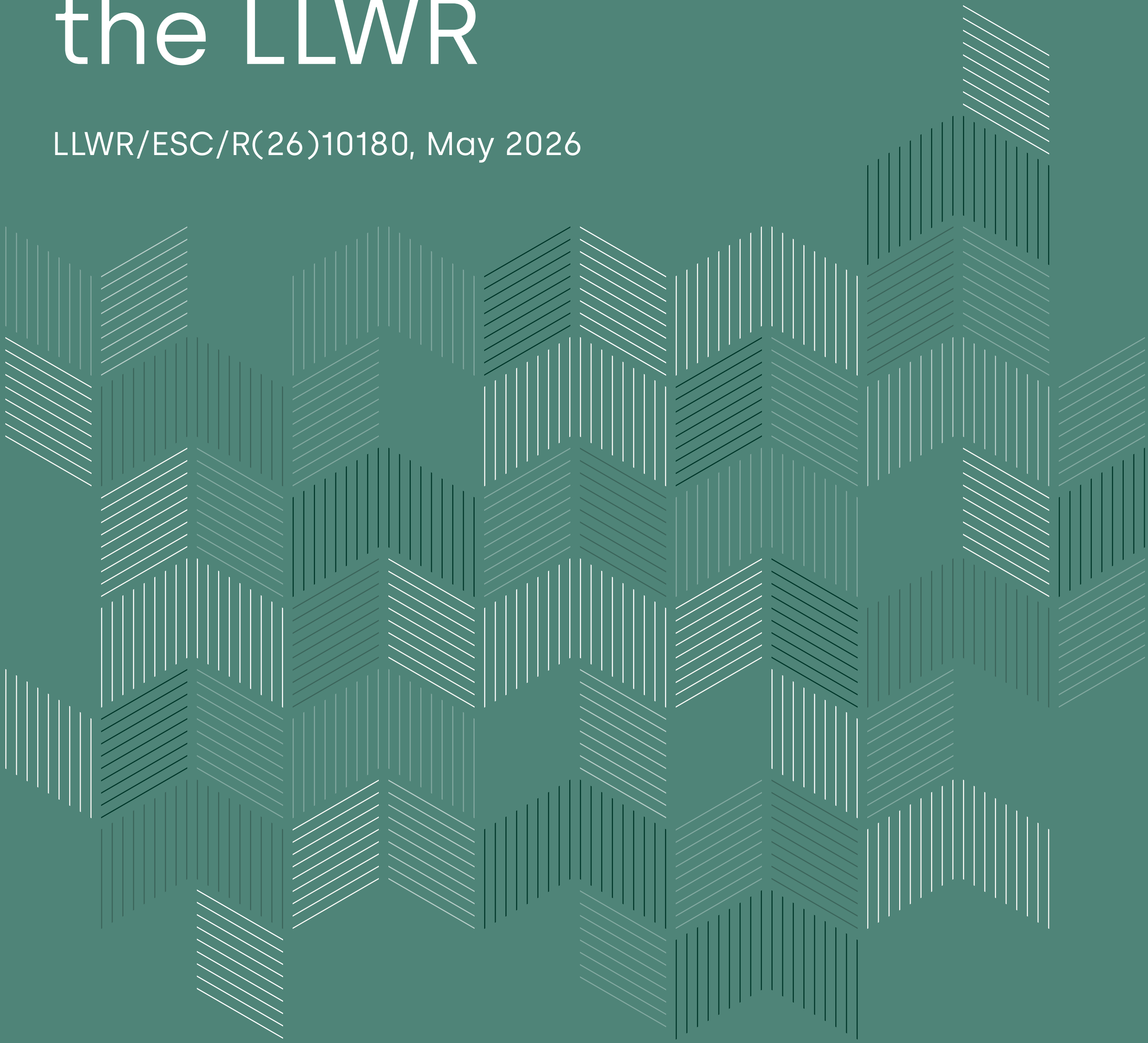
Nuclear Waste
Services

ASSESSMENT OF LONG-TERM RADIOLOGICAL IMPACTS

2026 Environmental Safety Case for the LLWR

LLWR/ESC/R(26)10180, 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 Kingdom[1].

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.

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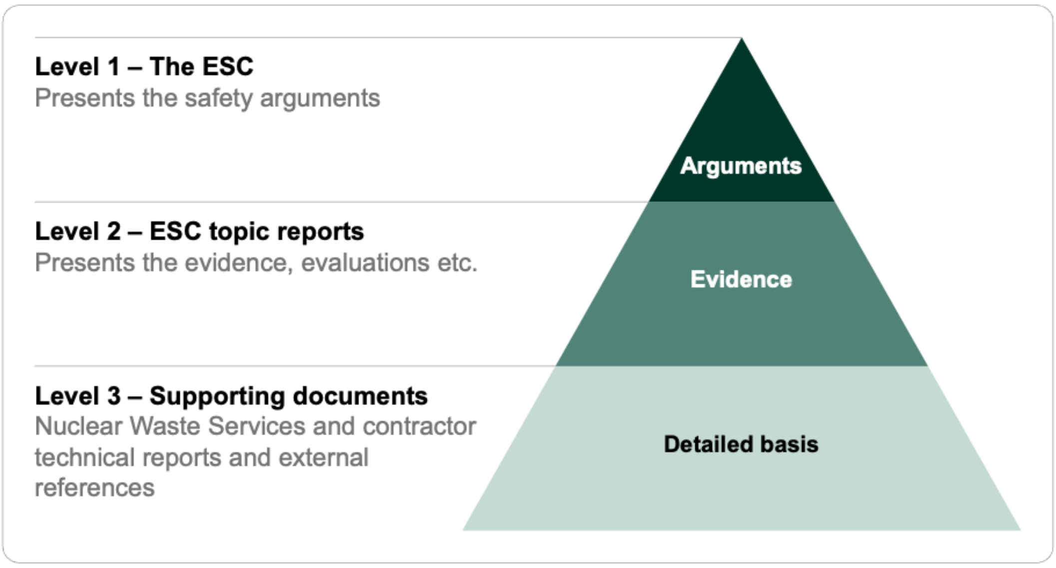
[1] In government policy, LLW is defined as radioactive waste having a radioactive content not exceeding four gigabecquerels per tonne (GBq t^{-1}) of alpha or 12 GBq t^{-1} of beta/gamma activity.

This is the Level 2 report ‘*Assessment of Long-term Radiological Impacts*’. 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 (this report)	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 [15]	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 report presents the assessment of long-term radiological impacts (LTRA) undertaken as part of the 2026 ESC for the Low Level Waste Repository (LLWR). The assessment includes the potential radiological impacts to humans and groundwater resulting from disposal of solid radioactive waste and includes the potential for a criticality event to occur in the repository. The 'long term' starts at the end of the Period of Authorisation (PoA), when the site Permit is surrendered and active controls on the site are withdrawn.

The key objective is to demonstrate the LLWR will be safe after the end of the PoA. The report aims to build confidence in our assessment results by showing our assessments are systematic, comprehensive, scientifically correct and have been undertaken and checked to a high standard.

To achieve the key objective, this report develops understanding of the disposal system behaviour, and shows that, accounting for all relevant uncertainties, the assessed radiological impacts are consistent with:

- Requirement R6 (risk guidance level after the PoA) and Requirement R7 (human intrusion after the PoA) of the environment agencies' guidance on requirements for authorisation for near-surface disposal facilities on land for solid radioactive wastes (the GRA);
- guidance from the Environment Agency on protection of groundwater.

This LTRA builds on and further develops previous long-term radiological assessments carried out in support of the Environmental Safety Case (ESC) submitted to the Environment Agency in 2011. There have been many developments since submission of the 2011 ESC. The following are the most significant.

- Update of the Reference Inventory, including exploration of the potential impacts of disposing certain, less hazardous, Intermediate Level Waste (ILW) at the LLWR site.
- Further optimisation of the vault design, including optimisation of the closure approach for the existing vaults with containers of the current designs, and optimisation of the design and closure approach for future disposals.
- Wider, fully integrated, ongoing optimisation work leading to improvements to the final cap and closure engineering design. Significantly increased confidence that infiltration into the closed repository will be low and remain low throughout the assessment timeframe. This has implications for conditions in the repository, with less leaching of cement from the vaults resulting in higher pH for longer, lower corrosion rates and less microbial activity than considered previously.
- Ongoing replacement of the current interim cap over the historic waste trenches after monitoring data indicated that the interim cap is not performing as well as expected, and intrusive investigations revealed damage to the low permeability geomembrane.

The assessment models are largely based on the models used in the 2011 ESC. They have been improved to better represent the system and updated to reflect the developments listed above and other developments, such as improvements to understanding of the geology and hydrogeology. The only exception is the post-PoA radon assessment, where a completely new model has been developed in response to changes in understanding of the long-term performance of the cap geomembrane and the important transport processes.

To provide clear, quantitative understanding of the key processes and results relevant to long-term radiological performance, we develop separate models to assess different 'pathways' by which contaminants are released from the waste, migrate through environmental media and give rise to exposure. The pathways that are assessed are the:

- groundwater pathway, which considers releases of radionuclides to groundwater;
- gas pathway, which considers releases of radioactive gases from the repository;
- coastal erosion pathway, which considers disruption of the repository by coastal erosion;
- human intrusion pathway, which considers inadvertent human intrusion into the repository.

Potential interactions between the pathways are considered and accounted for.

Each of the pathways is analysed and modelled, taking account of uncertainties. Models are developed to present a cautiously realistic representation of relevant processes. That is, we apply our knowledge to develop models of physical, chemical and biological processes that are realistic where data and understanding permits. Where there are uncertainties that are not quantifiable, we adopt data and model representations that are expected to result in the potential impacts being overestimated for the pathway under consideration, i.e. a cautious approach. We have used a systematic approach to comprehensively identify the biases and uncertainties in our assessments and decide how they should be treated. We have treated uncertainties according to the conventional formulation of scenario uncertainty, conceptual model uncertainty and parameter uncertainty.

All our assessments consider the potential implications of spatial and temporal variability, and spatial variations in the distribution of radionuclides. This includes potential radiological impacts from items with relatively high activity. In our underpinning work we have developed detailed models of biogeochemical processes in the near field and groundwater flow in the geosphere. We have used these models to investigate spatial variabilities in conditions, processes and properties that affect radionuclide concentrations.

We have assessed the radiation doses and risks to a person representative of the small numbers of people at greatest risk for each pathway. We expect people will be exposed to radionuclides released by coastal erosion of the repository. Therefore, for this pathway we calculate annual doses and compare them against a dose of 20 μSv, which is equivalent to the GRA risk guidance level assuming exposure occurs.

The calculated doses and risks are consistent with Requirement R6 of the GRA and the associated risk guidance level (Table ES.1), and with Requirement R7 and the associated dose guidance levels (Table ES.2). The doses and risks presented in Tables ES.1 and ES.2 are based on our Reference Inventory[2], which includes disposal of certain ILW.

Some people could be exposed by more than one pathway, but it is unlikely that a person would be representative of those at greatest risk for more than one pathway. Even in this unlikely situation, the combined risks from exposures resulting from more than one pathway are also consistent with the risk guidance level.

Table ES.1: Peak doses and radiological risks from reference cases

Pathway	Peak risk (y^{-1}) or dose (μSv)
Groundwater: Well biosphere pathway	Deterministic reference case 2 10^{-9} y^{-1} Probabilistic reference case mean 4 10^{-7} y^{-1}
Gas Total risk from C-14 and radon, cap vents closed at the end of the PoA	8 10^{-8} y^{-1}
Coastal erosion Recreational use of the coast	Trenches 22 μSv Vaults 11 μSv

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[2] The Stage 2 Reference Inventory is used in the assessment calculations, as described in the main text of this report.

Table ES.2: Peak doses from events where the probability of occurrence cannot reliably be assessed (additional events included in the main text)

Event	Duration	Peak dose (mSv)	GRA dose guidance level (mSv)
Site investigation and sample (core) analysis	Short	3.7	20
Construction of housing on the cap	Short	0.08	
Occupying a site contaminated by construction works	Long	2.1	3
Scavenging of waste exposed by coastal erosion	Long	1.8	
Living in a house built on the cap with a basement that intrudes into the cap (radon gas)	Long	1.3	

The calculated impacts are also consistent with the relevant parts of guidance from the Environment Agency on the protection of groundwater. The amount of fissile material in the inventory is too small, and too widely spread throughout the repository for a criticality event to occur.

The risks and doses reported above assume future disposal of LLW and certain ILW. We have also considered what the potential impacts of the repository would be if only LLW is disposed in the future. The peak potential groundwater pathway impacts are from existing disposals to the trenches, while the peak impacts from radon are from existing disposals of Ra-226 to the trenches and vaults. These impacts will not change if ILW is not disposed.

The potential impacts from future vault disposals are most significant for risks from C-14 gas, potential doses to people using the coast in front of the eroding repository for recreational use, and people consuming marine foodstuffs at a high rate while the repository is eroding. The peak area-averaged dose from C-14 gas over the vaults decreases by a factor of ten if only LLW is disposed, while the peak risk from the repository decreases by a factor of five. The peak doses from the vaults to recreational users of the coast and high-rate marine foodstuff consumers decrease by a factor of five. Peak doses to high-rate marine foodstuff consumers from the repository decrease by a factor of three.

Overall, our assessments demonstrate the LLWR is safe from the end of the PoA until the repository has been fully disrupted by coastal erosion. Thereafter, concentrations of radionuclides in surface coastal and marine sediments decrease as long-lived radionuclides are buried in marine sinks. The potential ongoing impacts from residual radioactivity in the coastal and marine environments decrease in parallel with the decreases in radionuclide concentrations.

We use the results of our assessments to:

- understand the disposal system behaviour and identify areas to prioritise for design optimisation and monitoring;
- calculate the amount of each radionuclide that can safely be disposed without exceeding the risk guidance level (termed radiological capacities);
- calculate the maximum concentration of each radionuclide that can safely be disposed without inadvertent human intrusion resulting in doses exceeding the dose guidance level;
- develop additional waste acceptance criteria (WAC) to control the waste that is accepted for disposal, such as controls on active particles;
- develop controls on how we manage disposals to the repository, such as emplacement requirements.

These controls ensure the potential impacts remain consistent with the GRA.

Nuclear safety is not discussed in this report but is addressed in our Nuclear Safety Case.

The forward inventory, our assessments, and these controls have been developed iteratively. The assessment results demonstrate that an overall assurance of safety can be provided by developing WAC and controls on the radionuclide activities and concentrations that can be disposed to the repository. The reference and uncertainty cases explore how the system performs, inform and build confidence in the WAC and these controls.

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1 Introduction

The Low Level Waste Repository (LLWR) provides 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 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. One of the means we use to operate the LLWR safely is to maintain and implement an Environmental Safety Case (ESC) for the site. The 2026 ESC is a major update based on a comprehensive review of our previous 2011 ESC and subsequent developments.

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.

This is a Level 2 report.

1.1 Objectives

This report presents the assessment of long-term radiological impacts (LTRA) in support of the 2026 ESC for the LLWR [1]. The assessment includes the potential radiological impacts to humans and groundwater resulting from disposal of solid radioactive waste and the potential for a criticality event to occur in the repository. The 'long term' starts at the end of the Period of Authorisation (PoA), when the site Permit is surrendered and active controls on the site are withdrawn[3].

The key objective is to demonstrate the LLWR will be safe after the end of the PoA. The report aims to build confidence in our assessment results by showing our assessments are systematic, comprehensive, scientifically correct and have been undertaken and checked to a high standard.

For some radionuclides and release pathways, evolution of the repository and releases during the PoA affect releases and potential impacts after the PoA. In these situations, our assessments, and models, consider continuous evolution of the repository, from first

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[3] The end of the PoA follows completion of waste disposal operations and installation of the closure engineering, and a subsequent period of active institutional control including monitoring that is expected to last for 100 years [10]. Reference assumptions for the 2026 ESC are completion of disposals in 2135, completion of closure engineering in 2142, end of active institutional control and the PoA at 2242.

disposals in 1959 to the end of the assessment time frame in around 5,000 years. However, only the potential post-PoA impacts are reported here. Potential impacts during the PoA are reported in reference [14].

The primary goals of this report are to develop understanding of the disposal system behaviour, and to show that, accounting for all relevant uncertainties, the assessed radiological impacts are consistent with (Box 1.1):

- Requirement R6 (risk guidance level after the PoA) and R7 (human intrusion after the PoA) of the environment agencies' guidance on requirements for authorisation for near-surface facilities for disposal of radioactive waste (the GRA) [19];
- guidance from the Environment Agency on protection of groundwater [20] [21].

The potential impacts on groundwater are assessed in greater detail in our hydrological risk assessment [15], which considers radionuclides and non-radiological contaminants.

Box 1.1: GRA requirements R6 and R7

GRA requirements R6 and R7

6.3.9 For any time after closure of the facility where the developer/operator does not claim, or we do not accept, that there will be active institutional control, our regulatory approach will be to apply a risk guidance level (see Requirement R6 below) and, for human intrusion, a dose guidance level (see Requirement R7 below).

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).

Requirement R7: Human intrusion after the period of authorisation

The developer/operator of a near-surface disposal facility should assess the potential consequences of human intrusion into the facility after the period of authorisation on the basis that it is likely to occur. The developer/operator should, however, consider and implement any practical measures that might reduce the chance of its happening. The assessed effective dose to any person during and after the assumed intrusion should not exceed a dose guidance level in the range of around 3 mSv per year to around 20 mSv per year. Values towards the lower end of this range are applicable to assessed exposures continuing over a period of years (prolonged exposures), while values towards the upper end of the range are applicable to assessed exposures that are only short term (transitory exposures).

1.2 Scope

This report presents assessments of the radiological impacts to hypothetical persons representative of those at greatest risk as a result of the disposal of radioactive waste at the LLWR. We examine the potential impacts assuming disposal of Low Level Waste (LLW) and certain, less hazardous, Intermediate Level Waste (ILW). Inclusion of less-hazardous ILW in the scope of the ESC provides a means to engage with stakeholders on the most appropriate use of the LLWR site, consistent with the policy framework for managing radioactive substances and nuclear decommissioning published by the UK government and devolved administrations in May 2024 [22]. It does not imply that any decisions have yet been made.

The report first sets out the context for and approach to the post-PoA radiological assessments consistent with the GRA. This report only assesses the potential impacts from waste disposed to the repository. This report does not assess potential impacts from other contamination on the LLWR site. The potential impacts from other contamination on the LLWR site are addressed in reference [1].

Then, based on the detailed characterisation and understanding of the LLWR and its future evolution as set out in other volumes of the 2026 ESC, especially references [3] [4] [5] [6] [7] [8] [13], this report presents a synthesis of our understanding of the LLWR and the evolution of the facility and its environment at a level needed to understand the scenarios and cases that we then analyse.

The radiological performance of the LLWR is analysed in terms of different mechanisms by which contaminants may be released, migrate or give rise to exposure to humans or the presence of contaminants in the biosphere, which we term pathways. Four pathways are considered – groundwater, gas, natural disruption (coastal erosion) and inadvertent human intrusion. Interactions between pathways and the potential for criticality in the repository are also considered.

For each pathway, and possible human intrusion events, a set of models, assessment cases and calculations are described that explore uncertainties about the characteristics and future evolution of the LLWR and its environment, and therefore the long-term radiological performance of the LLWR. Thereby, we calculate the range of radiological impacts such as may realistically or cautiously occur due to disposal of solid radioactive waste at the LLWR. The estimates are compared with the guidance levels set out in the GRA, see Subsection 1.1, and the important uncertainties and their influences are explored.

The key outputs in the assessments are radiological doses and risks to potential human receptors. Radiological risk is a combination of the estimated radiation dose that could be received, the estimated probability of this dose occurring, and the probability that this dose would cause detriment to human health (Subsection 2.13.3). Radiation dose is a measure of the energy deposited by ionising radiation in a target, in this case humans, measured in Sieverts (Sv) (Subsection 2.13.2).

This report is not concerned with non-radiological impacts or impacts to non-human biota, which are assessed separately [15] [16], though concentrations of radionuclides in environmental media calculated in the LTRA are used in the non-human biota assessment. The report is companion to a separate but compatible assessment of environmental safety during the PoA [14]. The report does not address waste acceptance, although radiological capacities (which describe the maximum amount of each radionuclide that can safety be disposed), 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 models, analyses and calculations summarised in this report are described in more detail in of a series of supporting assessment calculation reports, notably in references [23] [24] [25] [26] [27].

1.3 Structure

The structure of this report is as follows.

- Section 1, this section, has introduced the objectives and outlined the scope of the report.
- Section 2 describes the approach to assessing potential long-term (post-PoA) radiological impacts in the 2026 ESC.
- Section 3 provides an overview of the LLWR facility and its environs, the Site Development Plan (SDP), the radioactive inventory, and an integrated description of the evolution of the LLWR and its environment. The overview is at a sufficient level of detail to understand the motivation for the scenarios and cases that are assessed in subsequent sections of the report.
- Section 4 outlines the selection of radionuclides to be considered in the assessment calculations. A compilation of reference radiological data and a model of radiological hazard are described. This leads to a list of radionuclides to be included in the assessment calculations described in Sections 5 to 11.
- Section 5 presents the background, approach to assessment, models and data, case selection, and assessment calculation results for the groundwater pathway. A complete description of these elements is presented in reference [23]. The final subsection provides an assessment of the results against the regulatory guidance.
- Section 6 presents the background, approach to assessment, models and data, case selection, and assessment calculation results for the gas pathway. A complete description of these elements is presented in references [26]. The final subsection provides an assessment of the results against the regulatory guidance.
- Section 7 presents the background, approach to assessment, models and data, case selection, and assessment calculation results for coastal erosion. A complete description of these elements is presented in reference [25]. The final subsection provides an assessment of the results against the regulatory guidance.
- Section 8 presents the background, approach to assessment, models and data, case selection, and assessment calculation results for human intrusion. A complete description of these elements is presented in reference [24]. The final subsection provides an assessment of the results against the regulatory guidance.
- Section 9 presents an assessment of criticality. A complete description of the assessment is presented in reference [27].
- Section 10 presents assessments relating to the heterogeneous distribution of activity in the waste, including the potential impacts associated with individual waste items and particles derived from waste.
- Section 11 describes the potential risks from receipt of doses from more than one pathway.
- Section 12 summarises the assessed radiation risks and doses, and their consistency with GRA Requirement R6 and R7 and guidance on protection of

groundwater. Finally, it identifies key uncertainties that could most affect the outcome of our assessments.

A general glossary for the ESC is appended to the ‘*Main Report*’ [1].

2 Assessment Approach

2.1 Regulatory Guidance

Disposal of radioactive waste at the LLWR is regulated by the Environment Agency under the Environmental Permitting (England and Wales) Regulations 2016.

The Environment Agency, Natural Resources Wales, and the Scottish Environment Protection Agency (the environment agencies) jointly issued guidance in 2018: '*Management of Radioactive Waste from Decommissioning of Nuclear Sites: Guidance on Requirements for Release from Radioactive Substances Regulation*' (the GRR) [28]. It applies to all nuclear sites and describes what operators must do over the lifetime of their site to achieve release from radioactive substances regulation. The GRR consists of five principles and fifteen specific requirements. The GRR requires operators to develop and maintain a Waste Management Plan (WMP) and a Site-wide Environmental Safety Case (SWESC) and to ensure the condition of their site meets regulatory standards for protection of people and the environment, now and into the future.

Regulatory guidance for disposal of radioactive waste for near-surface facilities is contained in the GRA. The GRA consists of five Principles, fourteen top-level Requirements, and several hundred more detailed requirements. A key requirement of the GRA is to develop and maintain an ESC. We understand that the GRA is being updated by the Environment Agency, and we have responded to the consultation.

The GRR explains that a constructed disposal facility must meet the requirements of the GRA and will have its own ESC, which will define the WAC for the facility. The ESC for the disposal facility will be a component of the SWESC. We have followed this approach by embedding the ESC for the dedicated disposal facility within a SWESC. However, the main purpose of the LLWR site has been and will continue to be the disposal of radioactive waste in a dedicated disposal facility. Our focus is therefore on the requirements of the GRA [18], though we refer to the GRR where it provides more specific guidance that is not provided by the GRA.

The Environment Agency has also provided guidance on protection of groundwater [20] [21]. Criteria after the end of regulation are relevant to this report. Reference [21] says an input 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 regulation of a solid radioactive waste disposal is consistent with, or lower than, a risk guidance level of 10^{-6} per year.'*

2.2 Previous Submissions and Feedback from the Environment Agency

Environmental safety cases and related assessments for the LLWR have been prepared periodically since the late 1980s. Previous ESCs reflect the regulatory guidance, national policies and strategies, site conditions and understanding, and development plans that were relevant at the time.

The last major update to the ESC submitted to the Environment Agency was in 2011 (the 2011 ESC). Following a detailed review of the ESC, the Environment Agency issued a revised Permit for the LLWR, allowing continued disposal of Low Level Waste (LLW) at the facility.

The Environment Agency raised Forward Issues (FIs) and gave recommendations from its review of the 2011 ESC [29]. The Environment Agency recognised that the 2011 ESC was a complex submission involving a wide range of technical assessments that will evolve and improve in the future as technology and understanding advances. It also recognised that certain details will be developed further as the site advances, for example, toward construction of the final engineered cap over the waste. The FIs are areas of work where the Environment Agency saw scope for continued improvement in the ESC and its implementation. Our responses to the FIs are summarised in reference [18].

Improvement Condition 7 (IC7) of our Permit requires us to submit an update to the ESC by 1st May 2026. IC7 calls for the 2026 ESC to meet all the requirements of the latest version of the GRA and to address the findings of the Environment Agency's review of the 2011 ESC. Improvement Condition 8 (IC8) requires us to develop a SWESC to fulfil the requirements of the GRR by 1st May 2026. As noted previously, the ESC for the repository is embedded in the SWESC.

2.3 Key Developments

There have been many developments since submission of the 2011 ESC. The most significant developments that affect the LTRA are summarised below.

- We have explored options for disposing of ILW at the LLWR site, in line with a request from the NDA and Government policy to consider alternative options for the management of ILW. The 2026 ESC explores the potential impacts of disposing certain, less hazardous, ILW in the repository. As noted previously, this provides a basis for us to engage with the regulators and other stakeholders to explore the potential to dispose ILW at the LLWR site. However, it does not indicate that a decision to dispose ILW has been made.
- We have developed an updated Reference Inventory for disposals to the LLWR. This captures updates to the UK Radioactive Waste Inventory (UKRWI) and the impacts of the step changes in waste treatment and diversion implemented after the 2011 ESC. The future LLW inventory therefore includes relevant LLW from the UKRWI that could be disposed following diversion, and our best estimate of the secondary waste arising from waste treatment. The Reference Inventory also includes certain ILW that could potentially be disposed within constraints derived from the ESC and the nuclear safety case.

- We identified that it would not be possible to install the closure engineering in the way described in the 2011 ESC without damaging waste containers of the current designs [5] [10]. This would reduce the contribution of the containers to the multi-barrier repository concept. Therefore, we further optimised the vault design, including the closure approach for containers of existing designs, and the container design for future waste and approaches to protecting future containers from damage during closure. These updates are fully integrated with the optimal approaches to potentially disposing ILW in the vaults. The preferred approaches are optimal whether ILW is disposed or not.
- Wider, ongoing optimisation work, fully integrated with the above, has resulted in improvements to the final cap and closure engineering design. Combined with significant developments in understanding of the lifetime of cap geomembranes, there is significantly increased confidence that infiltration into the capped repository will be low and will remain low throughout the assessment timeframe. This has implications for conditions in the repository, with less leaching of cement from the vaults resulting in higher pH for longer, lower corrosion rates and less microbial activity than considered previously.
- Analysis of monitoring data indicated the interim cap over the historic waste trenches is not performing as well as expected from the design. Intrusive investigations revealed damage to the geomembrane in the cap, which likely occurred during construction. The type and extent of damage is consistent with the outcomes of the analysis of monitoring data. The interim cap over the trenches is being replaced to improve isolation and containment of the trench wastes until the trenches are covered by the final cap. The interim trench cap is not being replaced over the north end of the trenches as this area will be soon covered by the first strip of the final cap.

2.4 Assessment Framework

We follow the well-established approach to long-term assessment of radioactive waste disposal facilities, as described by the Nuclear Energy Agency (NEA) [30] and by the International Atomic Energy Agency (IAEA) [31] [32]. This includes activities that are interlinked:

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 from the repository to humans and the environment, and refinement or 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 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 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.

The acquisition of information and development of understanding (item 1) is set out in other ESC reports [3] [4] [5] [6] [7] [9] [10] [13] and summarised in Section 3. The pathways that could lead to transfer of radionuclides from the repository to humans and the environment are:

- transport in groundwater − the groundwater pathway;
- transport in gas − the gas pathway;
- disruption of the repository by coastal erosion;
- inadvertent human intrusion into the repository.

Scenarios for the long-term evolution of the disposal system and its environment (item 3) are set out in Subsection 3.5. The application in analysis and evaluation (items 4, 5 and 6) are described in Sections 5 to 11 of this report.

We have developed an Assessments Manual [33] to ensure assessments use a consistent, comprehensive and robust approach. Assessments are undertaken by multidisciplinary teams, to ensure all relevant factors are identified and assessed appropriately. This usually comprises a core team who consult wider experts as appropriate. Multidisciplinary workshops are an important tool for assessment development and review. Several mechanisms are used to share information between assessments. These include interactions between assessment team leads, written and meeting information exchanges, and following the ESC data management procedure [34]. The data management procedure ensures consistent use and understanding of data between assessments, and controls passing of results from one assessment as input to another assessment.

ESC assessments have been progressively developed since the 2011 ESC. Updated assessments have been shared with the Environment Agency as part of applying for a new Permit to dispose, meeting Permit requirements, updating radiological capacities, and ongoing dialogue with the Environment Agency. Assessments for the 2026 ESC build on the 2011 ESC assessments and subsequent developments. Assessments for the 2026 ESC have been developed through two phases, providing the opportunity for iteration. This includes addressing comments from the Environment Agency and our ESC independent peer review group on the Phase 1 assessments.

In the following subsections, we outline our approach to treatment of uncertainties, identification of pathways, identification of scenarios and cases, and achieving model quality. Assessment endpoints and calculation methods are discussed in Subsection 2.13.

2.5 Uncertainty and Bias

2.5.1 Uncertainty

Uncertainty is present where incomplete information is available about some aspect of the assessment. Uncertainties are particularly large when projecting the evolution of a passive environmental system for thousands of years into the future.

Properties of various media within the repository system will vary from place to place (e.g. soil density) and over time (e.g. rainfall rate). The intrinsic spatial or temporal variation in a specific property of a medium or process is referred to as its variability. Variability is distinct from uncertainty and is discussed in Subsection 2.6.

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 the long-term radiological assessment and the wider ESC.

Sources of uncertainty

Assessment uncertainties comprise:

- uncertainty in the conceptual model for the system, e.g. where different interpretations are possible;
- uncertainty about the values of parameters;
- uncertainty in the future evolution of the system.

In the context of the LLWR and its assessment, uncertainties arise from:

- uncertainties concerning past and future waste disposal operations, e.g. the radiological 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.

Our goal for treating uncertainty

Our goal within the LTRA is to present 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 risk and dose guidance levels.

The risk guidance level is deliberately set to a low level. Therefore, the Environment Agency advise that it is not necessary for assessments to apply a deliberate level of caution when calculating risks for comparison with the guidance level. We represent the system as realistically as possible in our assessments but take a cautious approach where the most realistic representation cannot be robustly quantified. We explore uncertainties to understand whether they could lead to significantly different outcomes when assessment results are compared against the risk guidance level.

Some uncertainties will remain unresolved, but this does not prevent an assessment of 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 systematically identify uncertainties as we develop our assessments. We classify uncertainties using a conventional approach to radioactive waste disposal assessment, e.g. [35] [36] [37] [38], 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;
- conceptual model uncertainty – uncertainty in 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.

Parameters are generally treated as uncertain where the range of potential values could have a first order effect on radionuclide release, transport or environmental concentrations. However, the following parameter uncertainties are excluded:

- biosphere parameters pertaining to human behaviours and habits;
- radionuclide half-lives;
- dosimetric quantities.

Uncertainties in human behaviour and habits are not explored because we select behaviours and habits that are protective of the most exposed people. Half-lives and dosimetric quantities are relatively well known compared with other biosphere parameters, and uncertainty in these parameters is much smaller than the wider assessment uncertainties.

We also undertake systematic and comprehensive audits to ensure all important uncertainties have been identified and appropriately treated in our assessments. Audits are further described in Subsection 2.5.3.

Our approach to the treatment of uncertainties is proportionate and commensurate with the amount of information available. A more comprehensive or sophisticated approach is applied where the system is complex and hard to understand.

Within the 2026 LTRA, we have:

- identified the pathways leading to the transfer of radionuclides from the repository to humans and the environment (Subsection 2.7) and characterised the uncertainties for each pathway;
- identified scenarios for the future evolution of the natural environment, and identified cases for assessment within each (Subsection 2.8);
- developed sound conceptual models of the pathways, including considering alternative assumptions, and implemented the models in quality assured codes (Subsection 2.10);
- applied these assessment models to analyse and assess each of the pathways including taking account of alternative assumptions and parameter uncertainty (Sections 5 to 8).

Our Engineering Performance Assessment (EPA) [13] describes the long-term evolution and performance of the repository engineered components, while our studies of the near field investigate the biogeochemical conditions and rates of processes in the repository. The outputs from the EPA and near-field studies include conceptual and parameter uncertainties. These uncertainties feed into our assessments. Uncertainties that are potentially significant for an assessment are identified and explored.

We ensure consistency in data that are common to different assessments [34]. Data that are common to different assessments, or results passed from one assessment to another, are recorded in Data Management Forms (DMFs). The DMFs describe uncertainty in parameter values. This helps us determine whether the uncertainty is potentially significant for an assessment and therefore should be explored in the assessment. If the uncertainty is potentially significant, the DMF provides parameter ranges for assessment calculations.

We have used formal review and elicitation approaches [39] to develop parameter values as outputs from the EPA [13], which are then documented in DMFs. We have also used elicitation and expert review of corrosion rate data to develop corrosion rates [6], which are an important input to our near-field studies and assessments. Other parameter values are developed by assessment teams, by drawing on literature data, making recorded judgements or deriving data from calculations.

For each pathway we develop a reference calculation case, which comprises our reference scenario, best estimate conceptual model and best estimate parameter values. We then use variant cases to explore alternative scenarios, conceptual models and parameter values. We also explore alternative approaches to representing the conceptual models in our assessment models. Scenarios and calculation cases are described in more detail in Subsection 2.8.

We evaluate most of our calculation cases deterministically. However, for some cases we have sufficient understanding of the uncertainty in parameter values to undertake probabilistic risk assessment (PRA) calculations. Our calculation methods are described in more detail in Subsection 2.13.

2.5.2 Bias

A bias is an assumption, condition, or caution within models (or any other aspect of a process in a model) that results in, or is likely to result in, an inaccurate representation of the system under consideration.

We have sought to reduce biases in assessment models through developments to the assessment made since the 2011 ESC, for example an updated assessment of the potential impacts of C-14 gas prepared in 2013 [40]. We have also used the results of the phase one assessments for the 2026 ESC to reduce biases in the phase two assessments. We use variant calculation cases to explore and quantify biases.

2.5.3 Audit of Biases and Uncertainties

We have developed a systematic process to comprehensively identifying and addressing the key biases and uncertainties in our assessments [33]. This ensures that a common, robust and proportionate approach is used for each assessment. We have undertaken audits of the key biases and uncertainties at the end of the phase one assessments. We have identified key uncertainties and biases to remove, reduce, or explore, as appropriate, during Phase 2. The audits have been updated at the end of Phase 2. The audits provide confidence that the main uncertainties and biases have been identified and make them visible.

We identify any biases or uncertainties that require further work and log those in our register of significant uncertainties [41]. The register of significant uncertainties includes uncertainties that are identified during assessments and those identified during other work that we undertake to develop and underpin the ESC.

2.6 Variability

Properties of various media and process rates within the repository system vary from place to place (e.g. soil density) and over time (e.g. rainfall rate) [33]. Therefore, there is spatial and temporal variability. Both are considered in our assessments. Variability is synonymous with heterogeneity.

Variability is addressed in our assessment models and underpinning detailed modelling work by:

- explicitly representing differences in properties and processes in space or time; or,

- adopting an implicit treatment with the use of appropriate effective properties.

Explicitly representing variability requires an appropriate level of discretisation of a model in space or time. Implicitly representing variability involves selecting and justifying a suitable effective property or time average.

Our assessment and detailed underpinning models are discretised into areas and time periods where the characteristics, properties, process rates and radionuclide concentrations are reasonably uniform. We use variant cases (Subsection 2.9) and 'side' models (that are additional to the main assessment model) to explore sensitivity to discretisation and parameter values.

The assessment end point is also an important consideration for the required discretisation (Subsection 2.13). Risk from the repository depends on the expectation value of doses averaged over the occupied area. Although greater discretisation (where possible and practicable) provides more information on spatial variation in doses, it does not significantly affect the expectation value of the dose and therefore the risk. The objective is to ensure that the level of discretisation in time and space is appropriate for the receptors described in Subsection 2.8 [42].

In many cases, people could be exposed to radionuclides distributed through certain areas of environment, such as the Irt Estuary, coast adjacent to the eroding repository, or marine environment. Radionuclides from many individual disposals are subject to mixing, dispersion and attenuation in the near field, geosphere and biosphere, so a reasonably coarse discretisation of the system is appropriate for assessment models. However, there are instances where smaller scale variations are important. People could inadvertently intrude into specific disposals or be exposed to particulates or discrete items of waste revealed by coastal erosion. Also, spatial variations in the distributions radionuclides within individual vaults could affect the activity exposed at a given time and therefore the risks from coastal erosion.

It is possible in principle to model individual waste items or specific disposals, but it is not feasible to obtain comprehensive data on the characteristics and future condition of individual items, nor to predict how items will be distributed and exposed in future disposals within a vault. These limitations introduce bias in our assessment models. We address this by exploring and implementing controls on activity burdens associated with particles and discrete items of waste, controls on heterogeneity in individual containers, and controls on waste emplacement. These controls are reflected in our WAC and in how we manage disposals to the repository [17].

We have developed detailed models of biogeochemical processes in the near field and, separately, groundwater flow in the geosphere. We have used these models to investigate spatial variabilities in conditions, processes and properties that affect radionuclide concentrations. The results inform our treatment of spatial variability in our detailed models and transfer of outputs from our detailed models into our assessment models.

We have explored the effects of spatial variability in the material inventories, conditions and processes in the near field, and in the properties of the geosphere (and therefore groundwater flows), that affect the spatial distribution of radioactivity at length scales from very small to large (Section 10).

2.7 Pathways

At any time, many processes may be occurring, and radionuclides or other contaminants may be being released, migrating, accumulating and giving rise to exposure by multiple routes and to multiple receptors.

To provide representations that provide clear understanding of the key processes and results, and to make the analysis tractable, we choose to develop separate models to examine characteristically different pathways by which radionuclides or contaminant may be released, migrate or give rise to exposure. We identify the following four pathways.

- Groundwater pathway − release of dissolved radionuclides to groundwater and subsequent transport by groundwater flow.
- Gas pathway − release of gaseous radionuclides and subsequent transport by diffusion, and by advection within much larger volumes of 'bulk' gas generated by degradation of the waste and containers.
- Coastal erosion pathway − disruption of the repository by coastal erosion and subsequent transport and dispersion of waste and associated radionuclides in the coastal and marine environment.
- Human intrusion pathway − human activities which could lead to inadvertent uncovering of the wastes and doses associated with the event or subsequent occupancy of a contaminated site.

In the groundwater pathway assessment, we consider four biosphere pathways:

- abstraction and usage of contaminated water from a well between the repository and the coast (well pathway);
- natural discharge of contaminants to the marine environment (marine pathway);
- discharge of contaminants to the Drigg Stream (stream pathway);
- discharge of contaminants to the Irt Estuary (estuary pathway).

These biosphere pathways are further described in Section 5.

We assess the potential impacts of the repository for each pathway (and biosphere pathway). We recognise that there is interaction, and the potential for addition, between the pathways. For example, changing vault or environmental conditions will simultaneously affect release and migration in more than one pathway, and it is possible that an individual could be simultaneously exposed to more than one pathway. The potential interactions are considered and accounted for in the uncertainties that are represented in the assessment of each individual pathway.

It is possible for people to be exposed by more than one pathway because releases from the pathways are to the same or adjacent areas of the biosphere. We have developed a habits handbook for the 2026 [43] which describes the habits of people who may be exposed. This ensures consistency in the calculations for each pathway, so risks can be summed where exposure to more than one pathway is possible. Although the habits handbook ensures consistency in calculation parameter values, adding the risks from different pathways may imply an extreme set of habits. These habits might not be reasonable, or only potentially relevant to one or a few individuals rather than a person representative of the people at greatest risk.

To decide whether people could be exposed by more than one pathway we compare the timing and location of impacts. In general, we find that one pathway dominates the risks, because one pathway has significantly higher peak risks than the other(s), or the peak risks occur at different times.

Details of the assessment of each of these pathways for the LLWR are set out in the main technical sections of this report (Sections 5 to 8). The potential combined impacts from different exposure pathways are discussed in Section 11.

All pathway assessments consider the potential implications of spatial heterogeneities in the inventory for potential doses and risks. The coastal erosion and human intrusion assessments consider people being exposed to small volumes of waste, individual waste items, and particles derived from waste. It is not possible to undertake assessments for individual waste items, combinations of items, or materials derived from items. For example, it is not known when individual items would be exposed by coastal erosion or might be exposed by human intrusion. Also, the condition when exposed, remaining activity burden and subsequent fate are uncertain. However, the implications of heterogeneity at all length scales are considered to develop appropriate controls on heterogeneity in the WAC, including controls on discrete items, sealed sources and particles present in the waste or that could be derived from the waste (Section 10).

2.8 Receptors

Potential receptors include people, non-human biota and groundwater. Non-human biota are discussed in reference [16]. Environment Agency guidance on groundwater is discussed in Subsection 2.1.

The GRA [19] requires radiological risk to be assessed to a person representative of those at greatest risk (Requirement R6). The person representative of those at greatest risk is the same as a representative member of the potentially exposed group (PEG) at greatest risk. The GRA also requires doses during the PoA to be assessed to a representative member of the critical group (Requirement R5). We use a consistent approach to identify representative persons for the for the PoA and post-PoA periods. Our approach is cautiously realistic at early times, with increasing caution applied at later times to reflect increasing uncertainty in human behaviours [44].

Whilst there is limited guidance on defining representative persons and their habits for near-surface disposal facilities, we have taken note of the guidance that applies to impacts from permitted discharges from nuclear sites. We have adapted such guidance to our circumstances, particularly to reflect the extended timescales of interest. We have also undertaken trials of alternative approaches to inform our approach.

We have retained the approach used in the 2011 ESC of defining what we judge to be cautiously realistic exposure groups for the post-PoA period, but the representative member of that group is now termed a potentially representative person (PRP), rather than a member of a PEG. They continue to be based on a narrative description of potential future human behaviours. This approach provides a balanced and proportionate mechanism for examining impacts from the repository over the longer term. The narrative approach is supported by the IAEA biosphere modelling and assessment (BIOMASS) methodology [45].

When defining habits for such PRPs, we consider both local and national generic habits information but are not constrained by this information if it is not consistent with the narrative description of the PRP. We consider additivity over pathways and different age groups.

Our Habits Handbook [43] includes 95th percentile occupancies and consumption rates from local and national surveys. These data are used in our assessments where the same habits are assumed in the future. Where we develop narratives of how people might use the site and adjacent land in the future, the narratives may be inherently cautious. For example:

- assuming that a smallholding is built on the cap maximises carbon intake rates and doses from C-14 gas compared with other activities such as using the cap as part of a gazing area for a large farm;
- assuming a well exists between the repository and the coast, in an area that is currently a SSSI, and people then drink the water and use it for irrigation.

We consider that it is disproportionate to calculate doses or risks to a range of age groups for each narrative description of potential human behaviours for the post-PoA period, because differences between age groups are generally smaller than the wider uncertainties. This is consistent with the recommendations from the ICRP to consider doses or risks to adults as an adequate representation of long-term exposure [46] [47]. However, we have reviewed habits information for different age groups and undertaken selected quantitative dose and risk calculations for adults, children and infants. The outcomes support this argument. We have also qualitatively assessed whether specific behaviours or elevated habit rates could lead to children or infants receiving substantially higher doses than adults, e.g. pica [24].

Compared with our approach in the 2011 ESC, our updated approach:

- provides a stronger link to local habits data, and a clearer logical link to the underlying habits data source;

- reflects increasing uncertainty in human habits over increasing timescales by moving from local to generic to narrative-based habit data and exposure pathways;
- easily facilitates a realistic representation of the contribution of multiple release pathways to a PRP.

2.9 Scenarios and Cases

General definitions

Scenarios are defined as 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.

Many calculation cases are required to adequately explore all aspects and uncertainties across these scenarios. 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 effects of:

- conceptual model uncertainties, including uncertainties in engineering performance, and alternative conceptual models;
- uncertainty in the waste that will be disposed in the future;
- parameter uncertainties;
- alternative modelling approaches or assumptions.

Scenarios and Cases for the 2026 ESC

We assess the potential radiological impacts assuming development of the facility according to an optimised SDP. This plan includes existing disposals, an illustration of relevant LLW in the UKRWI that could be disposed to the LLWR following treatment and diversion, and less-hazardous ILW in the UKRWI that might be suitable for vault disposal if this is permitted. We also assess alternative cases that assume disposal of LLW only.

For the assessments in support of the 2026 ESC, we define three scenarios that encompass the broad expectation for evolution of the disposed wastes, engineered barriers and natural environment. These three scenarios correspond to low, reference and high projections of future greenhouse gas emissions, and associated impacts on climate and sea level. The three scenarios are underpinned by common characterisation and understanding of the waste, engineered barriers, hydrogeology of the site and environmental processes and their impacts [3] [4] [5] [7] [48]. Uncertainties within these scenarios are treated by alternative assessment cases, appropriate to each of the pathways.

The repository is located close to the coast, which is slowly eroding. Even without climate change and associated relative sea-level rise, the repository is expected to be disrupted by coastal erosion. Climate change and sea-level rise is expected to increase the rate of coastal erosion. Within the three assessment scenarios, the repository is expected to be disrupted by coastal erosion on a timescale of several hundred to a few thousand years after present, with erosion of the repository being complete within one to a few thousand years after present [48]. Timescales are expected to be towards the earlier end of the range with high emissions, and towards the later end of the range with low emissions.

We also assess a Delayed Coastal Erosion case, in which we explore the proposition that the site will not be eroded within this time frame. This 'what-if' case is not plausible, but it is informative to examine the longer-term performance of the engineered barriers and examine the long-term ingrowth of daughter radionuclides.

Reference [49] provides guidance on regulators’ expectations for the use of UK Climate Projections 2018 (UKCP18) and credible maximum climate scenarios. The guidance notes that an application for a nuclear development or installation needs a high level of climate resilience built in from the outset. Applicants should be able to demonstrate their proposals can be adapted over their predicted lifetimes to remain resilient to a credible maximum climate change scenario. In this context, the 'predicted lifetime' of the LLWR is the PoA, because this is the period over which adaptations can be implemented. Credible maximum scenarios can also be used for sensitivity testing of different adaptation options over time periods appropriate for the nuclear industry.

Our high emission scenario is an appropriate 'credible maximum climate scenario' in this context [25]. This scenario is included in our assessment of potential impacts during the PoA [14] and in our assessment of potential post-PoA impacts (this report). More extreme 'what-if' climate evolution cases are also considered in our post-PoA coastal erosion assessment (Section 7 and reference [25]).

It is cautious to use a credible maximum climate scenario for changes during the PoA, for example changes in rainfall that site drainage systems need to accommodate. However, our High Emissions Scenario only leads to slightly greater changes than the Reference Emissions Scenario during the operational phase. The differences become increasingly greater over longer timescales.

A credible maximum climate change scenario is not always cautious for long-term assessments. For example, the High Emissions Scenario is anticipated to result in inundation of the trenches and lower doses for the coastal erosion pathway than the Reference Emissions Scenario. The High Emissions Scenario requires use of all the fossil fuel reserves that are currently potentially technically and economically viable, and similar use of reserves that are not currently economically viable or non-conventional resources.

Given the above factors, we use the Reference Emissions Scenario as the basis for our reference calculations of the potential long-term impacts of the repository. The High and Low Emissions Scenarios are assessed in variant calculation cases.

2.10 Assessment Models

General

Our conceptual model describes the features which comprise the multi-barrier system, and the events and processes which lead to slow release of radionuclides through the multi-barrier system (gas and groundwater pathways) or breach of the barriers by coastal erosion or human intrusion. The key Features, Events and Processes (FEPs) are represented in the assessment models. Typically:

- Features are represented by discretising the model domain;
- Events are represented by calculations for a certain set of conditions, or by step changes in model boundary conditions or material properties;
- Processes are represented by mathematical models that calculate evolving boundary conditions, barrier properties and transport of materials, water, gas and associated radionuclides.

The key FEPs are identified from understanding of the underlying science, characterisation of the disposal system and its environment, previous assessments for the LLWR, assessments for other disposal facilities, industry guidance and best practice. We undertake audits of the FEPs included in our assessment models to build confidence we have identified the most relevant FEPs and treated them appropriately. This is discussed in Subsection 2.11.

For key parts of the disposal system or environment we develop detailed models that are intended to be as realistic as we can achieve within the limitations of understanding and available data. Such models use measured data as input, for calibration, or in their derivation. The prime examples are:

- modelling of the regional and local and groundwater flows including through the facility [50];
- modelling of the spatially variable biogeochemical evolution of the trenches and vaults [6];
- modelling of photosynthetic uptake of C-14-bearing gas into plant material for varied plant canopy types and conditions [26];
- use of extensive UK datasets to derive empirical relations between radon in soils and radon in dwellings [26].

For other parts of the disposal system, and for overall assessment models for each pathway, we may use models that include a higher degree of spatial and, or temporal approximation.

Generally, the input data are the result of spatial or temporal averaging of measured data. The models may omit some processes that are of lesser importance, or combine processes, e.g. representing sorption processes using a linear equilibrium sorption coefficient (K_d). The models are generally simplified, and some data may be cautiously selected.

Where the importance of a FEP is uncertain, this may be explored in a variant calculation case. Biases associated with simplified representations of FEPs or simplifying or cautious assumptions may also be explored in variant calculation cases. This approach is consistent with guidance from the IAEA BIOMASS programme [51].

Assessment model quality

The first requirement is that the assessment model includes the relevant features, events and processes, 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 post-PoA assessment models used for the 2026 ESC are mostly built on those developed for the 2011 ESC or updated assessments undertaken post-2011. 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 [23]. The only exception is the post-PoA radon assessment, where a completely new model has been developed in response to changes in understanding of the long-term performance of the cap geomembrane and the important transport processes.

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.

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. For both phases, draft model results have been scrutinised in workshops, leading to updates and improvements.

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.

GoldSim is a key piece of software used in the groundwater, gas, coastal erosion and human intrusion assessments to calculate radionuclide decay and ingrowth, migration, environmental concentrations, and doses and risks to people. ConnectFlow and PFLOTRAN are used to develop detailed underpinning models of groundwater flow and near-field biogeochemistry, respectively. In using the software, each assessment contractor has verified the suitability of the software and checked the implemented models for accuracy using appropriate procedures.

GoldSim is described below. ConnectFlow and PFLOTRAN are described in references [7] and [6], respectively.

GoldSim

The suitability of the GoldSim software [52] is evident based on its development and use over a period of around 30 years (see www.GoldSim.com). GoldSim has been used by and 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 program 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 [52] [53].

A described above, a range of 'top down' approaches have been used to ensure assessment model quality, including iterative development, testing and agreeing development approaches, sense check of the results, and building confidence using simpler supporting models. Detailed 'bottom up' checking is also undertaken. This includes checking the configuration of the assessment models, e.g. transfers between compartments, mathematical models and calculations entered into GoldSim, data input to GoldSim, and post-processing and reporting results.

Detailed checking of model configuration and parameterisation have been complemented by assessment model walkthroughs. Here the model developer(s) 'walk' the assessment team through the model implementation in the GoldSim software, explaining the basis for the configuration, modelling choices and the calculations the model undertakes. This helps to identify any misunderstandings in the conceptual model and key FEPs, opportunities for improved implementation, areas for reassurance testing, and to agree model simplifications and assumptions.

2.11 Assurance

We have developed assurance procedures to provide confidence in assessments [33]. In addition to procedures to build confidence in assessments models (Subsection 2.10), our procedures describe requirements for review, checking and record keeping.

Contractors supporting assessments are required to demonstrate their staff are Suitably Qualified and Experienced (SQEP) for the tasks they are undertaking. At the start of each assessment Phase, each contractor developed a Project Quality Plan (PQP) for their work, which we reviewed and approved.

At the end of Phase 1 we undertook audits of each assessment against the PQP. We identified areas of best practice and shared them between contractors for Phase 2.

Other factors contributing to assurance of the assessments include:

- close working with contractors and across the pathway assessments;
- ensuring consistency and common understanding through meetings and our data management procedures [34];
- stepwise development of assessments, with workshop reviews and discussion at each step;
- using the assessment results to understand the controls on performance, and where practicable using simpler calculations to confirm our understanding;
- review of the Phase 1 outputs by our independent peer review group (PRG);
- presentation of key Phase 1 outputs to the Environment Agency, and review of the associated reports by the Environment Agency.

2.12 FEP Audit

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.

To build additional confidence, the conceptual models have been audited against two FEPs lists: the site-specific FEP list derived for the 2002 Post-closure Safety Case (PCSC) [54], including the list of FEPs external to the system [55], and the generic NEA FEP list [56].

The objectives of the FEP audits are to build confidence all relevant FEPs have been considered, including associated uncertainties, and to help identify biases in the assessment models. The FEP audits are undertaken using a ‘top down’ approach. This involves reviewing the FEPs lists, recording the FEPs that are relevant to each assessment to show they have been considered, and noting where simplifying assumptions could introduce significant bias to assessment models.

This is a proportionate approach that provides significant benefit to the assessment. A ‘bottom up’ audit, which might involve identifying every potentially relevant FEP, and then describing how each FEP is treated, would require disproportionate resources, especially where existing assessments and models have been iteratively developed over several decades. With potentially hundreds of relevant FEPs to consider, a ‘bottom up’ approach is unlikely to provide clarity on which FEPs are most important and what the overall conceptual model is.

2.13 Endpoints and Calculation Methods

In our assessments we present a variety of calculated endpoints focused on illustrating the long-term performance of the LLWR and providing the output required for comparison to the guidance levels set by the GRA [19].

2.13.1 Radionuclide Concentrations

GRA paragraph 6.3.21 states:

> 'In setting up a risk assessment, in general the developer/operator should aim for data and assumptions that represent realistic or best estimates of the system behaviour. However, where the data do not support this approach or where the assessment can usefully be simplified, the developer/operator may choose some data and assumptions to be conservative as long as the requirements are still shown to be met.'

Consistent with this, where the necessary understanding and data permit, we aim to calculate concentrations of radionuclides in relevant media that are realistic. There are, however, significant uncertainties in many physical and chemical processes. These include uncertainties in the modes of radionuclide release from wastes of uncertain characteristics, sorption and retardation processes, and resuspension and deposition of sediments, that we only approximate by our models. In cases where the uncertainty is difficult to characterise or treat robustly, we aim to make assumptions and choose parameter values so they include a degree of caution and concentrations of contaminants in environmental media will not be underestimated.

We also apply our models to calculate environmental concentrations (and doses, see below) for alternative scenarios, conceptual models, model conditions and assumptions, and parameter values. That is, the calculated environmental concentrations are intended to be ‘realistic’ for the defined input conditions that are themselves uncertain.

2.13.2 Doses and Annual Doses

Based on the concentrations of radionuclides in accessible media we calculate radiation doses or annual radiation doses.

Effective radiation dose is a measure of the energy deposited by ionising radiation in a target, in this case humans, adjusted for the type of radiation and the organ or tissue being irradiated. It is an overall measure of the stochastic health risk from radiation. The quantity ‘annual dose’ (with units of Sv, mSv or μSv) is calculated as the sum of effective doses received from external sources in a year, together with the sum of committed effective doses incurred from intakes (ingestion and inhalation) during the same year. In some cases we calculate ‘dose from the event’, by which we mean the sum of effective doses from external sources during the event, plus the sum of committed effective doses from intakes arising from the event.

For external exposure, inhalation of dusts, gases or aerosols, and inadvertent ingestion of contaminated material, the calculation involves assumptions of factors such as dust loadings, breathing rate etc., but importantly it also involves dosimetric assumptions that relate absorbed dose rate to effective dose rate and intakes of radionuclide to committed effective doses. There are uncertainties in this conversion, especially related to dose per unit intake of radionuclides. We adopt the reference dose coefficients as derived by the International Commission on Radiological Protection (ICRP) and the US National Commission on Radiological Protection (NCRP) that are intended for use in such calculations, as published at the time of undertaking the assessments [57].

In calculating doses due to the consumption of contaminated foodstuffs, we generally do not take account of uncertainty in biological uptake values. Rather, we take values from standard tabulations such as developed by the IAEA for the purpose, or from review of alternative authoritative sources [57]. In the case of radionuclides of particular concern, specific side studies ensure the values adopted are appropriate to the modelled and local conditions, see reference [57]. For the analysis of the uptake of C-14-bearing gas into vegetation and human foodstuffs, a bespoke model has been developed [26].

GRA paragraph 6.3.30 states:

> *'Risk assessments will need to consider different groups of people that could be at risk of exposure (potentially exposed groups) in order to identify a person representative of those people at greatest risk at a given time. There is a range of possible doses that each group might receive and, for each dose, an assessed probability of their receiving that dose.'*

We identify potentially representative persons (PRPs, Subsection 2.8) that make reasonable maximum use of the potentially contaminated local environments [19]. This includes, for example, residing in the specific environment (if credible) or visiting those areas, using well water for drinking, and consuming local foodstuffs. When defining the behaviours of people who could potentially be exposed in the future, it is appropriate to consider both generic and local behaviour as the characteristics of the site and local behaviour could change.

It follows that the dose (and risk) values calculated are representative of only a few individuals who might have such cautiously defined habits, although there may be a larger group of individuals who partake in similar occupancy, activities or consumption, but not at such high rates. Doses and risks to the larger group of individuals who have similar behaviours but at lower levels of intensity will be lower than those calculated

Given that we identify PRPs with occupancy and consumption rates towards the upper end of the range of observed or possible behaviours, we do not explore uncertainty in habits in our assessment calculations.

2.13.3 Risk and Annual Risk

Radiation risk is the probability per unit time that an individual will suffer a serious radiation-induced health effect as a result of the presence of a radiation source, for example, a disposal facility. In this context, a serious radiation-induced health effect is a fatal cancer or a severe hereditary defect. Radiation risk can only be assessed and not measured.

The concept of radiological risk is applied by ICRP to potential exposure situations, i.e. situations in which an exposure may or may not occur, or the dose that could be received is uncertain [58].

GRA paragraph 6.3.13 states:

> *'The assessed radiological risk associated with a potential exposure situation corresponds to the product of the estimated effective dose that could be received, the estimated probability (as a quantified uncertainty …) that this dose will be received and the estimated probability that detriment would occur as a consequence to the person exposed (see para. 6.3.15). For comparison with the risk guidance level, assessed risks must be summed over all situations that could give rise to exposure of the same person to radiation.'*

The GRA further states in paragraph 6.3.15:

> *'For annual effective doses below 100 mSv, HPA (see below) recommends the use of the detriment-adjusted risk coefficient and that a rounded value of 0.06 per Sv be used for waste management assessments.'*

and in paragraph 6.3.17:

> *'Making use of the risk coefficient of 0.06 per Sv, a dose can be calculated that gives rise to a risk of 10^{-6} per year. This calculated dose is around 20 µSv/year and represents the situation where the dose has been received, that is to say the probability of receiving the dose is one.'*

In our assessments, we calculate the annual dose for a given assessment case (see Subsection 2.8). If this dose is multiplied by the risk coefficient recommended by the Health Protection Agency (HPA) of 0.06 per Sv, the product is the 'conditional risk' for the case, i.e. the risk conditional on the various assumptions and parameter values of the case, or the risk assuming a probability of one for the case. GRA paragraph 6.3.17, above, indicates that for this circumstance, we may compare to an annual dose of 20 µSv.

In some of our cases we expect exposures to occur, so we calculate conditional risks. For example, we expect people to occupy the coast in front of the eroding repository and be exposed to radionuclides on the coast. However, in other cases it is uncertain whether exposures would occur, i.e. the probability of exposure is less than one, and in some cases the probability of exposure is low. We have calculated the probability of abstraction of water from a well for the groundwater pathway, and the probabilities of a range of exposure situations for the gas pathway. Therefore, in cases where we have calculated the probability of exposure, this is included in the calculated risks. This is further described in the following subsections.

In addition to people and non-human biota [16], groundwater is also a receptor. By demonstrating that the risks are consistent with Requirement R6 of the GRA, and the associated risk guidance level, we also show that the potential impacts on groundwater are low. Consistent with Environment Agency guidance [20] [21] this contributes to demonstrating that the repository meets the groundwater activity provisions of the environmental permitting regulations. The other lines of evidence required by the supplementary guidance are provided by references [14] [10] [15] [3] [7] [9].

2.13.4 Assessing Against the Risk and Dose Guidance Levels

2.13.4.1 Alternative Methods

As stated by GRA paragraph 6.3.22:

> *'In cases where the hazard presented by the waste warrants a detailed assessment of risks, we shall expect a probability distribution of dose to be one of the outputs from each risk assessment that the developer/operator undertakes. The probability distribution will cover the range of possible doses that a person representative of each potentially exposed group may receive and will provide the probability that this person receives any given dose.'*

One method of achieving this is to carry out a PRA. In PRA, the assessment model is run many times to simulate multiple possible realisations for the future performance of the disposal system for input parameters values sampled from appropriately derived probability density functions (PDFs). In this case, annual risk is calculated as the mean of annual dose as a function of time, multiplied by the risk coefficient multiplied by the probability of exposure. This method requires:

- confidence that the model can adequately capture all future evolutions and relevant uncertainties concerning the release pathway and exposure situation;
- confidence that PDFs can be derived that properly capture the uncertainties that each sampled parameter is designed to represent.

This can be very challenging in input data requirements and places practical constraints on the models applied, e.g. demanding simplifications in order that the model can be run probabilistically.

An alternative method based on deterministic analysis is to:

- run models for a range of separately defined assessment cases that span uncertainty in starting conditions and evolution, to yield a number of dose or dose vs time results conditional on each specified case definition;
- from the ensemble of results, define the key uncertainties and choose a case, or a weighted sum of cases, that is judged to best represent a reasonably expected result for the pathway and scenario.

This method may be more appropriate where there are discontinuities or step changes in outcomes that are not easily captured in a single model or simulation specification, or where probabilities cannot be confidently assigned to key parameters or conditions.

Approach in the 2026 ESC

As described in Subsection 2.8, we assess the risks for reference cases, which include our best estimate conceptual models and parameter values, and variant cases which explore uncertainties. Variant calculations are also used to understand the role and importance of individual barriers.

Many of our cases are not amenable to PRA because scenario and conceptual model uncertainties are at least as important as parameter uncertainty, and because it is not practicable or possible to collect sufficient data to develop robust parameter distribution functions. In some cases where systems are simple or linear, the additional insights offered by PRA are insignificant. In such cases, we follow the method based on deterministic analysis, described in the previous subsection. Where appropriate, we include the probability of exposure in our deterministic risk calculations. We use PRA where it is proportionate to do so, and sufficient data are available to develop robust parameter distributions.

The approach for each pathway assessment is described below.

Groundwater pathway

We evaluate deterministic reference and variant cases for the groundwater pathway. We have also evaluated a probabilistic reference case.

The variant cases investigate:

- uncertainties in the conceptualisation of the LLWR vault near field;
- uncertainty in evolution of the site (e.g. coastal erosion, sea-level rise);
- uncertainty in the evolution of the site engineering (e.g. cap, vault bases);
- other uncertainties (e.g. groundwater flow rates, effectiveness of chemical barriers).

We have also undertaken variant calculations to understand biases in the assessment models, barrier performance and the contributions of barriers to overall system performance.

Uncertainties in the key parameters for the groundwater pathway are sufficiently well characterised that it is possible to define parameter distributions and evaluate a probabilistic calculation for the reference case. Deterministic and probabilistic calculation results from earlier assessments show that parameter uncertainties sufficiently affect the calculated risks to justify expending the substantial resources needed to develop the robust parameter distributions required to meaningfully explore the effects of combinations of uncertainties.

In all cases, calculations for the marine, estuary and stream biosphere pathways assume that if there are radionuclide releases to the pathway then people are exposed, i.e. the probability of exposure is one. Therefore, conditional risks (i.e. risks conditional on the assumed presence of a PRP) are calculated for comparison against the risk guidance level.

A well could be located anywhere in the contaminated area between the repository and the coast, and it is uncertain whether a well would exist in the contaminated area. Therefore, for each deterministic calculation, and each realisation of the probabilistic calculation, the calculated risk for the well biosphere pathway depends on:

- the expectation values of radionuclide concentrations in the contaminated area;
- the annualised probability that a well exists in the contaminated area.

The expectation value of risk is calculated for comparison against the risk guidance level.

Gas pathway

We evaluate a reference case, and variant cases which explore conceptual model and parameter uncertainties, biases in the assessment models, and the roles of barriers. All cases are evaluated deterministically because the conceptual model uncertainties have bigger impacts on calculated risks than parameter uncertainties. This is demonstrated by the results of the cases that have been evaluated. Conceptual uncertainties cannot meaningfully be included in a probabilistic calculation.

The calculations consider spatial variations in discharges of radioactive gases to the surface of the cap, due to spatial variations in:

- the radionuclide inventory;
- radioactive gas generation rates;
- the rate of generation of bulk gases, which act as a carrier for radon gas.

People living on the cap would have the highest risks from radioactive gases. People may be living in a house, or in a house with a smallholding. People could be living anywhere on the cap, so risk depends on the expectation value of radioactive gas fluxes to the surface of the cap. Some potential exposure situations are more likely to occur than others. Therefore, the risk also depends on the probability of the exposure situation. The expectation value of risk is calculated for comparison against the risk guidance level.

Coastal erosion pathway

One of the key uncertainties for the coastal erosion pathway is future greenhouse gas emissions. This will be a consequence of sociopolitical decisions, so it is not amenable to probabilistic assessment. Other key uncertainties such as the relationship between the amount of sea-level rise and response of the coastal system are also not amenable to probabilistic assessment. Therefore, we evaluate the calculation cases deterministically.

People are expected to be exposed over a specific time period to radioactivity released by coastal erosion of the repository. Therefore, for the coastal erosion pathway the conditional risk is calculated, i.e. the probability of exposure is one.

We evaluate a reference case, and variant cases, which explore scenario, conceptual model and parameter uncertainties, biases in the assessment models, and the roles of barriers.

Human intrusion pathway

For human intrusion cases, we calculate doses for comparison to the dose guidance range of around 3 mSv y^{-1} to around 20 mSv y^{-1}, as specified in GRA paragraph 6.3.36. Values towards the lower end of this range are applicable to assessed exposures continuing over a period of years, while values towards the upper end of the range are applicable to assessed exposures that are only short term. We calculate dose from the event for short-term exposures, such as received by persons directly engaged in the event, and annual doses to persons exposed over longer times to environmental contamination arising from the event.

Given the uncertainties in future human actions, we use stylised assessment models [19] to assess potential human intrusion events. These stylised models are plausible and internally consistent. We use simple, logical conceptual models, supported by evidence and data from real world examples. Although the conceptual models and assessment model parameter values are supported by real world examples, the stylised models inherently include cautious assumptions.

There are conceptual model and parameter uncertainties for each event. However, uncertainty in future human actions is far greater. Given this, and the stylised and cautious nature of the assessment models, we do not quantitatively assess variant cases with alternative conceptual models or parameter values, and do not undertake probabilistic analysis. However, we do assess variant cases which explore alternative scenarios and institutional control assumptions. These variant cases explore sensitivity of the calculated doses to the timing when intrusion events could occur.

2.13.4.2 Additional Guidance on Erosion of the Repository

Recognising that waste items disposed to the repository would be exposed by coastal erosion (and possibly also human intrusion), in 2014 the Environment Agency issued advice to environmental assessors on disposal of discrete items to the LLWR [59]. This advice supplements the GRA [19].

The advice to assessors differentiates particles and visually identifiable objects, and exposure through casual curiosity and deliberate searches. The guidance identifies appropriate tests of significance for the radiological burden associated with particles and discrete items:

- Use of the risk guidance level for random encounters with particles.
- Use of the dose guidance level for encounters with discrete items through casual curiosity and deliberate searches.

In our assessments we define a discrete item as:

> *'A distinct item of waste that, by its characteristics, is recognisable as unusual or not of natural origin and could be a focus of interest, out of curiosity or potential for recovery and recycling/re-use of materials should the waste item be exposed after repository closure.'*

Our assessments include the potential radiological impacts from items with relatively high activity (Subsection 2.6). We discussed the appropriate test of significance with the Environment Agency in the context of developing WAC for waste items that might remain partially intact when exposed by coastal erosion, and therefore potentially become discrete items. Reference [60] summaries our discussions with the Environment Agency. A précis of the discussion described by reference [60] follows.

We have discussed issues related to the assessment of discrete items and particles with the Environment Agency in support of their review of our ESC. Stemming from these discussions the Environment Agency has issued 'Advice to Environment Agency Assessors' on these matters [59]. *The 'Advice' and our discussion with the Environment Agency staff indicates, amongst other things, that the Environment Agency would accept an assessment of encounter with discrete 'visually identifiable objects' against a significance test of an effective dose guidance level in the range of around 3 mSv/year to around 20 mSv/year. This is by analogy with the GRA Requirement R7 for assessment of human intrusion, which is to account for situations that 'cannot reliably be assessed in terms of a numerical value of probability'. We appreciate the Environment Agency 'Advice to Assessors' and associated discussions. Taking account of this advice and discussions, our view is as follows.*

Recognising that coastal erosion represents the expected natural evolution scenario[4] *(probability=1) and that the Environment Agency's 'Advice to Assessors' is complementary to the GRA, we conclude that it is cautious to assess casual and short duration encounters with discrete items against an effective dose of 20 µSv and this will be protective of the average beach user that also happens to encounter and briefly examine unusual items on the beach. The individual may examine or remain in close proximity with a number of such items over the period of a year, but only one or a few will bear radionuclides at higher levels, which will be limited by the Discrete Items Limits. The derivation of limits to control such 'casual*

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[4] Superseded by the Low, Reference and High emissions scenarios for the 2026 ESC.

inspection' is considered within the assessment of coastal erosion, consistent with 'expected' behaviours and comparison with the risk guidance level.[5]

On the other hand, the case of individuals deliberately seeking out, collecting, taking away or attempting to disrupt discrete items should be judged against the effective dose guidance range of around 3 mSv/year to around 20 mSv/year. The 3 mSv/year would apply to an individual that was carrying out such activities over a number of years and 20 mSv/year would apply to cases where the activity extended over about a year or less.

Whilst the advice to assessors is not formal regulatory guidance, it is consistent with the GRR, which was issued after we developed our approach. GRR requirements A4.90 and A4.91 state that the lower end of the dose guidance level should be used to compare results of assessments of interactions with local concentrations of substances or visually identifiable articles [28].

Therefore, the ESC takes a cautiously realistic approach by assuming casual encounters with discrete items will happen. We have developed discrete item WAC to limit potential doses from casual and short duration encounters with discrete items against an effective dose of 20 µSv, i.e. the dose equivalent to the risk guidance level assuming exposure occurs. Potential doses from deliberately seeking out discrete items are assessed against the dose guidance level.

We consider that this remains an appropriate and cautious approach to applying the risk guidance level and dose guidance level. Therefore, this approach is retained for the 2026 ESC.

The ESC considers adults, children and infants making recreational use of the coast, and adults making occupational use of the coast. Information from local and national habits surveys, knowledge and observations of activities undertaken on beaches and coasts, is used to identify activities that could lead to exposure to radionuclides in the coastal environment, particles, low-activity sealed sources and discrete items (sealed sources are identified as a specific category of discrete item).

Making recreational or occupational use of the coast would result in exposure to environmental radioactivity resulting from erosion of the repository. The types of activities that could lead to exposure to discrete items have been classified as casual encounters or deliberately seeking out discrete items (Table 2.1).

The above arguments are extended to deliberate searches to recover (scavenge) waste exposed by coastal erosion. The materials most likely to attract deliberate searches for recovery are durable materials such as resilient metals, rather than degraded organic waste, corroded thin steels, degraded grout, soils, pieces of concrete glass, etc. Clean materials from the cap are more likely to be attractive for reuse in landscaping or building than mixed contaminated waste materials.

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[5] This does not mean it is included within the main assessment model, but it is part of the wider assessment.

Table 2.1: Application of the risk and dose guidance level to scavenging waste and the types of activities leading to exposure to discrete items

Type of activity	Exposure situation	Classification	Test of Significance	Assessment
Occupying the coast to undertake a recreational activity, e.g. dog walking, angling, horse riding, playing, crabbing, paddling	Inadvertent close proximity to a discrete item, stopping to look at a discrete item, picking up a small discrete item to examine it, close inspection of a discrete item that is too large to pick up, stepping on a discrete item, clambering on a large discrete item.	Casual encounter	Effective dose of 20 μSv, i.e. the dose equivalent to the risk guidance level assuming exposure occurs	Costal erosion (Section 7)
Occupying the coast to undertake an occupational activity, e.g. fishing, bait digging				
Occupying the coast for a picnic, sunbathing, building sandcastles	As above, plus using a large discrete item (e.g. fuel flask) as a windbreak or to support a windbreak.			
Beachcombing	Picking up discrete items, carrying them in a bag or pocket, taking discrete items home, doses to other people if the discrete items are passed or sold on.	Deliberately seeking discrete items and materials	Dose guidance level	Human intrusion (Section 8)
Metal detecting				
Archaeology				
Material recovery and reuse	Opportunistic or occupational recovery of materials. Reuse with or without processing.			

2.14 Managing Future Disposals

We control the radionuclide inventory for disposal to ensure compliance with regulatory dose and risk criteria. Where required to support appropriate system performance, we also specify requirements for the physical and chemical properties of the waste.

Since the 2011 ESC we have learned more about the uncertainty in the forward inventory because we have started to track actual disposals and compare them with forecast disposals. We are now more aware of the uncertainty in inventory projections, which is greater than previously appreciated.

In response, the 2026 ESC places more emphasis on calculating radiological capacities, activity limits, and other controls that can be included in the WAC and used to manage the repository, to ensure the potential impacts from actual disposals remain consistent with the GRA. The '*Implementation*' report [17] develops capacities, limits and other controls from the results of our assessments, including the results presented in this report.

3 The LLWR and its Evolution

Other reports provide detailed descriptions of the characteristics and understanding of the:

- LLWR site and its history [3];
- inventory already disposed or emplaced and future inventory [4];
- engineering design for future vaults and closure [5];
- near-field system and its evolution [6];
- hydrogeological system and its evolution [7];
- natural evolution of the site, including the impacts of climate change, sea-level rise and coastal erosion [8].

The process by which an optimised SDP has been arrived at is described in reference [10]. Drawing on these reports, this section provides an overview the LLWR facility and its environs, the SDP, the radioactive inventory, and an integrated description of the evolution of the LLWR and its environment, at a sufficient level of detail to understand the motivation for the scenarios and cases that are assessed in subsequent sections of the report.

Descriptions of some aspects, notably the near field and hydrogeological systems and their evolution, are not included here but rather discussed within assessment sections.

3.1 History and Description of the LLWR as it Exists Today

The site of the LLWR was first developed in 1940 as a Royal Ordnance Factory for the production of TNT. Ownership later passed to United Kingdom Atomic Energy Authority, which in 1957 was granted planning consent for the disposal of waste in the northern 40 hectares of the site. The first Certificate of Authorisation for disposal of LLW was granted in 1958 under the terms of the Atomic Energy Act 1954, and disposal operations commenced in 1959. Ownership and responsibility for the site was transferred to British Nuclear Fuels Limited when the company was formed in 1971, and the site became a part of the NDA's estate when that body was established in 2005. Between 2008 and 2021, the management of the site was contracted to UK Nuclear Waste Management Ltd. The site is now operated by Nuclear Waste Services Limited, a wholly-owned subsidiary division of the NDA.

The LLWR receives wastes from a range of consignors, including nuclear power stations, defence establishments, general industry, hospitals, universities and from the clean-up of historically contaminated sites and redundant fuel cycle, isotope manufacturing and research facilities.

For the first thirty-six years of operation, disposals were by tipping of drummed, bagged and loose wastes into successive trenches (Figure 3.1) within the 'consented area' at the northern end of the site. The first trench followed the course of a railway cutting associated with the Royal Ordnance Factory. Subsequently, five wider and deeper trenches were excavated parallel to, and on either side of, Trench 1 such that their bases should lie within low-permeability clay at a depth of 5 to 8 m below ground level. In the case of the later trenches at least, if natural clay was locally absent, bentonite clay was rotovated into the bases to reduce the permeability of the trench bases. A final trench, Trench 7, of irregular shape was excavated to fully use the site area towards its north-eastern boundary. All trenches have a north to south fall, to facilitate the collection of leachate at the southern end of each trench, where it is diverted to an interceptor drain.

Figure 3.1: Photograph of trench disposal operations in the 1980s

From the early years, the disposed waste was covered by soil at the end of each working day. Periodically, a hardcore layer was placed to facilitate tipping operations. Trench 7 was closed in 1995.

From 1987 onwards, disposal operations were upgraded. Remedial work was also carried out on the trenches; this included installation of a low-permeability cut-off wall (to limit lateral movements of groundwater and radionuclides) to the north and east of the trenches, interim capping of the filled trenches, and upgrading of the leachate drainage system to allow for discharge directly to sea (rather than surface water) through the Marine Pipeline via the Marine Holding Tanks.

An engineered, concrete disposal vault was constructed, Vault 8, which allowed the orderly emplacement of containerised waste within an engineered concrete structure according to modern disposal standards (Figure 3.2). The emphasis of the Vault 8 design was largely on operational aspects of waste emplacement and storage. The vault has surface water drains to collect rainwater from the surface of the base slab, while an under-slab drainage blanket and perimeter drains relieve the possible build-up of hydrostatic pressure beneath the floor slab and collect any leakage through the slab. Vault 8 commenced operation in 1988, the first seven years of its operation overlapping with the operation of Trench 7, to use up the available capacity in the trench.

Figure 3.2: Vault Disposal

Wastes were disposed in Vault 8, and are now disposed in a second vault, Vault 9, in steel ISO (International Organization for Standardization) containers of different heights. Prior to disposal, the wastes are grouted on site at the Drigg Grouting Facility, which has been operational since 1994 (Figure 3.3). Waste containers consigned to the repository before this date were grouted once the Drigg Grouting Facility was operational. Sometimes wastes have been disposed in different waste containers or directly disposed to the vaults and grouted in situ. Sometimes, containers have been accepted that are too heavy to be fully grouted and then emplaced. The grouting of such containers was completed after emplacement.

Figure 3.3: Drigg Grouting Facility

The introduction in 1995 of waste monitoring and high-force compaction at the Waste Monitoring and Compaction plant (WAMAC) on the Sellafield site significantly improved the waste loading of the ISO containers. Some containers received before 1995 were sent to the Waste Monitoring and Compaction Plant for the waste to be compacted.

Vault 8 was mostly filled to its original planned capacity by 2009. A small area at the northern end of the central bay of Vault 8 was left unfilled to allow vehicular access to the vault via a ramp and to allow disposal of any overweight or un-containerised waste. The central bay of Vault 8 was also used to store ISO containers stacked up to an additional height of two half-height ISO containers, above the disposed containers stacked up to a height of four half-height ISO containers. These higher-stacked containers were placed in Vault 8 before Vault 9 became available. It has now been decided to leave these higher-stacked containers in place in Vault 8 for disposal. The movement of 171 containers from Vault 9 to the unfilled area of Vault 8 was completed in January 2025. The movement to Vault 8 of building rubble from one of the waste stores on site, which has been stored in Vault 9, is underway. The rubble is being used for infill of Vault 8, including the access ramp. Recent disposals to Vault 8 include additional LLW WAGR Boxes from Sellafield in 2019. LLW Treated Radwaste Store drums from Winfrith were disposed in a gap on the eastern side of Vault 8 between 2022 and 2024.

Construction of Vault 9 started in 2008 and was completed in December 2010. At the time, it was only permitted and only had planning permission for LLW storage. The vault had a ‘bathtub’ design, with a base and walls containing layers of bentonite-enhanced sand. The bathtub design aspect was not completed during construction. Design optioneering during the development of the 2011 ESC preferred a design with lower sealed walls and the bentonite-enhanced sand was not placed to the full height of the vault walls.

Following the 2015 Permit revision and subsequent planning permission in 2016, Vault 9 could be used for disposal. The vault contains approximately 2,000 disposal containers. Most wastes are now received within half-height or third-height ISO containers. The containers are first stored in Vault 9 and then grouted in campaigns. The ISO waste containers in Vault 9 have not been placed in their final disposal positions. The intention is to move the containers to the northern end of the vault and extend the final cap over them at the appropriate time. Some LLW WAGR Boxes have been emplaced in their intended final disposal positions at the northern end of the vault. Vault 9 has also been used to store demolition wastes, pending disposal in Vault 8.

The volume of wastes received in recent years has greatly diminished due to the introduction of the waste hierarchy into the management of LLW and the consequent diversion of wastes down other waste management routes. To date, no un-containerised waste or overweight containers have been disposed in Vault 9.

In the period 2015 to 2018 the security fence around the site was upgraded. It now does not enclose all the land at the southern end of the site. The NDA has also taken out a 999-year lease on the land between the site and coast, providing control over the use of the land.

Ten magazines were built as part of the Royal Ordnance Factory to store the manufactured TNT before shipment off site. These were subsequently used to store radioactive waste including ILW. Five of the magazines were emptied in the early 1980s and were subsequently demolished in 1995. The remaining magazines have now been emptied and partially demolished. The covering soil has been removed. The intention is to demolish the remaining concrete and brick constructions and use the material, along with the soil, in the construction of the final cap over the repository.

Preliminary civil engineering works for the initiation of final capping of the disposal facility were undertaken between 2019 and 2023. Stockpile areas, a haul road and run-off management infrastructure were constructed. Replacement of the interim cap over the southern area of the disposal trenches, which will not be covered by the final cap for some decades, began in 2024 to reduce infiltration into the trenches.

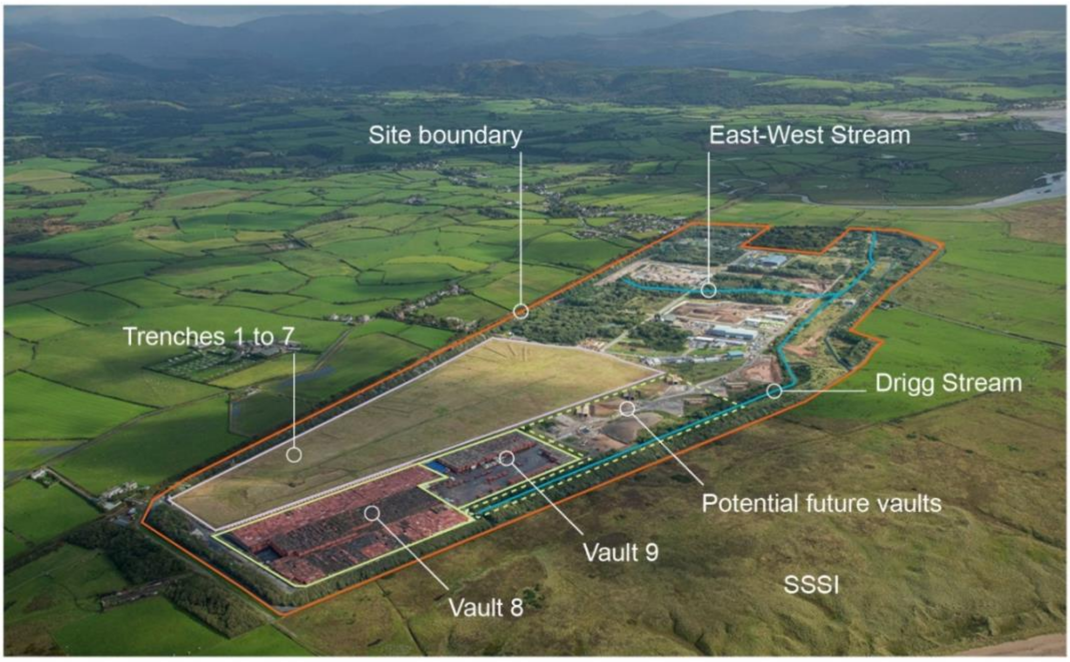

Figure 3.4: The LLWR site

Further information on the site is given in the '*Site History and Description*' report [3]. This includes information on how the regulation of the site has evolved.

3.2 Environmental Context of the LLWR

The LLWR is located on the West Cumbrian coastal plain, close to the village of Drigg and approximately five kilometres south-east of Sellafield. A map of the locality is shown in Figure 3.5 and the immediate environs in Figure 3.6. Apart from nearby Sellafield, the area is predominantly rural. The site is mainly surrounded by grazing land, but some cereal crops are grown in fields to the east.

The area along the coast adjacent to the site is designated as a Site of Special Scientific Interest (SSSI), known as the Drigg Coast SSSI. The area is also a Special Area of Conservation designated under the European Habitats Directive. The inshore area along the coast is also a Marine Conservation Zone. Along the north-eastern boundary is the Carlisle to Barrow-in-Furness railway line, sidings from which enter the site for the delivery of waste containers and other items and materials. The main north-south road through West Cumbria, the A595, runs about two kilometres to the east of the site. The Ravenglass Estuary lies to the south. The Cumbrian mountains rise further to the east. The LLWR lies outside the Lake District National Park, which is bounded by the A595 and the Ravenglass Estuary.

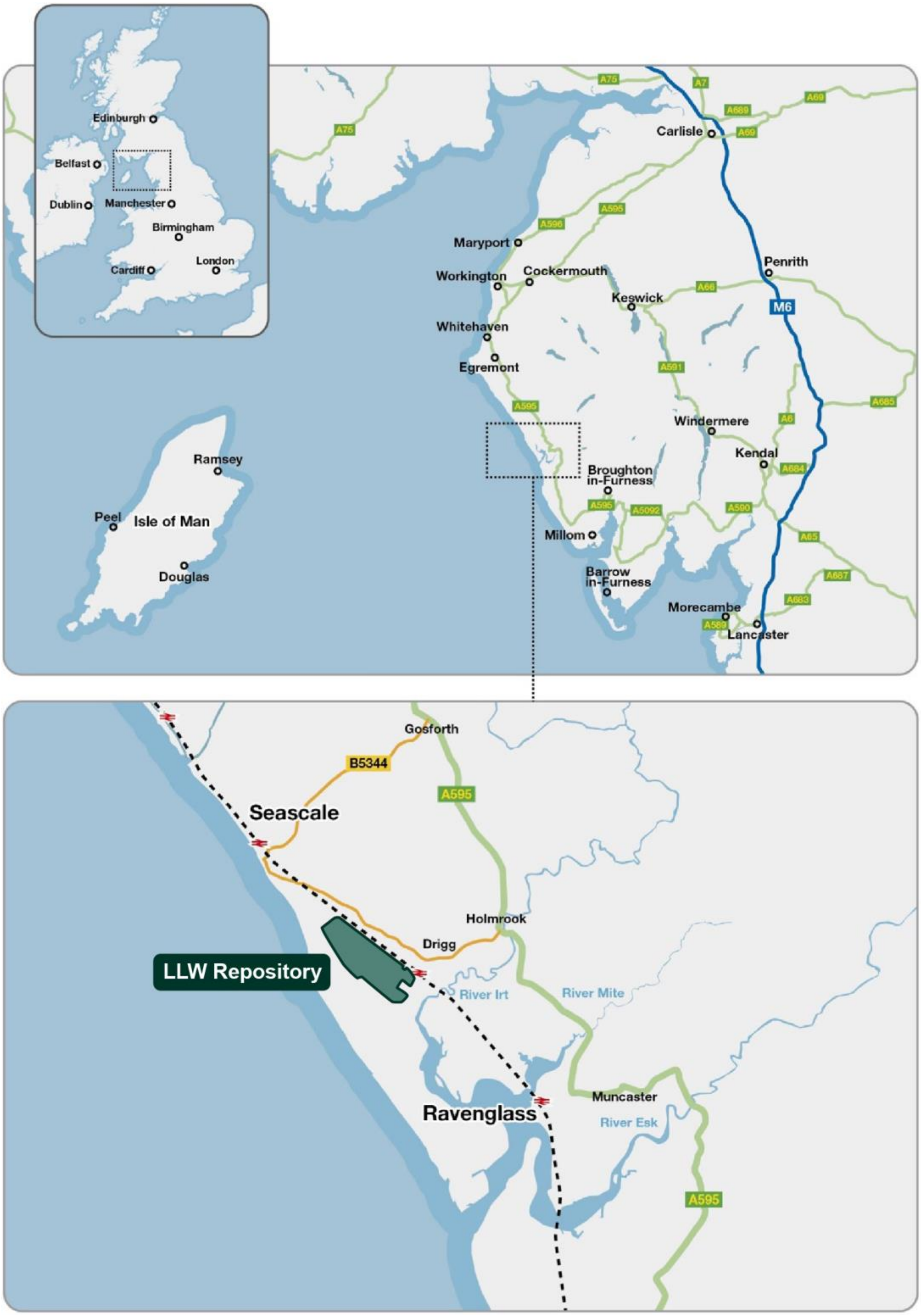

Figure 3.5 Map of LLWR location

Figure 3.6 The LLWR site and its immediate environs

The LLWR site is about two kilometres long and half a kilometre wide and lies on a north-west to south-east axis. A boundary fence, designed to prevent unauthorised access, encloses most of the site. The northern half of the site is used for waste disposal. The south-western boundary of the northern area of the site borders the SSSI. The height of the site varies from 20 m above Ordnance Datum (OD) to the north-east and west of the site, to less than 5 m above 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. The surface of the interim cap that covers the trench area is around 25 m above OD.

The Drigg Stream rises immediately to the south of Vault 8 and flows through the site roughly parallel with the western site boundary. Towards the centre of the site, the Drigg Stream is joined by the East-West Stream, which originates off the site to the north-east, draining farmland and taking water from a drain in the base of the Railway cutting. The Drigg Stream leaves the site to the south and discharges into the River Irt, which is tidal at that point. The Irt forms the northern arm of the Ravenglass Estuary, comprising also the rivers Mite and Esk, which discharges to the sea opposite the village of Ravenglass.

Further information on the environmental context of the LLWR can be found in the '*Site History and Description*' report [3].

The geological structure in the region of the LLWR consists of Quaternary age deposits (up to 2.6 million years old) overlying older bedrock. Quaternary deposits at the LLWR site are a result of complex glacial processes, which were responsible for the deposition of a sequence of deposits of clay, sands and gravels up to 70 m thick. The Quaternary deposits overly Triassic Ormskirk Sandstone (around 240 million years old) in the vicinity of the LLWR site.

A schematic representation of the hydrogeological conceptual model is shown in Figure 3.7. The Upper Groundwater occurs in the shallower Quaternary sediments. Regional Groundwater occurs within the deeper Quaternary sediments and the underlying bedrock. There is a high (approximately unit) vertical gradient in the Upper Groundwater, with water draining downwards into the Regional Groundwater, although in places flow has a significant horizontal component. Flow in the Upper Groundwater has an upwards component where it discharges to streams. Flow is sub-horizontal in the Regional Groundwater, driven by a weak horizontal gradient that is generally perpendicular to the coastline, i.e. approximately from north-east to south-west. In areas where the upper part of the bedrock is at relatively shallow depth, flow in the upper part of the bedrock makes a significant contribution to the regional groundwater flow. Flow in the Regional Groundwater discharges into the sea.

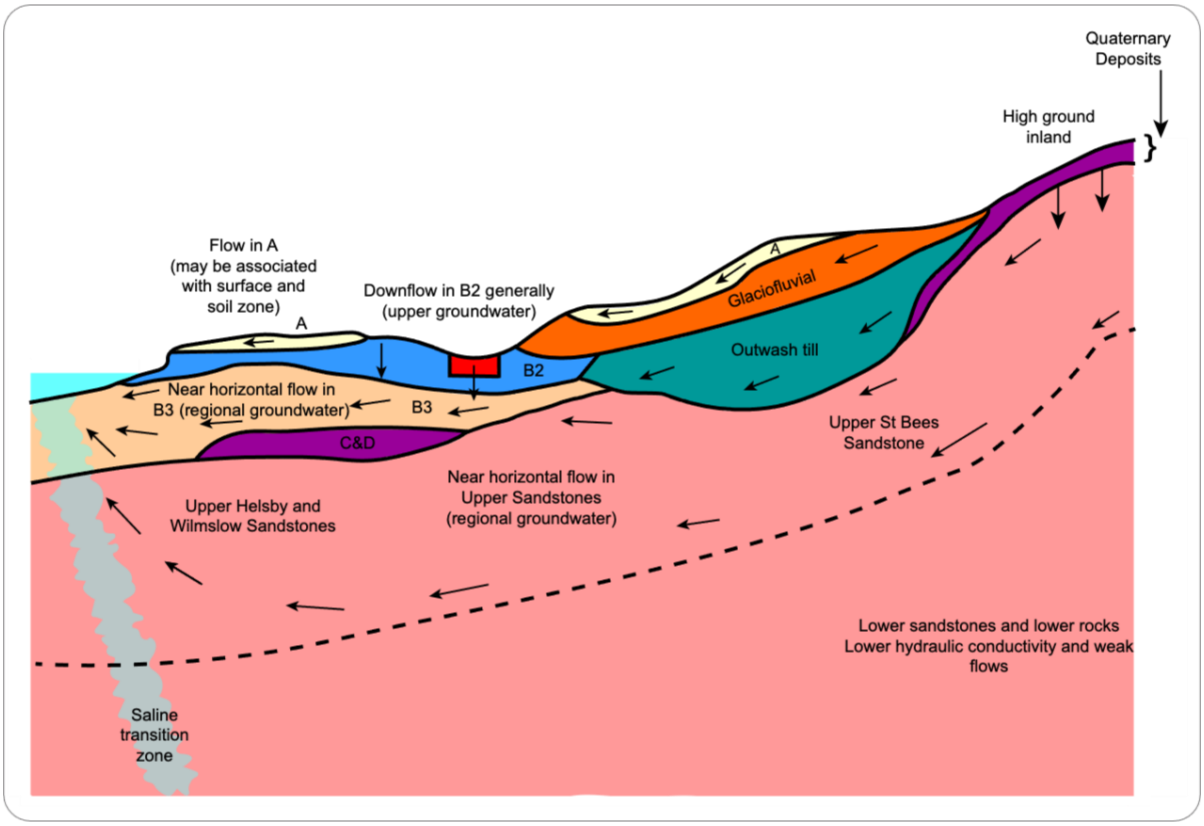

Figure 3.7 Hydrogeological conceptual model (schematic east-west section)

Further information on the geology and hydrogeology in the vicinity of the site is given in the '*Hydrogeology*' report [7].

At its north-western corner, the LLWR disposal area is only about 350 m from the present-day coastline, and the site is vulnerable to sea-level rise and coastal erosion. It is not believed that this vulnerability was a consideration when the site was first selected for waste disposal. Now, consideration of sea-level rise and coastal processes and assessment of their effects are important aspects of the ESC. Based on qualitative and quantitative evidence, including modelling studies, we have concluded that the site will start to be eroded on a timescale of several hundred to a few thousand years, with consequent disruption of the repository [8].

While this situation may appear unusual for a radioactive waste repository, we observe that in the long term all near-surface disposal facilities are vulnerable to disruption by natural erosion processes, human actions or combinations of natural and human events. This is accounted for by setting limits on the types and activity of waste that may be disposed to a near-surface facility to ensure that the impacts are consistent with regulatory guidance levels at the time of disruption. In the case of the LLWR, the nature and timing of coastal erosion and resulting limits means that there is some LLW that we are unable to dispose because of the quantities of longer-lived radionuclides it contains. Conversely, some ILW would be safe to dispose because of the mix and quantities of radionuclides it contains. Further information on the current understanding of the future natural evolution of climate and landscape in the vicinity of the LLWR is given in the '*Site Evolution*' report [8].

3.3 Site Development Plan

This subsection presents key features of our optimised SDP for the management of the LLWR. Development of the disposal facility, according to the SDP summarised in this subsection, would allow the site to continue to operate as the primary destination for disposal of LLW in the UK until 2135, based on current assumptions about future LLW arisings and application of waste segregation, diversion and treatment.

The SDP for the use of the site for the disposal of LLW is summarised as follows.

- No intrusive remediation of the trenches is planned, however, active leachate management, the closure engineering that will be constructed, and renewal of the interim trench cap that is underway, will optimise the long-term environmental performance of both the trenches and vaults.
- Waste will only be accepted for disposal and emplaced in the vaults consistent with the requirements of the ESC and our Permit to dispose of radioactive waste. Individual waste consignments will only be accepted for disposal where they meet the WAC or an assessed and agreed variation to the WAC. The quantities of wastes disposed will not exceed the assessed radiological, non-radiological and volumetric capacities of the facility. The capacity of the repository will be managed to ensure that it is used optimally. Waste will only be accepted for disposal when it has been shown that it is appropriate to dispose of it in an engineered vault, for example, the disposal of VLLW will be avoided unless disposal at the LLWR has been shown to be optimal (best available technology, BAT).
- Engineered vaults will be constructed to allow disposal of those wastes requiring vault disposal. The vaults will be constructed beside and to the south of Vault 9 in the area adjacent to the western side of the trenches.

- A final cap will be progressively constructed over the vaults and trenches. Eventually, the whole area of trenches and vaults will be covered by a single, gently domed, low permeability engineered cap, designed for stability and resistance to erosion and presenting acceptable visual impact. Suitable long-term vegetation cover will be established on the cap area and periphery.
- A passive gas venting system will be incorporated into the final cap to provide confidence that differential pressures will not threaten the performance of the cap as a barrier to infiltration. Final decisions on the vent design and whether the vent will be closed before the end of active institutional control will be made later, noting that the venting approach does not need to be part of the construction design for the first strip of the cap.
- The vaults will step down in the southerly direction following the natural slope of the site and containerised waste will be stacked in the vaults, utilising as much as possible of the profile volume below the engineered cap.
- Vault areas will be constructed as needed, filled, closed and capped progressively, at the same time capping over the adjacent strip of the trench area.
- In Vault 8 we will not pursue higher stacking beyond current stack heights. In the northern portion of Vaults 9 and 9a, which will be filled with ISO containers of current designs, our baseline assumption is that containers will be stacked to a maximum height of five containers. Stacking beyond that height will require further substantiation. Containers in Vault 8, and the northern portions of Vaults 9 and 9a will be surcharged prior to cap installation to ensure that the wastes will provide a stable base for cap installation. While this will reduce the effectiveness of the barrier the containers provide, this is more than compensated for by increased confidence in long-term cap performance.
- We will develop a new stronger ISO container design for wastes not yet committed to a disposal container. Stronger ISO containers will be higher stacked and will be disposed to the southern parts of Vaults 9 and 9a and we expect will comprise the majority of disposals in Vaults 10, 11 and 12. The top-most container of each stack will be fitted with a concrete container protection unit designed to protect the vulnerable container lid from the loads that will be imposed during and after cap installation. These containers will not be damaged during, or after, cap installation and will therefore provide an effective barrier to contaminant transport. The units will also provide some protection from precipitation.
- Interim storage warehouses will be used to protect waste containers in the vaults from local environmental conditions prior to emplacement of the containers in their final disposal positions and construction of the final cap over them.
- During operations, leachate from the trenches and rainwater run-off from the open vaults will continue to be managed by collection, monitoring and controlled discharge

to sea via the Marine Pipeline, subject to the requirements of our Permit. Progressive capping and replacement of the southern area of the interim trench cap will reduce infiltration to the trenches and hence progressively reduce trench leachate. It will also minimise the area of open vaults and hence degradation of the waste containers, reducing contaminant releases to water, as well as reducing overall leachate volumes. The use of warehouses to protect waste containers prior to emplacement in final disposal locations will also reduce the volumes of leachate generated, by intercepting and diverting clean rainwater runoff from their roofs.

- To promote unsaturated (that is, partially saturated) conditions in the vaults for as long as possible following closure, future vaults will be designed with side walls extending 1 m above the vault base, and incorporating engineered passive drainage arrangements so that, following final closure, residual infiltration through the cap may drain freely.
- An underground, low permeability cut-off wall will be constructed, integrated with the final cap, and the existing cut-off wall at the north-east corner of the site. The wall will extend to 2 m below the bases of the vaults. The wall will be of sufficient depth to limit inflow of surface water and shallow groundwater at the level of the vaults and trenches, and outflow of contaminated leachate close to the ground surface near the facility.
- Active leachate collection and management will continue during operations and after final disposals for as long as required.
- Site wastes will continue to be managed in accordance with our Waste Management Plan.
- The site will remain under a period of active institutional control, and it is assumed regulatory control, for around 100 years after completion of closure engineering. During this time, measures will be put in place to prevent damage to the closure engineering.
- Monitoring will continue during operations and afterwards during the period of active institutional control. It will provide reassurance that the process of engineering construction is consistent with design requirements for performance and associated Construction Quality Assurance, and after completion, that the repository is performing safely and as expected. It will support the decision on closing the gas vent. Remedial actions, such as addressing any problems with the final cap, will be taken if required.
- Records of the LLWR will continue to be archived at NDA’s national facility in Caithness. Other approaches to ensuring knowledge of the disposal facility and site is maintained after the end of the period of active institutional control, such as land covenants, may also be adopted. The ‘end state’ of the site (the state of the site at the time the NDA relinquish control) will be made consistent with local stakeholders

expressed desire for the site to become a sustainable amenity for the local community, thereby, helping to maintain knowledge and lower the likelihood of developments or uses that might lead to adverse impacts[6].

Additions to the SDP required for the disposal of less-hazardous ILW are summarised below.

- We would dispose all ILW in containers sufficiently strong to withstand the loads imposed during and after installation of the final cap. If the ILW could be handled in the same way as LLW, the containers would be stacked in the vaults alongside LLW (although stacked separately). The exact stacking arrangements would be dictated by several factors, including operational and Nuclear Safety Case considerations. The topmost container of each stack would be fitted with a container protection unit to protect its lid. Depending on waste receipt rates, containers would be interim stored in temporary warehouses alongside LLW containers prior to emplacement in final disposal locations.

- ILW that gives rise to dose rates necessitating additional shielding would be disposed in shielded modules, which would be constructed as required in Vaults 10, 11 and 12. We would allow for flexibility with respect to the location and size of the shielded modules in the vaults as this offers a means of managing uncertainty in the volumes of waste that may require shielding. It also provides flexibility in ensuring that operational and post-closure impacts from wastes in shielded modules are as low as reasonably achievable (ALARA). Bunding or drain arrangements would be used to isolate waters and leachate associated with the shielded modules 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 do not become leachate. This is consistent with the principle of minimising overall leachate volumes.

- Appropriate measures (for example, controls associated with the design, protective measures on plant, working patterns) would need to be identified and implemented to ensure that the operational safety impacts from ILW operations would be as low as reasonably practicable (ALARP).

- Consideration would need to be given as to whether wastes would require transport in Type B transport containers. If this were to be needed, then the site would require the capability to unpack disposal containers from transport containers. Buffer storage capability might also be required. Our current assumption is that this capability would be required.

Figure 3.4 shows the repository today. A schematic plan of the repository is shown in Figure 3.8 and representations of the further development of the repository are shown in Figure 3.9 and Figure 3.10. Figure 3.9 shows the first strip of the final cap completed over Vault 8 and

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[6] We do not rely on knowledge retention or amenity use in our demonstration that the regulatory risk guidance level will be met beyond the Period of Authorisation (including a 100-year period of active institutional control).

the adjacent area of the trenches, with a further area of the interim cap over the trenches being used to store excavated material. Figure 3.10 shows the final cap completed over the Repository.

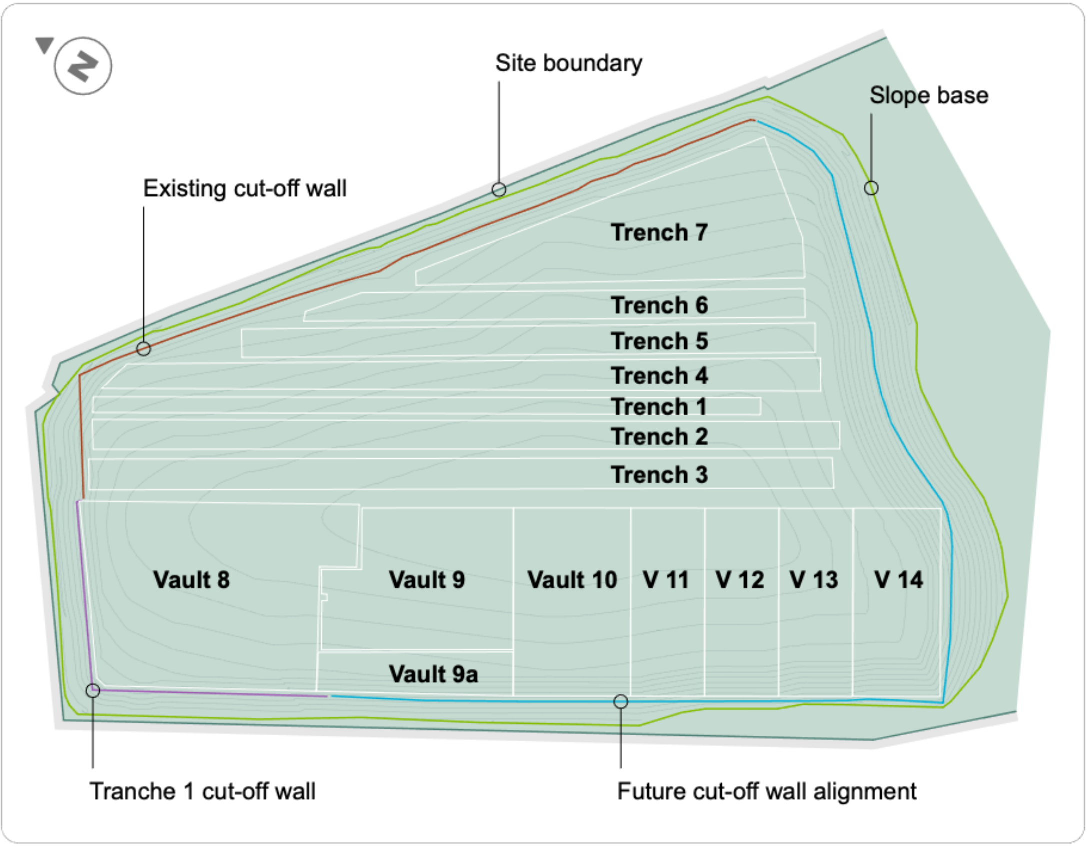


Figure 3.8: Schematic plan of the closed repository

Figure 3.9: Vault 8 final capped

Figure 3.10: Final cap completed over the repository

3.4 Inventory

3.4.1 Approach

General

We have developed a best estimate projection of the LLW inventory that will be disposed to the LLWR. This includes:

- the disposed inventory of the trenches;
- the waste received for disposal into the vaults, up to 31st March 2022;
- a projected forward vault inventory developed from the UKRWI, bounded by provisional constraints derived from the ESC and assumptions on waste treatment and diversion (see reference [3]).

The inventory disposed to the trenches and the vaults up to 31st March 2022 is collectively referred to as the Disposed Inventory. The projected forward vault inventory is referred to as the Reference Forward Inventory. The Disposed Inventory and the Reference Forward Inventory form the Reference Inventory used in the ESC.

The Reference Inventory also includes an estimate of the quantities of ILW that might be accepted if a decision is taken to accept ILW for disposal at the LLWR.

Disposed inventory

Since the 2011 ESC, we have enhanced the reliability and accuracy of the disposed inventory. The key improvements and initiatives carried out to date include:

- Reviewing the methodology used to derive the trench inventory. This involved reviewing the data collected as a part of the 2008 – 2009 RECALL exercise which involved interviews with current and former workers from Sellafield and the LLWR with the objective of developing a more informed view of practices involved in disposals to the trenches. The review concluded that the estimated uncertainties in trench radionuclide inventories of plus or minus an order of magnitude remain appropriate.
- Rederiving the disposed vault inventory to reflect an improved understanding of radionuclide and material composition. For legacy consignments to the vaults, the methods for estimating radionuclide and materials fingerprints have been refined and now rely on data from analogous streams reported in the UK UKRWI or waste consignments contained on Nuclear Waste Services' Tracking System.
- Introducing the eMWaste Tracking System on April 1st 2018, which replaced the previous Low-Level Waste Tracking System (LLWTS) and provides more detailed data for recent consignments, enhancing the level of detail and the comprehensiveness of the LLWR disposed vault inventory.

- Introducing improved waste consignment information forms (WCIs) facilitating the collection of more detailed information from consignors and consequently enhancing the quality of the inventory.

Reference forward inventory

Since submission of the 2011 ESC, management of waste within the UK nuclear industry has been revolutionised over the past decade through the application of the waste hierarchy, with the majority (by volume) of LLW now being disposed to suitable landfill facilities, incinerated or recycled. It has become more evident that consignors are increasingly making use of the range of disposal routes available and only consigning wastes to the repository that require the level of protection that the LLWR offers. Consequently, the Reference Forward Inventory presented in this document reflects the latest waste management practice and strategy, and accounts for updated programmes associated with new nuclear build.

Key aspects of the forward inventory critical to assessments and engineering design have been refined, including more accurate volume estimates to better calculate vault fill dates, enhanced data on the activities of key radionuclides, and detailed information on material composition, including non-radiological contaminants. We have focused on characterising and estimating the inventory as best as we reasonably can to provide a suitable basis upon which to analyse the performance of the repository.

The forward inventory is underpinned by the 2022 UKRWI, which was the latest available UKRWI at the time of undertaking our work. We have worked with waste producers to enhance and develop our understanding of the UKRWI to make amendments and improve confidence in the data. The UKRWI provides data related to waste on consignor sites including waste volumes, radionuclide activities, the timing of arisings, packaging plans and different waste treatment and disposal routes, and other information relevant to analyses of operational and post-emplacement safety. We process the data to more accurately reflect the expected waste composition at the time of disposal.

The Reference Inventory has been produced over three stages, with the Reference Inventory being the output of Stage 3. At each stage we have developed the inventory through iteration with the assessments. The Stage 1 Inventory was used in the Phase 1 assessments. The Stage 2 inventory was used in the Phase 2 assessments. This report uses the Stage 2 inventory and reports the findings of the Phase 2 assessments. The outputs of the Phase 2 assessments have been used to refine our understanding of the waste that could be accepted for disposal and develop the Stage 3 inventory.

The Reference Forward Inventory includes all LLW that is expected to arise within the UK that will not be disposed to a suitably permitted landfill site or Dounreay’s LLW disposal facility. This includes LLW that will be directly disposed and LLW that is subject to super-compaction, metallic treatment or incineration. Note that these routes sometimes result in reduced or zero waste volumes and activities being received at the LLWR. The LLW Reference Inventory also includes some waste streams associated with nuclear new build power stations and several opportunity wastes and decay-stored streams that have been identified as being suitable for disposal at the LLWR.

We include ILW in the Reference Forward Inventory, selecting waste streams following screening against stream constraints and radiological capacities. These stream constraints and radiological capacities have been calculated considering environmental and nuclear safety criteria. The radiological constraints are specified in terms of specific activity and the activity within a disposal package, whereas the radiological capacities are specified in terms of total activities that the repository may be able to accept.

Rates of waste receipt at the LLWR have decreased significantly over the last decade due to the success of waste treatment and diversion. Once the effects of waste treatment and diversion are accounted for, rates of receipt have been significantly below those expected from the UKRWI. Speaking to waste producers, we anticipate that rates of waste receipt will continue to be below those anticipated by the UKRWI in the near term. In response we have developed three waste arisings profiles (Figure 3.11): UKRWI profile, reduced UKRWI profile and further reduced UKRWI profile. The total amount of waste disposed is the same in all three profiles. Near-term receipts are lower in the reduced UKRWI profile than the UKRWI profile but catch up later so the date of the final disposal is only a little later than the UKRWI profile. Near-term receipts are even lower in the further reduced UKRWI profile, but again they catch up later, so the date of the final disposal is only a little later than the reduced UKRWI profile.

The further reduced UKRWI profile is the reference assumption for our assessment calculations. The choice of arising profile does not change the total activity, volume or mass disposed. It does affect how much waste would be in containers of current designs and how much would be in proposed future stronger containers. Almost all arisings from c.2030 are anticipated to be in future stronger containers.

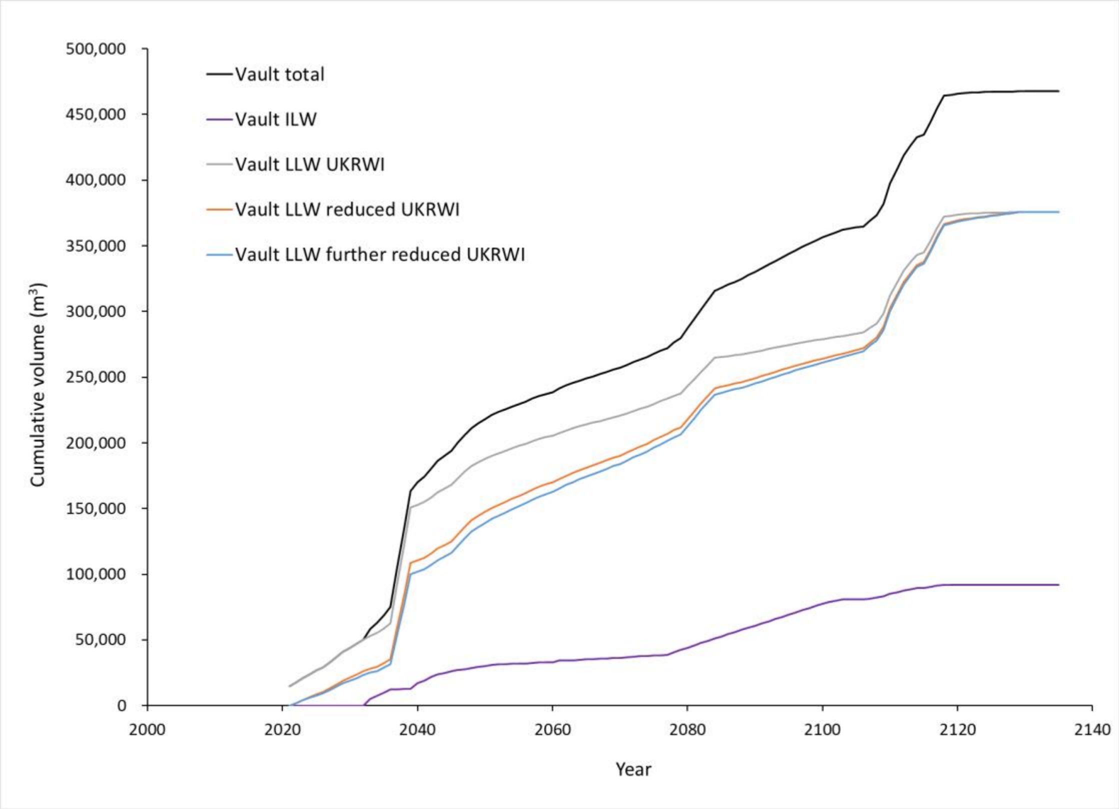


Figure 3.11: Waste arisings profiles (Vault total is LLW UKRWI + ILW)

Uncertainty in the forward inventory

There are two types of uncertainty in the forward inventory.

Firstly, there is uncertainty in the characteristics of a given waste stream arising from the measurement techniques used by the consignor. Measurement and characterisation techniques are likely to be inherently cautious.

The second type of uncertainty lies in the inventory forecasts. Since the 2011 ESC, we have learned more about uncertainty in the inventory forecasts because we have started to track actual disposals and compare them with forecast disposals. Learning from this process has informed the development of the three arisings profiles described above, and our approach to managing forecast inventory uncertainty in the ESC (see below).

There is uncertainty in the UKRWI, for example due to the inherent uncertainty in the LLW inventory that will arise from future decommissioning, and uncertainty over the effects of diversion, including the potential to dispose of some wastes at other disposal facilities. However, the UKRWI is relatively mature, and we therefore anticipate that future changes to the forward LLW inventory will be limited. We will continue to review any changes and their implications as they arise.

If ILW were to be disposed of at the facility in the future, there would also be an additional source of uncertainty relating to which waste streams would be accepted. The radiological capacity of the LLWR will not be able to accommodate all the ILW identified in the UKRWI. Radiological capacity constraints would therefore be used to make informed decisions on the subset of ILW that could be accepted, adopting these into an optimised plan that also takes account of the strategic benefit offered by early disposal for each waste stream. The ILW streams included in the Reference Forward Inventory have been selected using this approach, and the selection has already been updated through several iterations including drawing on the results of our Phase 1 assessments, wider generic work undertaken by the NDA on near surface disposal of certain ILW, and through initial discussions with waste producers. The inventory will be further iterated as uncertainty in the inventory reduces and decommissioning plans develop.

For both ILW and LLW, uncertainty in the inventory forecast would be resolved no later than the point of consignment, leaving only characterisation uncertainty.

Although there are uncertainties in the forward inventory, the ESC provides confidence that disposal of LLW and ILW would be safe. For ILW and LLW, the ESC also provides wider insights into the key radionuclides and their behaviours, safety functions and the performance of barriers, optimisation of the repository, and the implications of uncertainties to inform future work.

To manage these uncertainties, the 2026 ESC emphasises using assessment results to develop radiological capacities, WAC, repository management controls and approaches, that build confidence the potential impacts from the facility will be consistent with the requirements of the GRA [17].

3.4.2 Inventory

This subsection summarises the Stage 3 Reference Inventory, except for Table 3.1, which provides data from the Stage 2 Reference Inventory, as used in the assessment calculations. Further information is provided in reference [4] [61].

LLW

A total waste volume of 815,000 m^3 (excluding cover material) has been disposed to the trenches. Compaction of the waste during disposal, under its own weight and the weight of operating plant, reduced the volume of waste and daily cover materials to around 469,000 m^3. The volume has since further reduced through degradation of the waste and compaction under its own weight and the weight of the interim trench cap.

The packaged volume of waste disposed to the vaults as of 31st March 2022 was 234,000 m^3. A further 376,000 m^3 of packaged LLW is expected to be disposed to the vaults in future, giving a total packaged LLW volume of 610,000 m^3.

A total of 565 TBq of activity has been disposed to the trenches, a further 122 TBq had been disposed to the vaults as of 31st March 2022, and a further 1,820 TBq of LLW is expected to be disposed to the vaults after this date (Table 3.1). Thus, a total LLW inventory of 2,510 TBq is anticipated to be disposed to the LLWR, although this value does not account for any radioactive decay after the point of disposal.

Although we are expecting to receive smaller volumes of LLW compared with the 2011 ESC, we expected the total radioactivity will be similar. The reduction in LLW volume will be compensated for by segregation of LLW from Higher Activity Waste (HAW) streams that were originally destined for disposal at the Geological Disposal Facility (GDF).

Table 3.1: Summary of the inventories of selected radionuclides that are most important for long-term safety (according to the Stage 2 inventory used for the assessments)

Radionuclide	Trenches (TBq)	Vaults (TBq)				Total (TBq)
	LLW	Existing LLW	Future LLW committed to current containers	Future LLW in new stronger containers	Future ILW in SWTC compatible strong boxes	
C-14	1.0 10^{-1}	6.4 10^{-1}	2.4 10^{0}	1.7 10^{1}	8.1 10^{1}	1.0 10^{2}
Cl-36	1.9 10^{-2}	6.0 10^{-1}	4.2 10^{-1}	6.6 10^{-1}	1.2 10^{0}	2.9 10^{0}
Nb-94	7.9 10^{-5}	7.0 10^{-3}	7.4 10^{-3}	3.4 10^{-2}	1.6 10^{0}	1.7 10^{0}
Tc-99	2.1 10^{-1}	2.7 10^{0}	7.0 10^{-2}	1.2 10^{-1}	1.1 10^{0}	4.2 10^{0}
I-129	3.0 10^{-3}	3.3 10^{-3}	3.3 10^{-2}	8.3 10^{-4}	1.5 10^{-1}	1.9 10^{-1}
Ra-226	2.9 10^{-1}	1.1 10^{-1}	4.1 10^{-3}	2.4 10^{-2}	8.4 10^{-4}	4.3 10^{-1}
Th-232	1.3 10^{0}	7.5 10^{-2}	7.6 10^{-4}	1.1 10^{-3}	3.3 10^{-3}	1.4 10^{0}
U-234	5.9 10^{0}	7.3 10^{-1}	1.0 10^{0}	2.3 10^{-1}	1.3 10^{0}	9.1 10^{0}
U-238	6.7 10^{0}	1.1 10^{0}	3.3 10^{-1}	5.1 10^{-2}	4.0 10^{-1}	8.6 10^{0}
Pu-239	1.6 10^{0}	5.6 10^{-1}	2.3 10^{0}	1.9 10^{0}	3.0 10^{1}	3.6 10^{1}

ILW

Disposal of ILW would bring the total packaged waste volume upon closure to around 1,151,000 m^3, filling up to Vault 11 and partially filling Vault 12. If ILW was accepted for disposal, a further 15,000 TBq could be disposed to the vaults bringing the total inventory plus ILW disposals to 17,600 TBq.

ILW can be divided into two types: ILW that can be managed in the same way as LLW, and ILW requiring different management arrangements, i.e. operational shielding. ILW requiring different management arrangements would be disposed in shielded modules (Subsection 3.3). Waste package dose rates can vary significantly within a waste stream. Waste packages would be allocated for disposal in shielded modules based on their dose rate not their waste stream. Therefore, the highest dose rate packages from all waste streams would be directed for disposal in shielded modules.

There is limited information on the activity and dose rate distributions within waste streams. Based on initial analyses from our optimisation work [10], we have assumed 40 vol% of each ILW waste steam would be placed in shielded modules. Correspondingly, 40% of the inventory of each radionuclide is assumed to be in shielded modules. These assumptions underpin the assessment calculations. If ILW is disposed in the vaults, the percentage of activity disposed in shielded modules might be higher than the volume percent.

3.4.3 Spatial Variability in the Distribution of Key Radionuclides

Our assessments use gridded representations (in plan view) of the radionuclide inventories. The discretisation reflects the spatial distribution of key radionuclides for the potential long-term impacts from the repository, the characteristics of the pathways, and the human habits and actions that might lead to exposures. The discretisation used for each pathway assessment, and treatment of heterogeneity at different length scales, is described in subsequent sections of this report.

The key radionuclides and spatial scales of interest are similar for the coastal erosion assessment, radon gas assessment and human intrusion assessment. Therefore, these three assessments use the same assessment model grid and gridded inventory data. The assessment model grid is shown in Figure 3.12. The spatial distribution of key radionuclides is shown in Figure 3.13 to Figure 3.15.

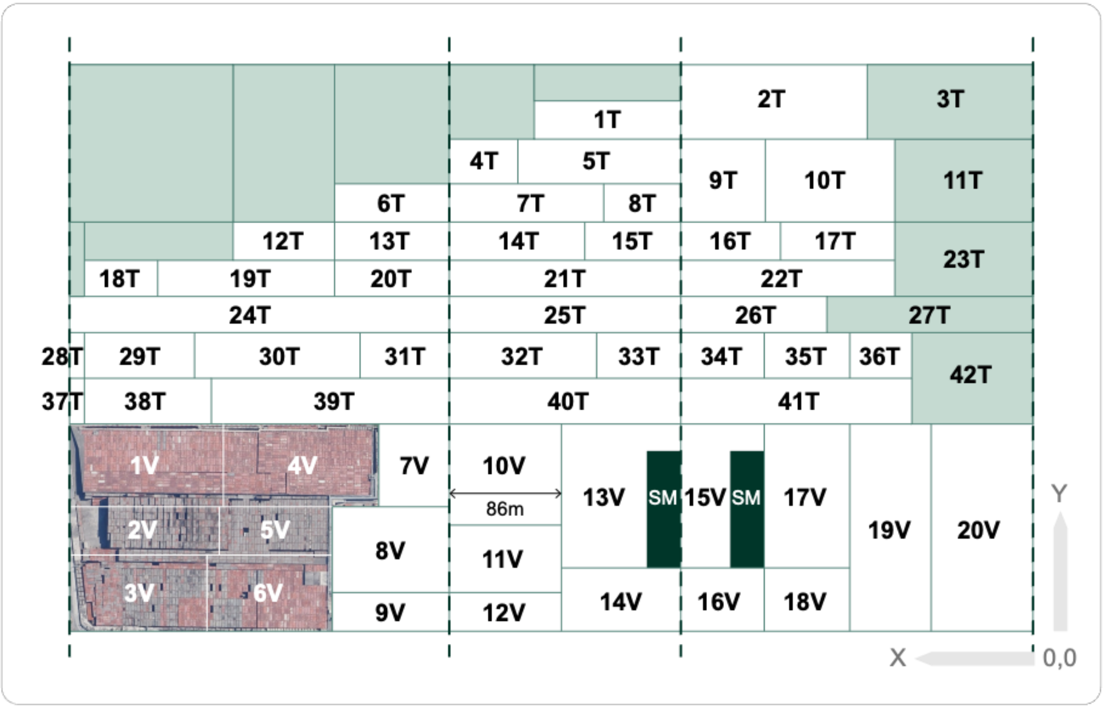


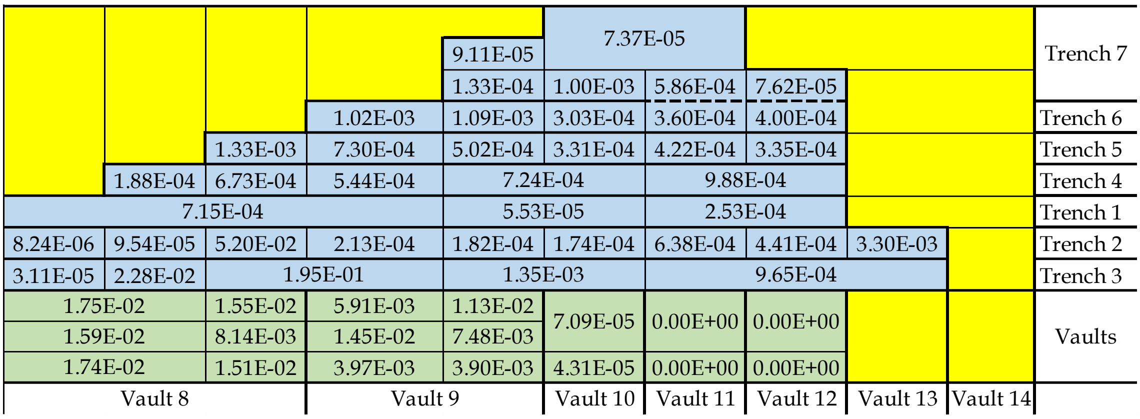
Figure 3.12: Assessment model grid for the coastal erosion, radon gas and human intrusion assessments

Ra-226 (Figure 3.13) is mainly associated with wastes from Thorium Ltd disposed to Trench 3. The disposal forms record the waste as ‘filter press cake’ containing Ra-226 as a daughter of U-238. The only mineral ore used in the commercial production of thorium containing identifiable quantities of U-238 is thorite sand. Therefore, the source material is assumed to be thorite sand containing Th-232 and U-238 in secular equilibrium with Ra-228 and Ra-226 [62]. Ra-226 is also present in existing vault disposals, including in sealed sources.

Th-232 (Figure 3.14) is mainly present in monazite sands disposed to the north end of Trench 2 and thorite sands disposed to the south end of Trench 2, and in material from Streetly chemicals disposed more evenly along trenches 4 and 5 [62].

Pu-239 (Figure 3.15) is mainly present in ILW plutonium contaminated material (PCM) wastes, which could potentially be disposed to Vaults 10 and 11, including within shielded modules, if ILW disposal is taken forward.

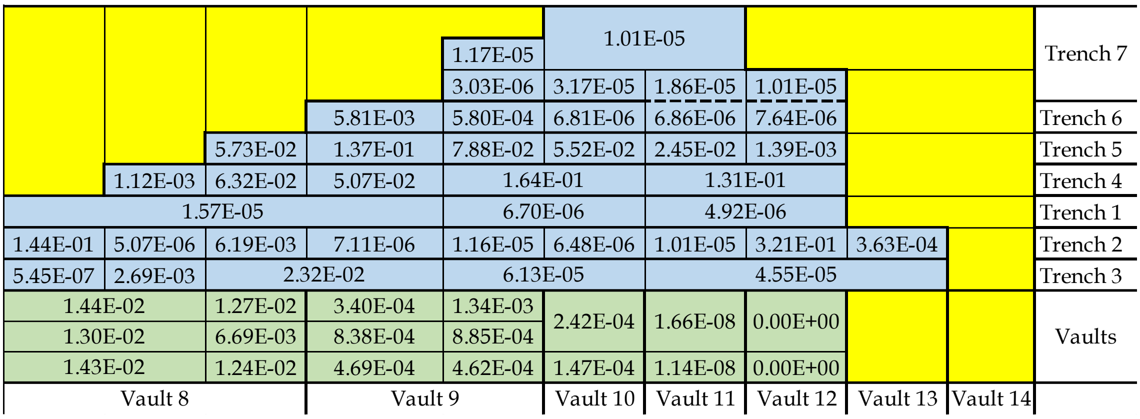
Ra226



				9.11E-05	7.37E-05					Trench 7
				1.33E-04	1.00E-03	5.86E-04	7.62E-05			
			1.02E-03	1.09E-03	3.03E-04	3.60E-04	4.00E-04			Trench 6
		1.33E-03	7.30E-04	5.02E-04	3.31E-04	4.22E-04	3.35E-04			Trench 5
	1.88E-04	6.73E-04	5.44E-04	7.24E-04		9.88E-04				Trench 4
7.15E-04				5.53E-05		2.53E-04				Trench 1
8.24E-06	9.54E-05	5.20E-02	2.13E-04	1.82E-04	1.74E-04	6.38E-04	4.41E-04	3.30E-03		Trench 2
3.11E-05	2.28E-02	1.95E-01		1.35E-03		9.65E-04				Trench 3
1.75E-02		1.55E-02	5.91E-03	1.13E-02	7.09E-05	0.00E+00	0.00E+00			Vaults
1.59E-02		8.14E-03	1.45E-02	7.48E-03						
1.74E-02		1.51E-02	3.97E-03	3.90E-03	4.31E-05	0.00E+00	0.00E+00			
Vault 8			Vault 9		Vault 10	Vault 11	Vault 12	Vault 13	Vault 14	

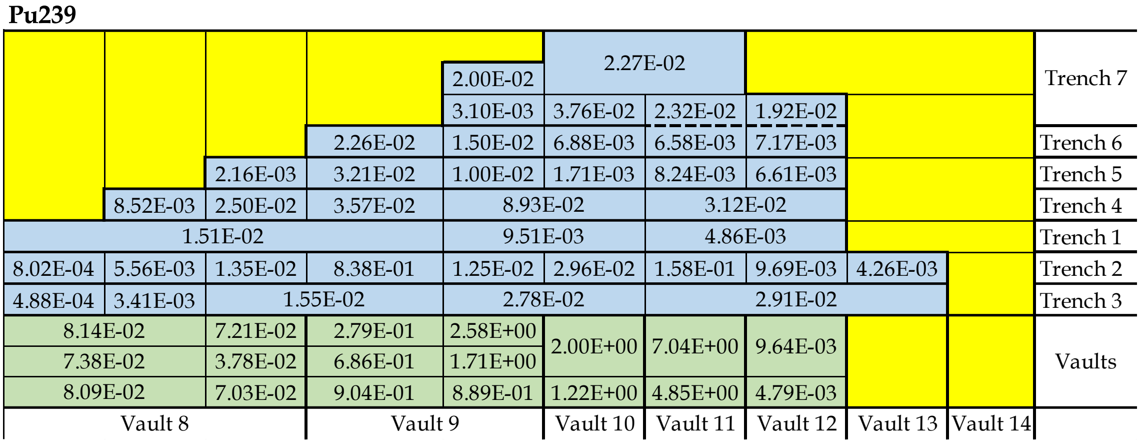
Figure 3.13: Distribution of Ra-226 (TBq) in the assessment model grid shown in Figure 3.12 (inventory in shielded modules in Vaults 10 and 11 is not shown but is included in the assessment)

Th232



				1.17E-05	1.01E-05					Trench 7
				3.03E-06	3.17E-05	1.86E-05	1.01E-05			
			5.81E-03	5.80E-04	6.81E-06	6.86E-06	7.64E-06			Trench 6
		5.73E-02	1.37E-01	7.88E-02	5.52E-02	2.45E-02	1.39E-03			Trench 5
	1.12E-03	6.32E-02	5.07E-02	1.64E-01		1.31E-01				Trench 4
1.57E-05				6.70E-06		4.92E-06				Trench 1
1.44E-01	5.07E-06	6.19E-03	7.11E-06	1.16E-05	6.48E-06	1.01E-05	3.21E-01	3.63E-04		Trench 2
5.45E-07	2.69E-03	2.32E-02		6.13E-05		4.55E-05				Trench 3
1.44E-02		1.27E-02	3.40E-04	1.34E-03	2.42E-04	1.66E-08	0.00E+00			Vaults
1.30E-02		6.69E-03	8.38E-04	8.85E-04						
1.43E-02		1.24E-02	4.69E-04	4.62E-04	1.47E-04	1.14E-08	0.00E+00			
Vault 8			Vault 9		Vault 10	Vault 11	Vault 12	Vault 13	Vault 14	

Figure 3.14: Distribution of Th-232 (TBq) in the assessment model grid shown in Figure 3.12 (inventory in shielded modules in Vaults 10 and 11 is not shown but is included in the assessment)



Pu239

				2.00E-02	2.27E-02					Trench 7
				3.10E-03	3.76E-02	2.32E-02	1.92E-02			
			2.26E-02	1.50E-02	6.88E-03	6.58E-03	7.17E-03			Trench 6
		2.16E-03	3.21E-02	1.00E-02	1.71E-03	8.24E-03	6.61E-03			Trench 5
	8.52E-03	2.50E-02	3.57E-02	8.93E-02		3.12E-02				Trench 4
1.51E-02				9.51E-03		4.86E-03				Trench 1
8.02E-04	5.56E-03	1.35E-02	8.38E-01	1.25E-02	2.96E-02	1.58E-01	9.69E-03	4.26E-03		Trench 2
4.88E-04	3.41E-03	1.55E-02		2.78E-02		2.91E-02				Trench 3
8.14E-02		7.21E-02	2.79E-01	2.58E+00	2.00E+00	7.04E+00	9.64E-03			Vaults
7.38E-02		3.78E-02	6.86E-01	1.71E+00						
8.09E-02		7.03E-02	9.04E-01	8.89E-01	1.22E+00	4.85E+00	4.79E-03			
Vault 8			Vault 9		Vault 10	Vault 11	Vault 12	Vault 13	Vault 14	

Figure 3.15: Distribution of Pu-239 (TBq) in the assessment model grid shown in Figure 3.12 (inventory in shielded modules in Vaults 10 and 11 is not shown but is included in the assessment)

The features and spatial scales of interest for groundwater pathway assessment and C-14 gas assessment are different. Therefore, these two assessments each use a less detailed spatial representation of the inventory, for the reasons discussed in Subsections 6.4.3, 10.6.3, and 10.7).

3.5 Scenarios for Site Evolution

As described in Subsection 2.9, the repository is expected to be disrupted by coastal erosion on a timescale of several hundred to a few thousand years after present, with erosion of the repository being complete within one to a few thousand years after present [8]. It is possible that people may choose to maintain current coastal defences along the West Cumbrian coast for decades to come or build new defences in the future. Coastal protection measures would only be effective while they are actively maintained, and maintenance would become increasingly difficult as the magnitude of sea-level rise increases. We cannot assume that existing defences would be effectively maintained or new defences built after the end of the PoA. The potential influences of existing or new defences on the timescales for disruption are small compared with the uncertainties in future global greenhouse gas emissions, climate change, sea-level rise, response of the coastal system and the future rate of erosion.

Three emissions scenarios (Reference, Low and High) are considered for the 2026 ESC (Subsection 2.8). These scenarios are considered for all pathways. Climate and landform changes are different for each scenario, and so are the impacts on hydrology and hydrogeology.

We also assess a 'what-if' case for the groundwater pathway in which the repository is not eroded on the timescale of a few thousand years. We consider potential risks up to 5,000 years after present, assuming there is no coastal erosion. This is not plausible as the coast is currently eroding and would continue to erode even with no relative sea-level rise. However, it is useful to examine the longer-term performance of the engineered barriers and examine the long-term in-growth of radioactive daughter radionuclides.

3.5.1 Expected Evolution

Future environmental changes are uncertain, but as summarised in reference [8] and detailed in references [63] [48], we have sufficient scientific understanding to be able to estimate the range of possible future conditions and also set out credible future evolutions that are the basis for our assessments.

The three main environmental factors/variables that impact on our assessments of the groundwater, gas and coastal erosion pathways are as follows.

- Sea level is a boundary condition for hydrogeological modelling (hence flows in the near field and groundwater) and sea-level rise is a factor in coastal recession, below.
- Coastal recession (erosion) is a factor in hydrogeological modelling, determining the groundwater path length to coast and the time at which this length is reduced to zero and erosion of the facility begins.
- Climate, in particular, Hydrologically Effective Rainfall (HER), is a factor in hydrogeological modelling (hence flows in the near field and groundwater). Climate also affects agricultural practice including irrigation.

3.5.1.1 Sea Level, Timing and Style of Erosion

Based on measurements and on climate and oceanographic modelling, the current scientific consensus is that the present-day global temperature conditions and committed anthropogenic components will lead to substantial global sea-level rise due to thermal expansion and loss of continental ice. In the regional setting (NW England), this will be countered by a comparatively small isostatic land rise.

Integrated projections of Low, Reference (central) and High greenhouse gas emissions, eustatic and local sea-level rise have been developed for the 2026 ESC [63] [48]. These are based on Intergovernmental Panel on Climate Change (IPCC) [64] eustatic (global) and regional sea-level rise projections for different emissions scenarios. IPCC's regional projections include several factors that modify eustatic sea-level changes. The magnitudes of these factors vary with location on the earth. IPCC's regional projections are then modified to account for local isostatic changes.

The emissions scenarios included in the 2026 ESC are the:

- Reference Emissions Scenario (equivalent to the IPCC shared socio-economic pathway (SSP) 2-4.5), in which CO_2 emissions remain around current (2010-2020) levels until the middle of the century before decreasing;
- Low Emissions Scenario (equivalent to SSP1-2.6), in which CO_2 emissions decline to net zero after 2050, followed by net negative CO_2 emissions;
- High Emissions Scenario (equivalent to SSP5-8.5), involving CO_2 emissions that roughly double from current levels by 2050 and peak at three times current levels around 2090.

Projections of local sea-level rise for the Low, Reference and High Emissions Scenarios are shown in Figure 3.16. The projections used in the 2011 ESC are also shown for comparison. The high projection is a little lower than the 2011 ESC. It excludes the effects of ice sheet and ice cliff instability that could lead to additional melting of the Eastern Antarctic Ice Sheet and additional sea-level rise. However, there is not international consensus on the effects of ice sheet and ice cliff instability, and there is significant uncertainty at such high levels of emissions and sea-level rise. The effects of ice sheet and ice cliff instability are considered as a 'what-if' case in the coastal erosion assessment [25].

The Reference sea-level rise projection is a little higher than the low projection for the 2011 ESC. However, it still includes several metres of sea-level rise. This would have major impacts on the coastline, coastal towns and cities around the UK. It assumes net zero by 2050 is not achieved, and there is no reduction in emissions until 2050.

The Low projection is a climate scenario that was not included in the 2011 ESC. It assumes net zero by 2050 is achieved and net negative emissions are achieved post-2050, i.e. greenhouse gas emissions fall below the rate at which they are removed from the atmosphere-ocean system.

For a given climate scenario, sea-level rise is unlikely to be a continuous process. There may be sudden step changes as large volumes of ice pass a tipping point and melt. These tipping points cannot readily be projected, but their potential effects are considered in the coastal erosion assessment.

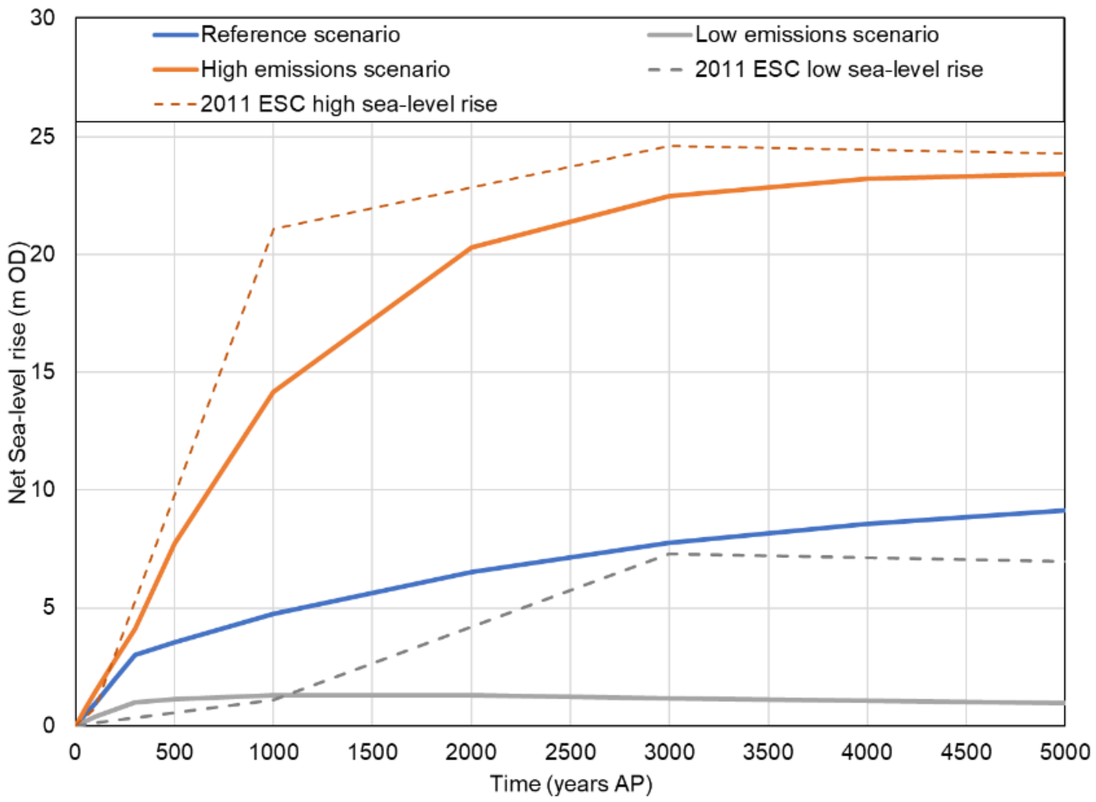


Figure 3.16: Projected sea level and rate of sea-level change, from reference [8]

The amount of sea-level rise affects the rate and mode of erosion of the repository. The integrated projections of emissions and sea-level rise have been extended to also include the timing, rate and mode of erosion of the repository [8]. Three modes of erosion are possible:

- Undercutting − the elevation of the erosion front is below the base of the repository. Waste is exposed in the cliffs at the back of the storm beach, above the elevation of the storm beach. Storm waves, coincident with high spring tide, overtop the storm beach and erode sediments from the base of the cliffs. The repository is undercut, and material collapses from the cliffs onto the storm beach.
- Direct erosion − waste is exposed in the cliffs at the back of the storm beach, at the elevation of the storm beach. Storm waves, coincident with high spring tide, overtop the storm beach and erode materials from the repository. Again, undercutting occurs and repository materials higher in the cliffs collapse onto the storm beach. Some waste may be below the level of the back of the storm beach and may not be eroded. This waste may remain in situ and later exposed on the foreshore as erosion continues.
- Inundation − sea-level rises above the elevation of the land adjacent to the repository. The land surrounding the repository is flooded by the sea, initially at high spring tide, but increasingly over a wider range of tidal conditions if sea-level rise

continues. The elevation of the LLWR cap is higher than the surrounding land, so it forms an island at high tide, which is attacked and eroded by the sea.

A summary of the projected timing, rate and mode of erosion, and the fate of the eroded material is provided in Table 3.2. Inundation is only possible for inland trenches in the High projection.

Table 3.2: Summary of coastal erosion projections [48]

	Reference Emissions Scenario	High Emissions Scenario	Low Emissions Scenario
Relative sea level when repository first exposed (at least 350 m erosion has occurred)	6 metres above Ordnance Datum (m OD)	10 m OD	2 m OD
Relative sea level when repository completely eroded (at least 675 m erosion has occurred)	8 m OD (0.28 to 0.36 m y^{-1})	21 m OD (0.47 to 0.60m y^{-1})	2 m OD (0.20 to 0.26 m y^{-1})
Timescale for first exposure of the repository (northern area that is closest to the coastline)	1,000 to 1,500 years AP	500 to 1,000 years AP	1,500 to 2,000 years AP
Timescale for first exposure of the repository − modelling assumption	1,250 years AP	750 years AP	1,750 years AP
Timescale for complete loss of the northern part of the repository − modelling assumption	Eroded in 1,500 years (0.22 m y^{-1}) after first exposure Loss in 2,750 years after present (AP)	Eroded in 1,000 years (0.33 m y^{-1}) after first exposure Loss in 1,750 years AP	Eroded in 2,000 years (0.16 m y^{-1}) after first exposure Loss in >2,500 years AP

Mechanism of repository disruption	Undercutting. Indirect wave splash of wastes during periodic storms, but no potential for direct wave attack. Wastes will fall from the cliff to form a temporary talus slope (<5 m from base of wastes to base of cliffs). Wastes in deep vaults (base at 10 m OD) will experience direct erosion with wastes that are not eroded cropping out on the shore platform below mobile sand bars.	Undercutting initially, with direct wave erosion for most of the repository. Waste will form a temporary talus slope. As sea-level rise continues, the eastern (i.e. more inland) part of the repository (vaults and trenches) will experience direct erosion with wastes that are not eroded cropping out on the shore platform below mobile sand bars.	Undercutting. Indirect wave splash of wastes during periodic storms, but no potential for direct wave attack. Wastes will fall from the cliff to form a temporary talus slope (around 5 m to 10 m from base of wastes to base of cliffs).

3.5.1.2 Boundary Conditions for Groundwater Modelling

The impacts of sea-level rise and coastal erosion on groundwater heads and flows are modelled for the Low, Reference and High projections using ConnectFlow [65]. The results are used to parameterise the groundwater pathway assessment model.

For each climate scenario, ConnectFlow is used to calculate groundwater heads and flows for several time snaps (Table 3.3). At successive time snaps, the distance between the repository and the coast decreases, and sea-level increases. The final time snap is shortly before the repository starts to be disrupted by coastal erosion.

Table 3.3: Hydrogeological model time snaps

Time snaps (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 Emissions Scenario erosion time)
3280	1,150 years after as-built (similar to Reference Emissions Scenario erosion time)
3780	1,650 years after as-built (similar to Low Emissions Scenario erosion time)

In addition to changes to the position of the coast and sea level, each snapshot model includes changes to groundwater recharge (top surface boundary condition) and the hydraulic conductivity of the repository engineering components [13] [65] [66].

The model results are illustrated in Figure 3.17 [65]. This shows groundwater pathlines from the repository, superimposed on the deeper geological units for the Reference Emission Scenario.

Once the erosion front reaches the boundary of the repository there is no groundwater pathway [67] [68]. This is the end of the groundwater pathway assessment. The potential impacts of discharges of water from the eroding repository to the storm beach are considered in the coastal erosion assessment. The amount of activity in water seeping from waste exposed in the cliffs is expected to be trivial compared with the amount of activity in the exposed waste [25].

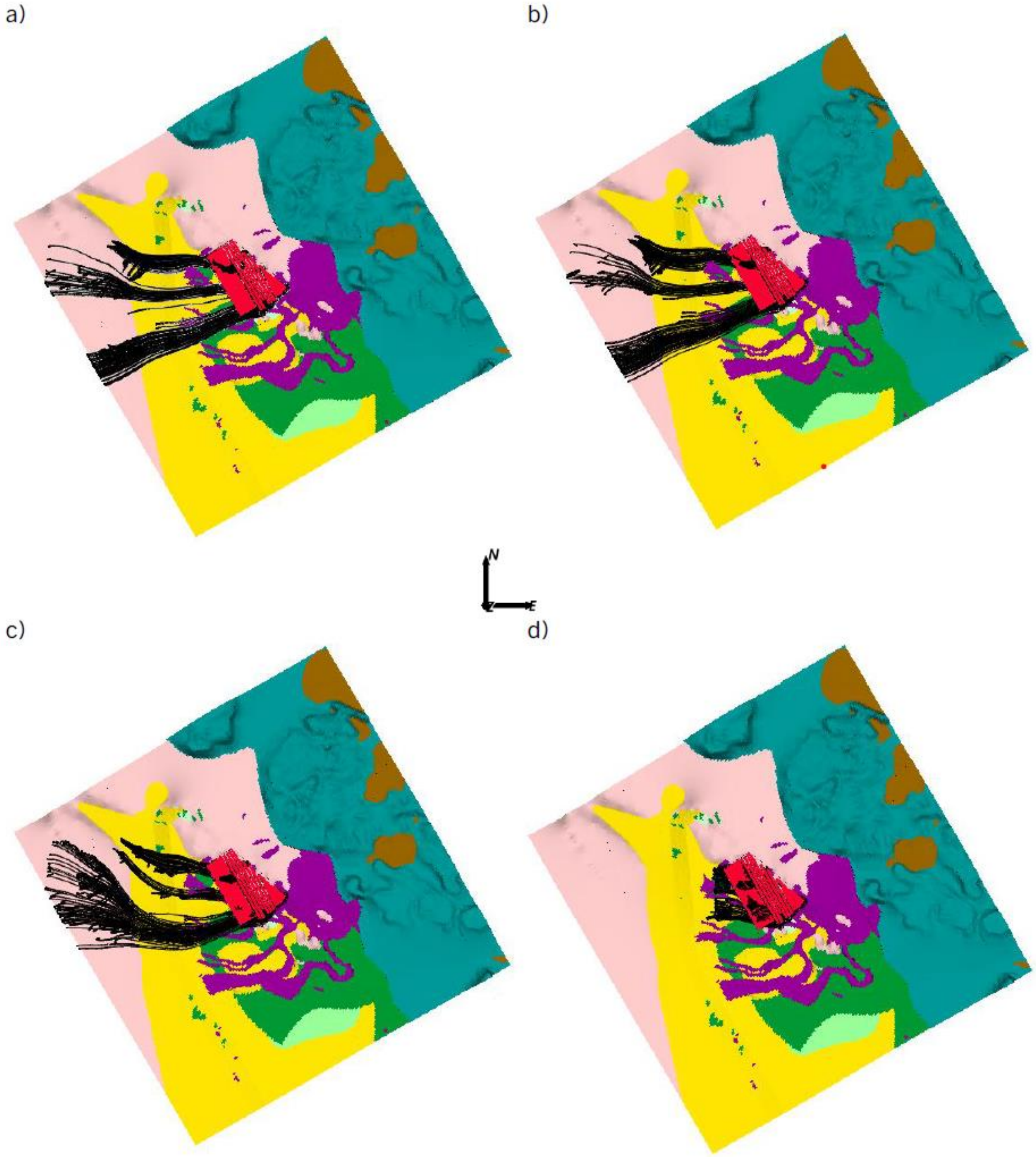


Figure 3.17: Surface view of the Reference Emissions Scenario site-scale models showing the pathlines (in black) from the repository overlaid over the geology for a) 2130 b) 2230 c) 2430 and d) 3280. Colouring shows geological units - pink is sandstone, green is D, purple is C, yellow is B3SS, teal is outwash till and brown is inland lake.

3.5.1.3 Climate and Hydrologically Effective Rainfall

Changes to temperature, rainfall and evaporation have been projected for several times within each climate scenario [63]. Climate changes to 2100 AD are based on UKCP18 data. Climate changes post-2100 AD have been considered in studies supporting the IAEA MODARIA programme and national radioactive waste management programmes e.g. references [69] [70] [71].

Long-term global temperature change projections have been calculated and compared with projections from UKCP18 for the period to 2100 AD. Using these two projections, scaling factors between global and local (UKCP18) temperature changes have been calculated. These scaling factors are then applied to global temperature changes post-2100 AD to calculate local temperature changes post-2100 AD. The same scaling factors are also relevant to changes in precipitation [8].

Reference [8] argues that simple evapotranspiration models provide similar estimates of changes in evapotranspiration to more complicated models. Reference [8] uses a simple evapotranspiration model to calculate changes in evapotranspiration for each climate scenario. The change in HER (i.e. precipitation minus evapotranspiration) can then be calculated. Reference [65] also calculates changes to groundwater recharge, i.e. HER minus water that forms quickflow (overland flow and interflow) to site streams and drains.

Table 3.4: Projected changes to temperature, precipitation and evaporation [8] [65]

Climate scenario	Parameters (mm y^{-1})	Present day groundwater flow model (2011 to 2021 data)	Climate baseline (2003 to 2010 data)	Time after present				
				100 y	300 y	1,000 y	3,000 y	10,000 y
Reference	Precipitation	1,229	1,057	1,102	1,102	1,102	1,093	1,088
	Potential Evapotranspiration	427	416	470	470	470	461	453
	Hydrologically Effective Rainfall	802	641	632	632	632	632	635
	Groundwater recharge	382	305	301	301	301	301	302
Low emissions	Precipitation	1,229	1,057	1,080	1,080	1,076	1,069	1,057
	Potential Evapotranspiration	427	416	445	445	441	432	416
	Hydrologically Effective Rainfall	802	641	635	635	636	637	641
	Groundwater recharge	382	305	302	302	302	303	305

Climate scenario	Parameters (mm y^{-1})	Present day groundwater flow model (2011 to 2021 data)	Climate baseline (2003 to 2010 data)	Time after present				
				100 y	300 y	1,000 y	3,000 y	10,000 y
High emissions	Precipitation	1,229	1,057	1,130	1,152	1,214	1,186	1,120
	Potential Evapotranspiration	427	416	507	536	615	578	495
	Hydrologically Effective Rainfall	802	641	623	616	599	608	626
	Groundwater recharge	382	305	296	293	285	289	298

3.5.2 'What-if' Delayed Coastal Erosion Case

There is no scientific basis for this 'what-if' case. It would require extreme measures to protect the coast for thousands of years. We cannot assume this in our assessment as it could imply an extreme degree of institutional control of the site and therefore impacts. We include this case to gain insight to the potential longer-term function of the engineered barriers and to follow the impact of long-term radioactive decay and ingrowth. In particular, the case allows the potential impact of the long-term ingrowth of Ra-226 from U-234 via Th-230 to be assessed; this was an issue in the 2002 post-closure radiological safety assessment (PCRSA) [72].

The case is assessed to 250,000 years after present. Since we see no natural basis for the case, we cannot assign evolving properties to the environment for this case. Therefore, to assess the performance of the facility within this case we assume an unchanging present-day sea level and coastline.

3.6 Evolution of the LLWR

The objective of this LTRA report is to present a detailed quantitative assessment of the radiological performance of the LLWR. It is useful, however, to first set out a qualitative description of the functions and expected evolution of the LLWR and its environment. This is based on the understanding developed during our current and previous assessments.

3.6.1.1 Function of the LLWR

The function of the LLWR, as for any near-surface disposal facility for radioactive waste, is to isolate and contain the radionuclides and associated chemotoxic materials. These functions are discussed in more detail in reference [12]. Some discharges are planned during the period of operations and institutional control and some releases to groundwater and gas are inevitable. The repository is expected to be disrupted by coastal erosion on a timescale of several hundred to a few thousand years after present, with erosion of the repository being complete within one to a few thousand years after present. We observe that, in the long term, all near-surface disposal facilities are vulnerable to disruption by natural erosion processes, human actions or combinations of natural and human events. This is accounted for by setting limits on the waste that may be disposed in the facility. The key question is whether the potential impacts when erosion occurs are consistent with regulatory guidance levels, i.e. appropriately low, which is addressed in Section 7 of this report.

Here we set out an account of the expected evolution of the LLWR over the lifetime of the disposal facility until it is fully eroded. Note that safety is achieved by controlling the radioactive and non-radiological contaminants that are disposed and optimising long-term isolation and containment by the engineered barriers [10] [12].

3.6.1.2 Period of Authorisation

Aqueous and aerial discharges are made under the terms of our Permit. Releases of contaminants (mainly tritium) to groundwater have occurred in the past and are likely to be continuing, albeit to a lesser degree today. Radioactive decay and releases that occur during the PoA, including historical releases, affect the radionuclide inventory remaining at the start of the post-PoA period. Where this is potentially significant, our assessment models represent disposals, radioactive decay (and ingrowth), and releases for the full lifetime of the repository, from when it opened in 1959 until it starts to be disrupted by coastal erosion. Calculated radionuclide releases during the PoA are provided to the PoA assessment to calculate potential doses [14].

A key feature of the LLWR, and other near-surface disposal facilities, is that substantial radioactive decay of shorter-lived radionuclides, and hence diminution of radiological hazard, occurs between the time of disposal and the end of the period management control. During this time, unauthorised access to the disposal site and waste is prevented, and hence direct exposure to the waste, human intrusion into the waste and deleterious actions or uses of the site area are all prevented.

The change in total activity in the LLWR due to decay and ingrowth over the lifetime of the facility is shown in Figure 3.18. The figure excludes any decrease in activity due to releases. The total activity decreases during the period of management control, and most of the short-lived activity decays during this period. The activity remaining at the end of the period of management control is due to longer lived radionuclides and changes little through the remainder of the assessment timeframe. Evolution of the hazard is discussed in reference [12].

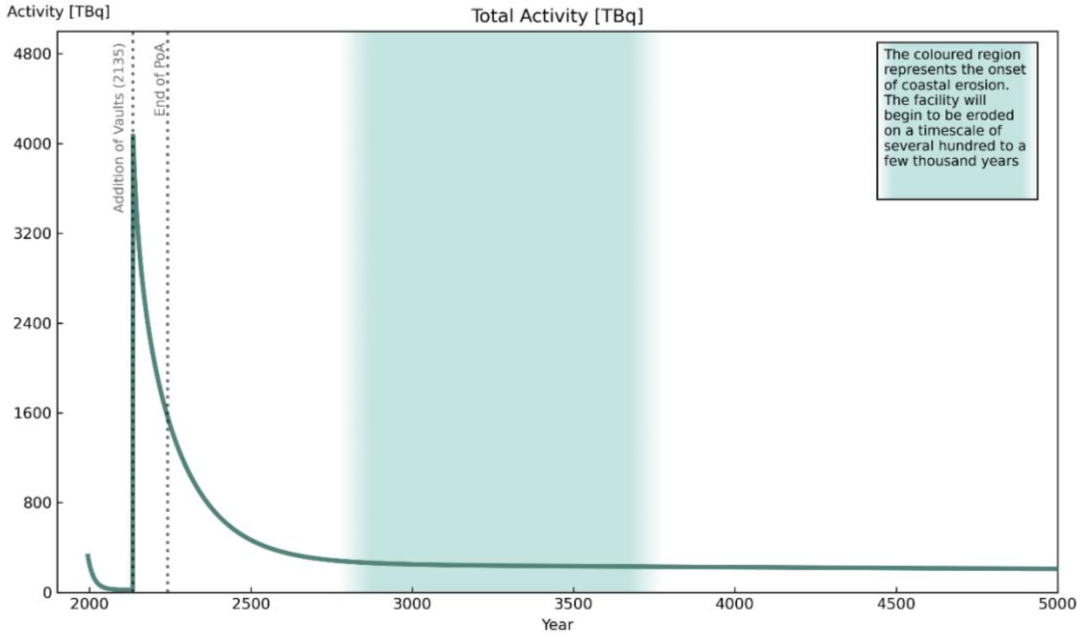


Figure 3.18: Evolution of total activity in the LLWR due to decay and ingrowth

3.6.1.3 Long-term Evolution and Performance

In the trenches, degradation of waste and leaching of more mobile and accessible contaminants began at the time of waste disposal. The interim cap over the trenches and the north-east cut-off wall have limited water infiltration, lateral flow into the waste, and lateral flow into the drain in the base of the railway cutting. The trench base drains have collected contaminated leachate, which is managed as described above. A proportion of the contaminated leachate will, however, have passed through the trench bases and percolated downwards to groundwater.

The final cap and complete cut-off wall will limit water inflows to the trenches throughout the assessment timeframe. The small amount of water that infiltrates the trenches will drain through the trench bases, carrying a small flux of radionuclides with it. At the end of the PoA, the waste will be substantially degraded, shorter-lived radionuclides will have decayed, and a fraction of the more mobile, longer-lived radionuclides will have been collected and discharged to the sea. Conditions in the trenches will be reducing, with approximately neutral pH [6]. Release of contaminants from the waste will then be controlled by the amount of infiltration flowing through the waste, degradation of the waste forms, sorption of contaminants on wastes and trench fill materials, and for a few contaminants by solubility limitation. Some chemicals disposed to the trenches, such as ethylene-diamine tetra-acetic acid (EDTA), may form complexes with radionuclides (and non-radiological contaminants) increasing their mobility.

In the vaults, the wastes are largely protected during the ‘open’ period by the containers and grout. Nevertheless, during the operational phase some degradation, especially of soft organic waste forms, will have begun and the ISO containers will have begun to corrode. As described previously, several actions are planned to improve protection of the containers from the weather, reducing leaching and corrosion of the outside of the containers. Sufficient water is expected to be available within the containers for corrosion of metal waste and the insides of the containers, and degradation of organic waste, to continue [6].

Once the containers are covered by the cap, conditions inside and outside the containers will be reducing. There will be little infiltration into the vaults through the final cap. Water infiltrating the vaults will be conditioned by the cap BES layer and corrosion products in the vaults to mildly to moderately alkaline pH. The pH in the gaps between container stacks is expected to be around pH 8 to pH 10 [6]. High pH conditions, around pH 12.5, will persist inside the containers. Low corrosion rates will limit release of contaminants present in the matrix of metal wastes, and the grout and metal corrosion products will be substrates for sorption.

While the cap geomembrane is intact, infiltration could be sufficiently low that most or all the water is consumed through anaerobic corrosion of metal waste and containers. Once the cap geomembrane fails there will be an increase in infiltration, but it will still be low. Infiltration that is not consumed by anaerobic corrosion will interact with the containerised waste before draining through the vault bases.

Infiltration may enter top of stack containers of existing and committed designs, leaching radionuclides from the waste. Some water may also flow into containers further down the waste stacks. Infiltration will be diverted down the gaps between stacks of future stronger containers with top of stack container protection units. Radionuclides may diffuse through perforations in the containers into the water flowing down the gaps. EDTA present in the vault waste, and isosaccharinic acid (ISA) formed from cellulosic waste under high pH conditions, may form complexes with radionuclides (and non-radiological contaminants), increasing their mobility.

If the permeability of the vault bases is sufficiently low, water levels in the vaults will rise to the 1 m level, when water will be passively directed into the sub-vault drainage blankets and drain to deeper geosphere pathways. This is discussed in more detail in reference [13].

Migration in groundwater

The cap and cut-off walls direct any leachate from the trenches and vaults downwards to the regional groundwater system. Contaminants moving down to, and in, groundwater flowing in the sediments and sandstones beneath the site may be sorbed, especially to clay minerals. Directing radionuclides into the regional groundwater system provides two safety functions [12]: radionuclides are isolated from the biosphere for as long as possible, and strongly sorbing radionuclides are contained.

The radionuclides of greatest concern for the groundwater pathway are long-lived and sorb weakly or not at all. Dilution in groundwater flowing beneath the repository provides a safety function for all radionuclides but is particularly important for the radionuclides of greatest concern. The groundwater that passes beneath the disposal area flows towards the coast to discharge to the marine environment below low water with some possibility of discharge through the foreshore sediments [65]. In either case, significant further dilution occurs.

A key possibility is that a well for domestic or agricultural use could be constructed between the repository and the coast, with consequent potential for exposures via drinking water and agricultural practices (leading to consumption of animal products and, or vegetables and fruit). No such well exists at present and most of the land between the disposal area and the coast is part of the SSSI, so housing or agricultural developments are unlikely in this area. Nevertheless, it is feasible that a water abstraction well could be utilised in the area between the repository and the sea. We take the view that we cannot rely on controls to ensure that such a well is not constructed. We note that such a well would provide the most direct access and exposure to potentially contaminated groundwater from the site.

Migration with gas

Bulk hydrogen gas will be generated from anaerobic corrosion of metal waste in the trenches and vaults, and metal containers in the vaults. Bulk carbon dioxide and methane will also be generated from microbial degradation of organic wastes in the trenches. pH conditions in the vault waste will be too high for significant microbial activity. However, small amounts of carbon dioxide and methane might also be generated in microbial niches on the surfaces of cellulosic waste. Some carbon dioxide may precipitate in siderite in the trenches and react with cement grout in the vaults forming carbonate [6]. The remainder of the carbon dioxide is likely to be microbially reduced to methane by microbes utilising hydrogen generated by anaerobic corrosion.

Trace radioactive gases may migrate through the cap by diffusion and be transported through the cap by advection within bulk gas. Potentially some bulk and trace gases could migrate through or under the cut off wall, and discharge to the ground around the perimeter of the cap [73].

The key radioactive gases are radon and C-14-bearing gases. The transport time through the cap will be sufficient for significant radioactive decay of radon. However, there will be negligible decay of C-14. Tritium is present in the waste but is of little concern to the long-term risks because of its relatively short half-life (12.3 years). The half-life of thoron (Rn-220) is too short for it to migrate through the cap. Isotopes of Cl and I can be volatilised. However, they are very soluble in water, so these gases are only likely to be present in very small quantities, and the cap will help to retain them in the repository.

Radon gas can potentially accumulate in houses (or other occupied buildings), resulting in doses to the occupants. C-14-bearing methane is expected to be metabolised to carbon dioxide by soil microbes. Some C-14 carbon dioxide will dissolve in soil porewater and some will emanate from the soil. C-14 can be taken-up in plants and vegetables by transpiration (roots) and photosynthesis (leaves). If the cap is being used for agricultural purposes, then C-14 will be incorporated into fruit and vegetables grown on the cap or products from animals grazing on the cap.

Longer-term evolution and coastal erosion

If the engineering performance is broadly consistent with expectations [13], infiltration into the repository will remain low until the repository is disrupted by coastal erosion. Water levels in the repository will be low, and all radionuclides leached from the waste will be transported into the regional groundwater system, and groundwater pathways to the coast [23].

Poor performance of the close engineering (especially early failure of the cap geomembrane and near maximum infiltration) are needed for water levels in the repository to rise to the level where radionuclides are released to shallow groundwater pathways [23] [65]. This is considered unlikely.

Once the repository starts to be disrupted by coastal erosion, waste will be exposed in the cliffs and will collapse from the cliffs forming a talus at the cliff toe. Large pieces of ISO containers and vault concrete are expected to be present in the talus. The ISO containers will tend to break up as they are eroded disgorging their contents, so large pieces of grout and waste will also be present. The ISO containers, grout and waste items will be broken down into smaller pieces by coastal processes until they are small enough to be transported offshore by wave action.

Understanding of coastal processes [8], conditions in the near field [6], the EPA [13] and evidence from analogue sites [25] [48] have been brought together to develop a conceptual model of what the eroding repository may 'look' like (Figure 3.19) and how the wastes will break up and disperse.

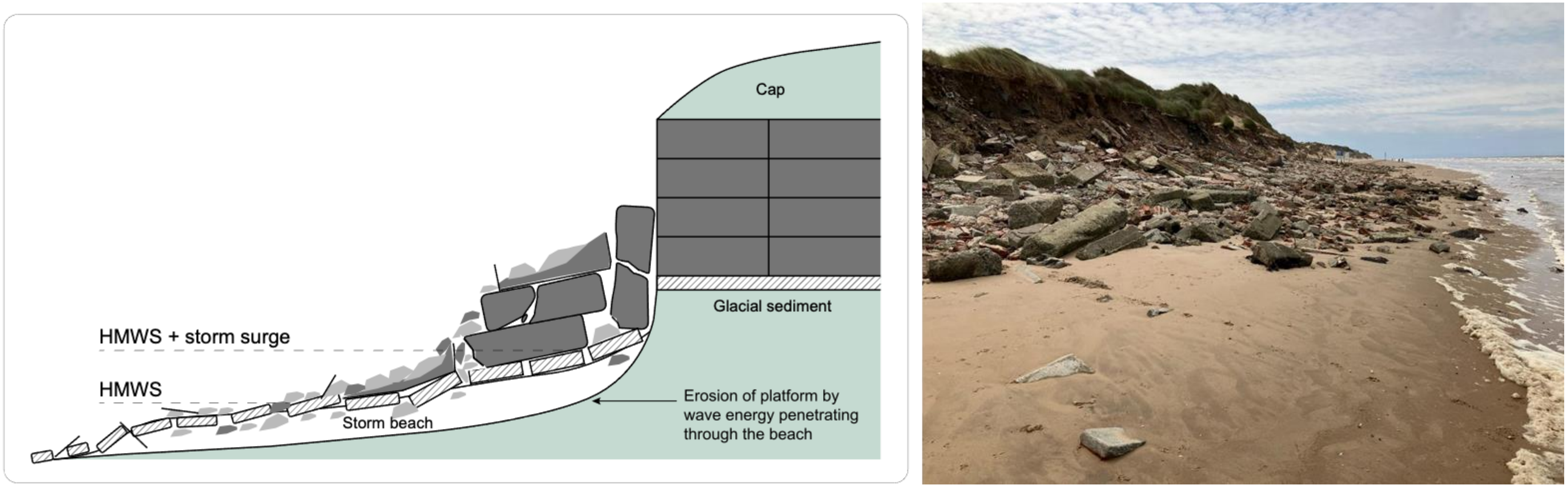


Figure 3.19: Conceptual model of the eroding repository left under faster rates of coastal erosion and an analogue site (demolition waste exposed by coastal erosion at Formby) [25]

Radionuclides associated with sand-sized materials will ultimately be transported into the regional sediment sink cell between St Bees and Ravenglass. Radionuclides associated with silt and clay sized materials will be transported further offshore and be deposited in the Eastern Irish Sea mudbelt. Radionuclides deposited in the sediment sinks will mix with and be buried by material eroded from the whole coastline. Seabed and shallow sediments in the sinks will contain radionuclides while erosion of the repository is ongoing. Once the repository is completely eroded, radionuclides will be buried by clean materials. Eventually the residual radionuclides will be buried to depths where they are isolated and potential environmental impacts are very low, and far lower than the peak potential impacts when the repository was eroding.

4 Radionuclides and Radiological Hazard

This section outlines the selection of radionuclides considered in the assessment calculations for the 2026 ESC. The screening process used to select the radionuclides is summarised and the resulting list of radionuclides to be considered in the assessments is presented. The screening and radiological hazard calculations are described in detail in reference [74].

4.1 Screening of Radionuclides

The assessments for the 2026 ESC include all radionuclides assessed in the 2011 ESC and any additional radionuclides identified as potentially important contributors to radiological impact. The screening process for determining which additional radionuclides should be included in assessments is outlined here.

4.1.1 Radiological Handbook Screening

Radionuclides present in the inventory in only very small amounts (less than one MBq) or with radiological properties that mean they will not contribute to the radiological impact of the repository (such as short half-lives, less than one year) are screened out according to the rules set out in the Radiological Handbook [57].

The screening process simplifies radionuclide decay chains by treating some radionuclides as being in secular equilibrium with their parents. The activity associated with, and doses arising from radionuclides treated as being in equilibrium is not lost from the assessment. Dose factors for radionuclides treated as being in secular equilibrium are added to the relevant parent radionuclide [57]. Figure 4.1 shows a schematic of simplified long decay chain radionuclides considered in the 2026 ESC [57]. Short-lived radionuclides that might be important during the PoA are discussed in reference [14].

The Radiological Handbook leaves the treatment of Po-210 at the discretion of assessment modellers. Po-210 has a short half-life, so it can be treated as in secular equilibrium with Pb-210. However, Po-210 has very different physicochemical behaviour to Pb-210 in the biosphere, which may require it to be treated explicitly. For example, in the groundwater assessment, the transport and uptake of Po-210 is modelled explicitly. For the coastal erosion and human intrusion assessments, where impacts are primarily from exposure to the exposed waste, it is reasonable to treat Po-210 as in secular equilibrium with Pb-210.

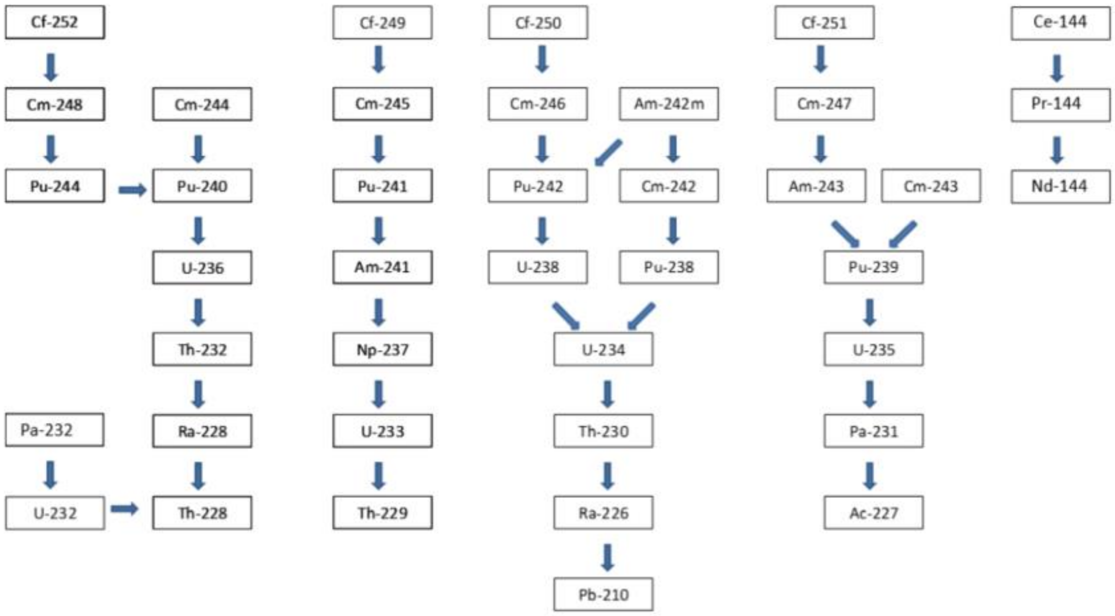


Figure 4.1: **Schematic of simplified long decay chain radionuclides - shorter-lived progeny are included in secular equilibrium with the parent [57]**

4.1.2 Radiological Hazard Index Model

A simple screening model (the Radiological Hazard Index model) is then used to help identify the potential radiological impact of the inventory and the important radionuclides present [74]. This is a very cautious, stylised model which calculates the annual dose that would hypothetically occur if the total radionuclide inventory of a given section of the repository is transferred to an area of ground, at the same concentration as in the original waste volume, and an individual is constrained to remain permanently within the area, while consuming water drained from the area and consuming contaminated foodstuffs. The radiological hazard is calculated as a function of time, only taking account of radioactive decay, not leaching or other losses from the inventory or environmental changes. This is not an expected or realistic situation can be used to:

- calculate a radiological hazard that may be useful in presenting the temporal evolution of the hazard of different waste inventories;
- indicate the pathways and exposure modes that are liable to be most important for different radionuclides;
- identify a set of radionuclides that should be treated in the assessment calculations of the pathways considered in the 2026 ESC.

The pathways considered are external irradiation, inhalation of dust, inadvertent ingestion of contaminated material (dust), water exposure, and gas exposure (Figure 4.2).

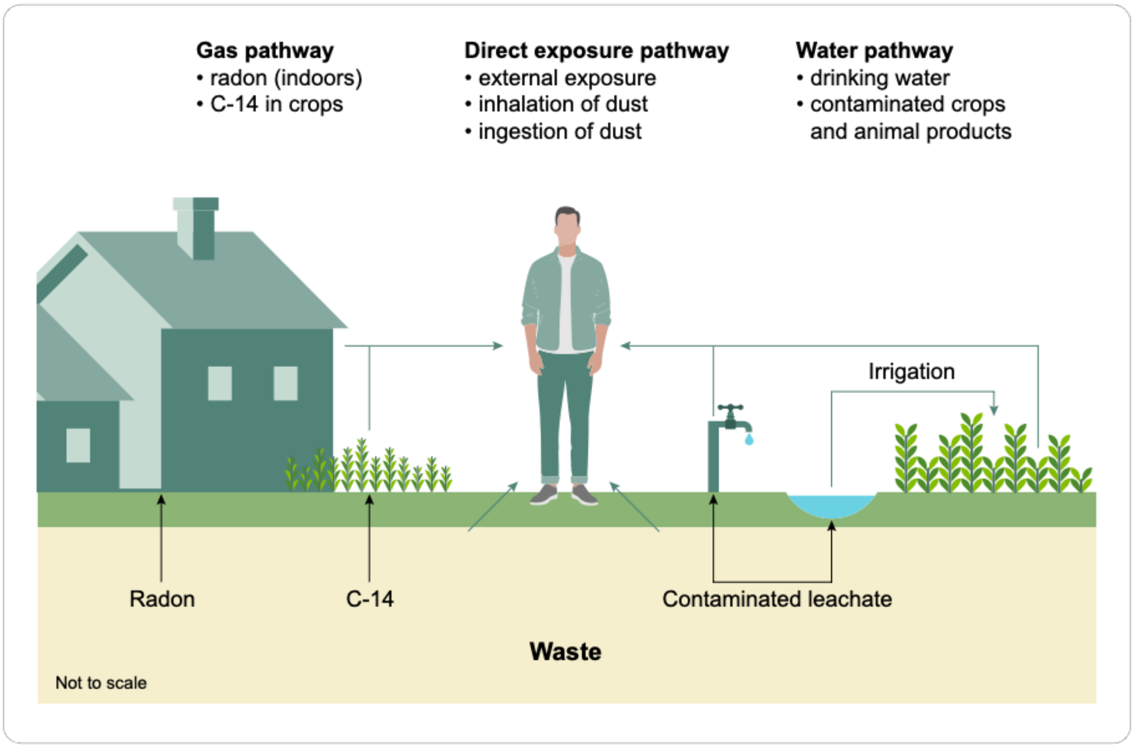


Figure 4.2: Schematic illustration of the radiological hazard model

The radiological hazard model is used to identify the radionuclides contributing to the top 99.9% of dose by any pathway at any time for both the trenches and vaults. All radionuclides which were identified and were not in the 2011 ESC assessment list were then reviewed to determine whether they should be included or excluded from the assessments. The radionuclides additionally included were: Eu-152, Ho-166m, Ni-59, Cs-134, Ru-106, Eu-154 and Sb-125. Evolution of the radiological hazard with time is described in more detail in reference [12].

4.2 Radionuclide Selection

The final list of radionuclides included in long-term assessment calculations in support of the 2026 ESC is shown in Box 4.1.

Box 4.1: Radionuclides retained for long-term assessments in support of the 2026 ESC

Radionuclides retained (includes both radionuclides in the disposal inventory and progeny radionuclides)

H-3, C-14, Cl-36, Ca-41, Co-60, Ni-59, Ni-63, Sr-90, Zr-93, Nb-93m, Nb-94, Mo-93, Tc-99, Ru-106, Ag-108m, Sb-125, I-129, Cs-134, Cs-135, Cs-137, Eu-152, Eu-154, Ho-166m, Pb-210*, Ra-226, Ra-228, Ac-227, Th-228, Th-229, Th-230, Th-232, Pa-231, U-233, U-234, U-235, U-236, U-238, Np-237, Pu-238, Pu-239, Pu-240, Pu-241, Pu-242, Pu-244, Am-241, Am-242m, Am-243, Cm-242, Cm-243, Cm-244, Cm-245, Cm-246, Cm-248

(53 radionuclides)

Notes: * Po-210 considered in secular equilibrium with Pb-210.

5 Assessment of the Groundwater Pathway

5.1 Assessment Scope

The scope of the groundwater pathway assessment is to address the potential risks from releases of radionuclides to groundwater, consistent with Requirement R6 of the GRA [19]. The assessment also considers protection of groundwater, consistent with supplementary guidance from the Environment Agency [20] [21].

Radionuclide releases to groundwater occur as rainwater infiltrates the waste, radionuclides dissolve in the water, and then the water drains into the ground. The repository engineering [10] aims to stop water entering areas of the repository that are closed, by covering them with a low permeability cap, and to capture all leachate from operational vaults. The containerised and grouted vault wasteform reduces radionuclide releases to operational vault leachate to very low levels.

At the end of operations, the whole repository will be covered by the final cap, which includes multiple barriers. Infiltration into the waste should be very low. Passive drainage minimises water contact with the waste. The containerised vault wasteform provides an additional barrier, as part of a multi-barrier system.

The cut-off wall is designed to prevent releases to shallow soils and groundwater by directing any water that does infiltrate the repository to the regional groundwater system. This is an optimised approach [10]. Previous assessments (and the results of this assessment) show the risks associated with releases to the regional groundwater system are lower than the risks associated with releases to surface soils and shallow groundwater (see Safety Functions provided by the regional groundwater pathway in Subsection 3.6).

The trenches do not have engineered bases or walls, so historically there have been releases to groundwater from the trenches. An interim trench cap was constructed to limit infiltration and since then releases have substantially reduced. Over the next decade, the northern part of the trenches will be covered by the first strip of the final cap, and the interim cap over the southern trenches will be replaced. This will further reduce infiltration into the trenches and releases to groundwater.

The assessment timeframe starts in 1950, prior to first disposals in 1959, as historic and current releases affect potential impacts now, during the remainder of the PoA, and after the end of the PoA. The PoA assessment [14] considers the potential impacts of releases to groundwater during the PoA. This report considers impacts after the end of the PoA.

The assessment finishes when the repository starts to be disrupted by coastal erosion. The groundwater pathway has been fully eroded at this time. The coastal erosion pathway assessment considers releases of water from the eroding repository to the coast [25]. The potential impacts are small relative to the potential impacts from direct exposure to the eroding waste.

5.2 Previous Assessments and Environment Agency comments

The groundwater pathway assessment approach was significantly revised for the 2011 ESC.[7] Key improvements for the 2011 ESC included groundwater flow models which provided improved quantification of groundwater pathway properties, and improved models of radionuclide release and transport in groundwater.

The Environment Agency's review of the 2011 ESC identified FIs that needed to be addressed and made recommendations for potential improvements. We have taken these recommendations into account when developing the groundwater pathway assessment. A full description of how they have been considered is provided in the *'Addressing Regulatory Requirements and Feedback'* report [18].

5.3 Approach in this Assessment

The groundwater pathway assessment approach for the 2026 ESC builds on the approach developed for the 2011 ESC. Substantial updates and refinements have been made to the assessment. The groundwater pathway assessment for radionuclides is consistent with the Hydrogeological Risk Assessment (HRA) [15], which uses the same assessment model.

The main components of the groundwater pathway assessment are as follows.

- Geological and hydrogeological conceptual models underpinned by site investigation and monitoring data [25].
- A site-scale groundwater flow model developed using ConnectFlow software [7] [65]. The groundwater model is used to test and build confidence in the hydrogeological conceptual model. The groundwater flow model is calibrated against present day hydrogeological data. The calibrated model is then used to project the impacts of repository engineering, climate change, sea-level rise and coastal erosion on water flows through the repository and groundwater pathways through the geosphere, using steady state calculations for important future time snaps. Groundwater pathway cross-sectional areas and flow rates calculated by ConnectFlow are output for use in the groundwater pathway assessment model (and HRA).
- Models of evolving biogeochemical conditions in the repository, including pH and Eh, developed using PFLOTRAN software [6]. Understanding from these models 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. However, we do not use PFLOTRAN outputs directly in the assessment model.

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[7] The history of assessments prior to the 2011 ESC is discussed in reference [89].

- The groundwater pathway assessment model implemented using GoldSim software.

The sources of information and flow of data between models is shown in more detail in Figure 5.1. Quantitative outputs from one model do not always directly feed into another model. Often expert judgement is applied to interpret the results from one model before feeding them into another model. The flows of data illustrated in Figure 5.1 are as follows.

1) Projections of climate change affect sea level rise and the timescales for, and mode of, coastal erosion of the LLWR site. These feed into the definition of appropriate climate scenarios, described in Subsection 3.5.
2) The principal effect of climate change on the biosphere relates to the rate at which land between the site and coast is eroded. While the climate will evolve over the assessment timeframe, the climate state will be sufficiently close to the current climate that it is appropriate to use the same constant biosphere parameterisation in each scenario.
3) In the assessment model, a key pathway is the well pathway, which considers the impacts arising from consumption and use of contaminated water abstracted from a well that intercepts the contaminant plume between the site and the coast. As the coast is eroded, the probability a well exists in this area reduces.
4) Biosphere data (e.g. habits data, sorption coefficients for biosphere environmental media, transfers factors and concentration ratios) are used to parameterise models to calculate the transfer of radionuclides through the biosphere and to calculate the incurred radiation doses and risks.
5) A regional-scale finite element model of groundwater flow implemented in ConnectFlow is used to calculate groundwater flows through the B3 lithofacies unit. These flows are used by the groundwater assessment model to calculate contaminant transport through the B3 unit.
6) A site-scale finite element model of groundwater flow implemented in ConnectFlow is used to provide boundary conditions for the Compartment Flow Model (CFM) within the near field and the B2 hydrogeological unit and to verify the flows calculated within the CFM
7) A PFLOTRAN model of the near field is used to understand the evolution of pH and Eh over the assessment time frame. This understanding informs desk-based studies which have identified sorption and solubility parameters for the expected chemical conditions. Calculations undertaken with PFLOTRAN have also been used to understand the effects of complexation with key organic species on contaminant mobility.
8) The solubility and sorption parameters are used as input parameters in the groundwater assessment model.

9) The EPA sets out the understanding of the evolution of the engineered barriers, which are used to identify appropriate engineering evolution assessment cases in the groundwater model. The evolution of each barrier affects the calculation of groundwater flow in ConnectFlow and water flows through the repository in the CFM.

10) The CFM is used to calculate water flows and levels in the repository near field and in the B2 lithofacies unit. These are used to determine the transport of contaminants.

11) For each climate scenario, hydrogeological models have been constructed which are used to calculate flows through the repository and the B3 unit at the present day and several future times. Flows and water levels in the repository are used to calibrate the CFM, while calculated flows through the B3 unit are input into the GoldSim assessment model to calculate contaminant transport through the B3 unit.

The groundwater assessment provides several outputs which are used in the ESC: concentrations of radiological (and non-radiological [15]) contaminants in groundwater and environmental media; calculations of doses and risks after the PoA; and outputs to inform uncertainty analysis and analysis of safety functions (the roles of different barriers).

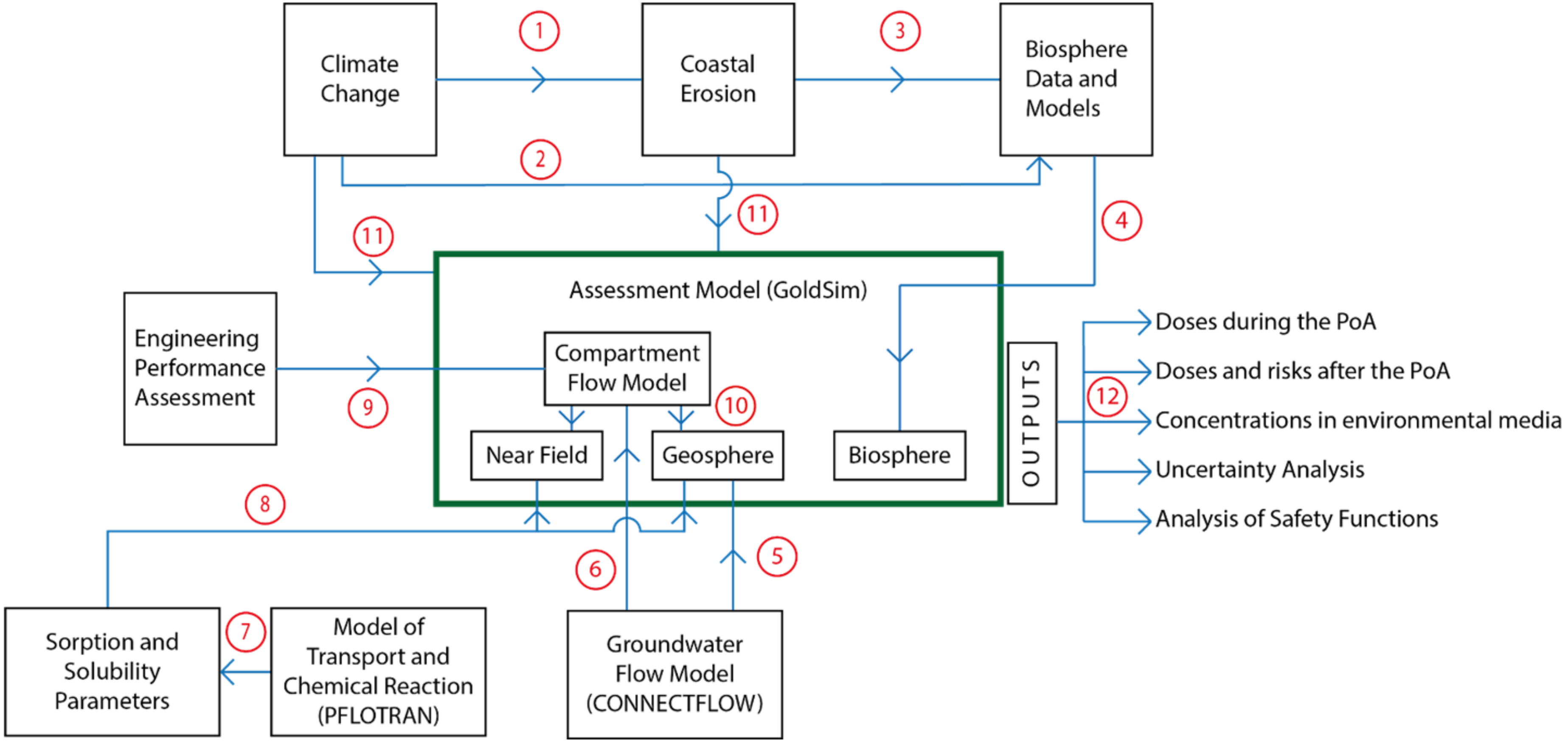

Figure 5.1: Sources of information and flow of data in the groundwater pathway assessment

The assessment model is used to calculate impacts of radionuclides on groundwater and risks to people for several biosphere pathways:

- abstraction and usage of contaminated water from a well between the repository and the coast (well biosphere pathway);
- natural discharge of contaminants to the marine environment (marine biosphere pathway);
- discharge of contaminants to the Drigg Stream (stream biosphere pathway);
- discharge of contaminants to the Irt Estuary (estuary biosphere pathway).

Radionuclide releases to the Drigg Stream post-PoA would only occur if water levels in the repository were to rise sufficiently for water to flow out through, or overtop, the cut-off wall. This is a low likelihood situation given the expected long-term engineering performance [66]. Our groundwater flow and assessment models confirm that extreme parameter values, reflecting very poor engineering performance, are needed for releases to the Drigg Stream pathway.

Radionuclides that discharge to the marine environment or the stream can be transported into the Irt Estuary.

Risks are calculated for a reference case and variant cases. It is uncertain whether a well would be drilled into the contaminated area. If this occurred, the radionuclide concentrations in the well water would depend on the location of the well. Therefore, the risks for the well pathway depend on the expectation value of radionuclide concentrations in the contaminated area and the probability that a well exists in the contaminated area.

All cases are evaluated deterministically. A PRA is also undertaken for the reference case, to explore uncertainties such as the cap performance. The expectation value of risk from the probabilistic calculation is used to derive radiological capacities. Use of radiological capacities to manage disposals to the repository is described in the '*Implementation*' report [17].

There have been improvements to our understanding of the evolution and performance of the repository compared with the 2011 ESC, and this is reflected in changes to the configuration and parameterisation of the assessment model. Important changes are listed below.

- Recognition that the cap will perform better and for longer than considered in the 2011 ESC. There will be less infiltration into the repository and lower releases to the groundwater pathway.
- Changes to the inventory, number of future vaults required, repository design and the range of waste types that could be disposed.
- Improved understanding of the long-term performance and evolution of different types of containers.

- Updated understanding of biogeochemical conditions in the repository, which affect radionuclide release and transport.
- Our assessment model for the 2011 ESC did not take any credit for the containers. We have retained this model but given our updated understanding of conditions in the near field, we have developed a variant model that includes diversion of infiltration down the gaps between waste stacks. This is termed the dual porosity model. The dual porosity model represents diffusion of contaminants through the grouted waste form into the water flowing down the gaps between waste stacks but does not take credit for the containers as a barrier to diffusion.
- We have improved our understanding of metal corrosion rates, radionuclide solubility and sorption, and the effects of complexants present in the waste (e.g. EDTA) and generated as waste degrades (e.g. ISA).

5.4 Assessment Model and Data

5.4.1 Conceptual Model of Radionuclide Release and Migration

5.4.1.1 Water Flows in the Near Field

While the cap geomembrane is intact, infiltration into the vaults is expected to be very low. Most of the infiltration is likely to be consumed through anaerobic corrosion of mild steel containers. There will be no flows of water through the containers and waste, or down the sides of the container stacks, to establish transport pathways for radionuclides in solution.

Once the cap starts to degrade, and infiltration into the repository starts to increase, water is expected to flow into existing and committed containers with damaged lids; and be drawn into all other containers by capillary suction through areas where the grout is exposed by grout ports, vents or local container corrosion. Once the grout has substantially re-saturated, water may start to reach the bases of the vaults and pond on the bases.

Water that enters the top of a stack of existing and committed containers is likely to find routes out of the bases of the damaged containers, e.g. tears between the container bases and sides panels and small perforations from corrosion. Some water may then flow down the sides of the container stacks, and some may flow into the underlying containers. The same situation is expected for each subsequent container layer down the stacks. The amount of water flowing through the containers would be greatest for the top of stack containers and would decrease down the waste stacks (Figure 5.2).

The waste types, amount of grout in the containers, and continuity of the grout around the waste items are very variable. The grout is expected to develop cracks as it sets and cures, and the number of cracks is expected to increase with time as the waste and grout degrade.

The grouted wasteform in existing and committed containers will be carrying some profile and cap loads. The grout is weak, so this is likely to result in additional cracking. Water would likely preferentially flow through the cracks in the grout and transport radionuclides in solution from the waste. Further interactions can occur as radionuclides are transported down through the waste stacks, and in water ponded at the bottom of the vaults (Figure 5.2).

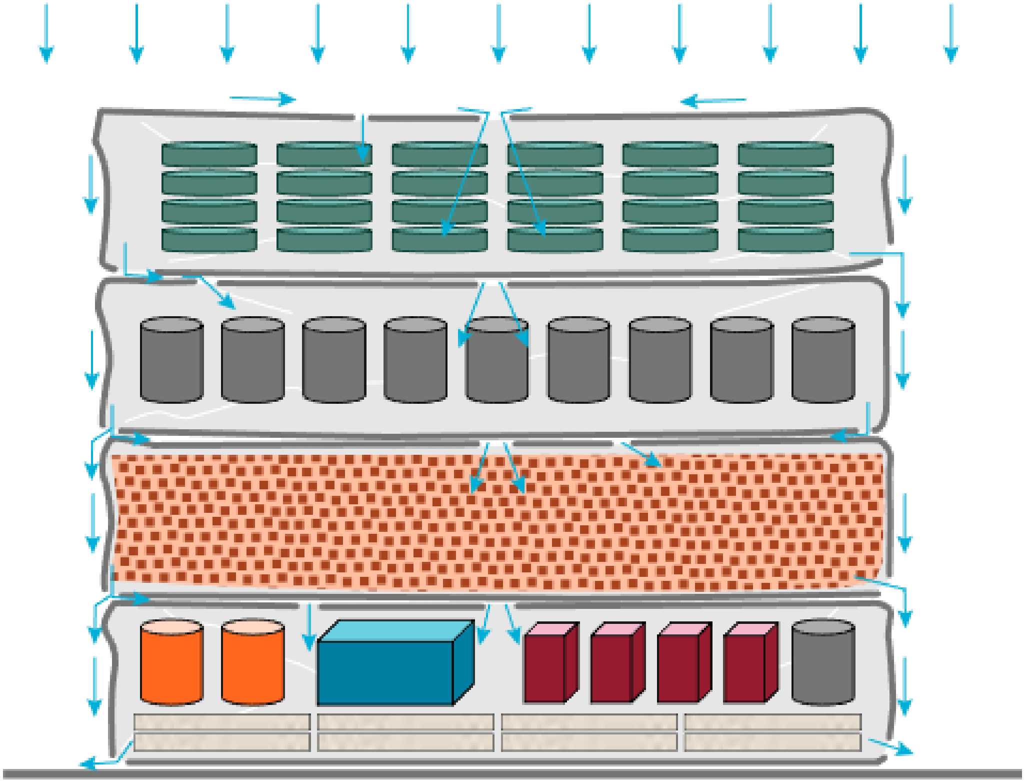

Figure 5.2: Conceptual model of post-closure water flows through a stack of deformed existing and committed ISO containers (red lines) illustrating variable waste types, unsealed grout ports, small tears, local corrosion perforations and extensively cracked grout (grey)

Future stronger containers would mostly be intact. CPUs at the top of waste stacks would direct water down the sides of the waste stacks (Figure 5.3). This would be in the form of a thin flowing film where the gaps between the stacks are narrow and granular void fill is not present in the gaps. Where the gaps are wider and granular void fill is present, there would be unsaturated flow through the void fill, with flow mainly occurring close to the surfaces of the containers.

There would not be flow of water down through stacks of future stronger containers. Radionuclides in solution could diffuse out of these containers where the containers are locally perforated and water flowing over the sides of containers contacts the grout. Local perforations could develop while the containers are exposed during the operational phase, e.g. where rainwater collects in crevices, and during the operational and post-closure phases where the container metal contacts more noble waste metals (galvanic corrosion), or in local areas where environmental conditions lead to higher general corrosion rates. Potentially, radionuclides in solution could also diffuse out of grout ports if they are left open, and there is liquid water present on the surfaces of the container lids. General corrosion of the containers is expected to be slow once the containers are protected by the cap, with little general corrosion occurring during the assessment timeframe [13].

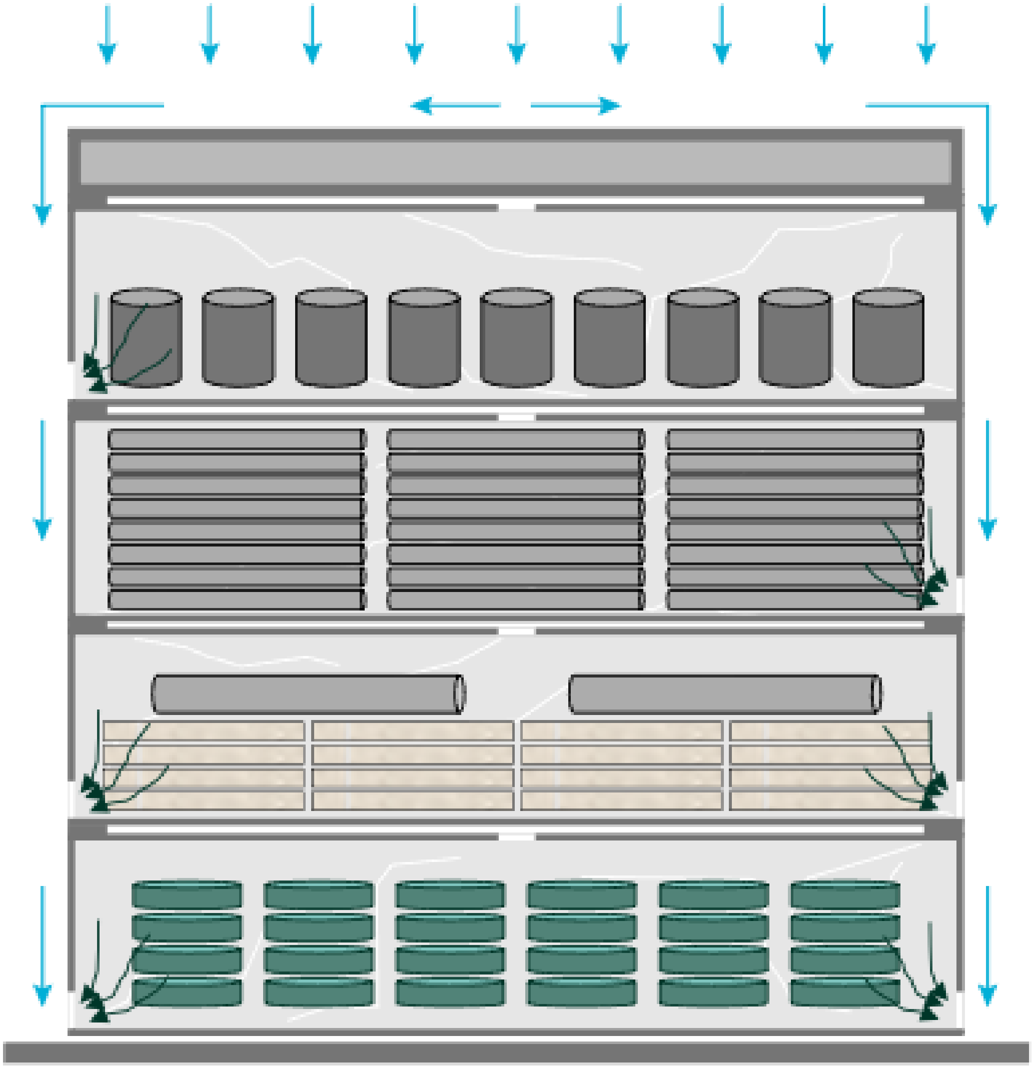

Figure 5.3: **Conceptual model of post-closure water flows over a CPU and down the sides of future stronger ISO containers (red lines), and diffusion of aqueous radionuclides from variable waste types through the cracked grout (grey) and small perforations in the containers**

As described for existing and committed containers, the waste types, amount of grout in the containers, and continuity of the grout around waste items, are expected to be very variable. The grout may be less cracked compared with existing and committed containers because the grouted wasteform is carrying much less load.[8] Overall, the performance of the grout as a barrier to diffusion of radionuclides in solution may be variable and low in some containers. However, in all containers, there would be a 'geometrical resistance' associated with

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[8] A small amount of load may be transferred into the wasteform by elastic bending of the container, and through friction as the container compresses under load.

diffusion to one or a few small perforations in the container. Diffusion into a small perforation concentrates radionuclides in the area close to the perforation. This reduces the concentration gradient, and therefore the diffusive flux, compared with the situation where the container is not present and radionuclides can diffuse through the full surface area of the grouted waste form. This is analogous to the situation in a geological disposal facility, where radionuclides are diffusing through engineered backfill or buffer into a water filled crack in a crystalline host rock [75] [76].

Once radionuclides have diffused out of the containers into water flowing down the sides of the container stacks, interactions can occur in the gaps between the waste stacks and as waters pond and mix in the bottom of the vaults.

Once the cap geomembrane has failed, the hydraulic conductivity of the vault bases is expected to similar to the infiltration rate through the cap, so the depth of water ponded on the vault bases will be small. However, if net[9] infiltration exceeds the hydraulic conductivity of the vault bases, then more water will pond on the bases. If the water level reaches 1 m above the vault base, water in the future vaults will flow over the vault walls to access the drainage blankets. Water in Vault 8 and Vault 9, which do not have drainage blankets, will be directed into the future vaults so it can access the drainage blankets.

If net infiltration exceeds the drainage capacity of the underlying ground, then water levels in the repository will rise until water starts to discharge to soils and shallow groundwater on and around the perimeter of the cap. This situation is unlikely. It could only be caused by cap infiltration rates towards the upper bound and, or substantial reduction in the drainage capacity of the underlying ground, as discussed below. There is not expected to be significant clogging of the drainage blankets [13].

If water levels rise sufficiently for water to discharge to soils and shallow groundwater on and around the perimeter of the cap, radionuclide concentrations in the discharging water would be lower than in water in contact with the waste. The water would comprise clean infiltration mixed with limited quantities of water that has contacted the waste (Figure 5.4). Radionuclides in any overtopping waters would likely be sourced from wastes higher stacked in the vaults.

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[9] Infiltration minus water consumed by anaerobic corrosion.

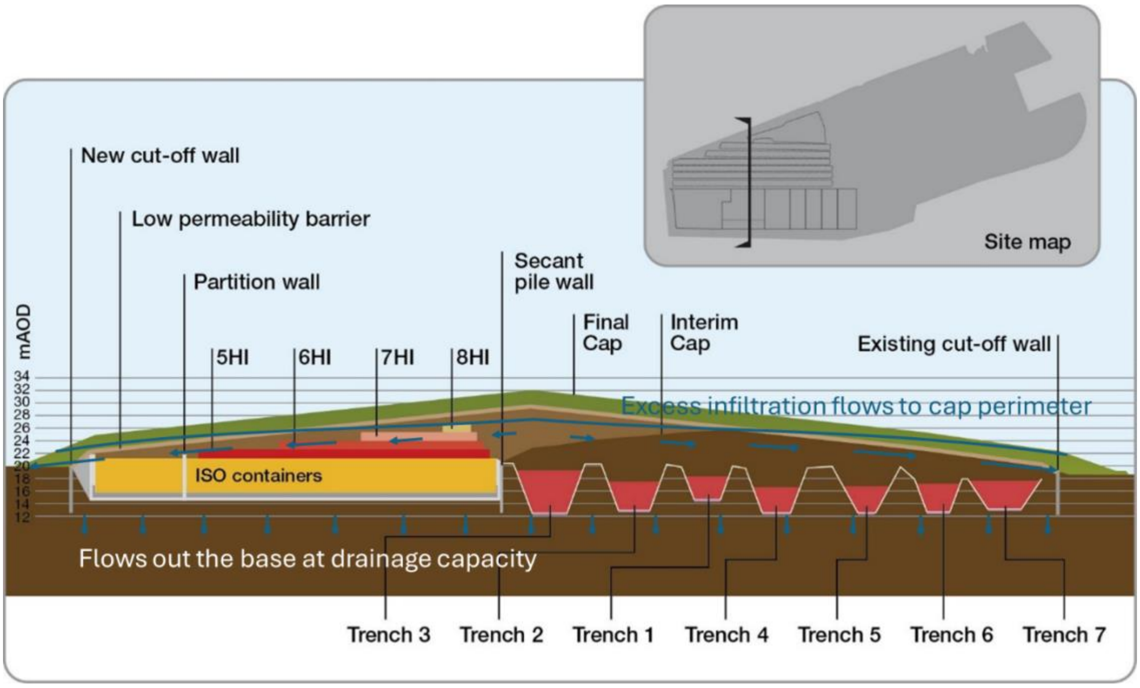


Figure 5.4: Conceptual model of overtopping flow and transport paths

5.4.1.2 Water flows in the Geosphere

Water drains through the bases of the trenches, existing vaults, and the drainage blankets below the future vaults, into the underlying sediments.

The underlying geology in the vicinity of the LLWR can be described in terms of 'lithofacies units' [25]. These lithofacies units define sections of the underlying geology that can be treated as single units within the hydrogeological modelling. In brief: unit B2 consists of clay-rich sediments and sands; unit B3 consists of thick sand and gravel deposits.

The trenches and vaults are constructed in the B2 unit (Figure 3.7). Water draining through the base of the repository will flow downwards through the B2 unit into the B3 unit. Water then flows laterally within the B3 unit, discharging at the coast.

While the engineered cap is performing well, groundwater recharge will be low below the footprint of the repository, so the upper parts of the B2 unit will be unsaturated below the footprint of the repository (Figure 5.6). As the engineering degrades and more water infiltrates the repository, the B2 unit below the repository will become increasingly saturated. As sea level rises, groundwater heads in the B3 unit will increase, and the elevation to which the B2 unit is saturated below the repository will increase. The effects of sea-level rise will be offset to some extent by erosion of the coast and movement of the groundwater discharge location inland.

If groundwater heads in the B3 unit rise to the level of the drainage blankets, the drainage capacity will be greatly reduced, and water levels in the repository may rise to the level where groundwater starts to discharge to soils and shallow groundwater on and around the perimeter of the cap. The magnitude of sea-level rise needed for groundwater heads in the

B3 unit rise to the level of the drainage blankets is large. A large rise in sea level would also result in high rates of coastal erosion. The repository may be disrupted by coastal erosion before groundwater heads rise sufficiently to impact the performance of the drainage blankets.

Chemical processes

The wastes disposed in the trenches and vaults are present in several different physical forms. For example, metallic wastes may contain radioactivity at the surface of the metal or embedded within the volume (matrix) of the metal. For the assessment of the groundwater pathway, the main interest is in the fate of the radionuclides, rather than the wasteforms.

When water contacts the wasteforms, the radionuclides at the surface of the wasteforms can potentially enter solution and then be transported away in flowing water. The timescales over which radionuclides in the matrix of materials enter into solution depends on the nature of the wasteform. Metallic wastes may corrode and release radionuclides relatively slowly. Other wasteforms, for example irradiated graphite, will release radionuclides into solution more rapidly.

The release and transport of radionuclides will depend on whether conditions in the trenches and vaults are saturated or unsaturated, and on the ambient chemical conditions.

In the unsaturated regions of the trenches and vaults, only some of the waste will be in contact with water. The fractions of the waste that are in contact with water will depend on the degree of water saturation in the unsaturated zone. Radionuclides will only be released from waste that is in contact with water.

The understanding of the biogeochemical evolution of the near field is informed and quantified through use of models developed in PFLOTRAN [6], and comparison with older models built using the GRM software for the 2011 ESC [6] [77]. Key results for the groundwater pathway assessment include confirmation that the prevailing chemical conditions in the trenches will be reducing with neutral pH, while highly alkaline conditions (around pH 12.5) will be maintained in most containers throughout the assessment timeframe. Conditions will be less alkaline (pH 8 to 10) outside the containers [6]. High pH, reducing conditions will minimise corrosion rates of most metals (aluminium is an exception), and therefore release of radionuclides from the matrix of metal waste.

Once radionuclides have been released and have entered solution, the concentrations of some radionuclides in solution will depend on the local solubility limits for those elements, which will depend on the local chemical conditions. The concentrations will also vary with degree of sorption to solid substrates present in the trenches and vaults. Important substrates include soil, grout, concrete, the wastes themselves, and in the vaults the products from the corrosion of the ISO containers. EDTA present in the trench and vault waste, and ISA formed from cellulosic waste under high pH conditions in the vaults (Subsection 3.6.1.3), can form complexes with some radionuclides increasing their mobility.

As water from the repository enters the geosphere and mixes with groundwater the concentrations of EDTA and ISA will decrease. Some dilution occurs in the B2 unit under the footprint of the repository, but most dilution occurs as water flows from the B2 unit into the B3 unit. It is difficult to predict whether complexes will be stable or disassociate. The behaviour is expected to depend on the nature of the ion being complexed, the complexant, and the presence of competing ions such as iron, calcium, and nickel. The pH of water from the vaults will decrease, and microbes could metabolise ISA.

Colloids are not observed to have a significant impact on contaminant transport. They are not expected to be stable in the grout and transported easily in the near field [78].

Some of the radionuclides, notably C-14, may be released in gaseous form. These gases may dissolve in water, or they may migrate upwards through trench and vault materials and be released to the ground surface, as assessed in Section 6.

Radionuclides in solution will be transported with the water flows within and between the trenches, vaults and other components of the LLWR. A more detailed discussion of the near-field processes within the trenches and vaults at the LLWR can be found in reference [6].

5.4.2 Hydrogeological Modelling

5.4.2.1 Conceptual Model

The objective of the hydrogeological model is to calculate regional water flows and local (site-scale) water flows through the repository. To achieve this, it is necessary to define appropriate model domains. The model domains for the calculation of regional and site-scale flows are shown in Figure 5.5:.

The boundaries of the regional domain were taken to correspond to locations where physical boundary conditions for the groundwater flow model can be suggested on the basis of the topographic and hydrological features of the terrain. The model is roughly rectangular with the sides of the model aligned parallel and perpendicular to the coast. All onshore boundaries are about 5 km from LLWR, and the offshore boundary extends about 2 km from the coast.

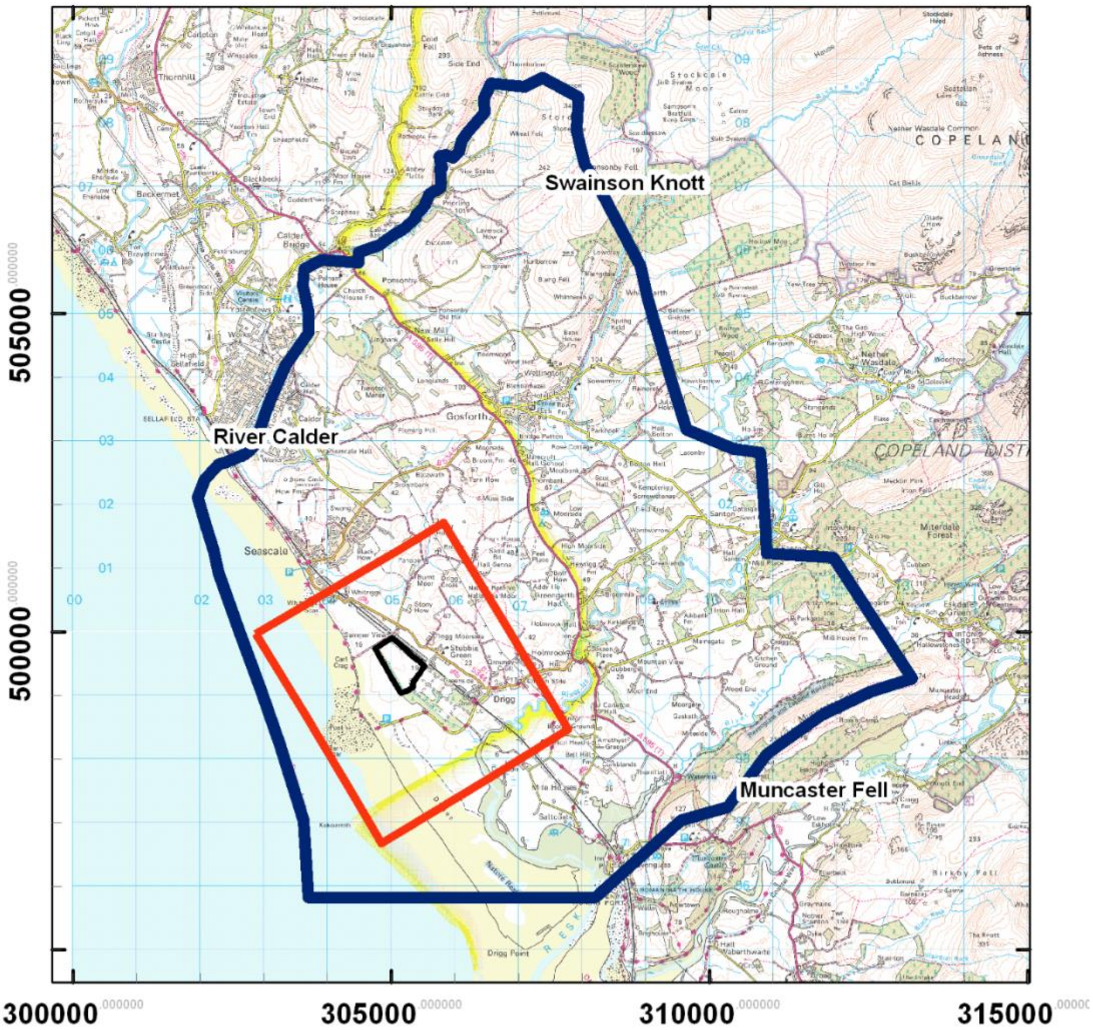


Figure 5.5: Location of LLWR (black), and the boundaries of the regional submodel (dark blue) site-scale submodel (red) used in the flow modelling.

The regional hydrogeological model needs to accommodate observations that arise from data collected from many sources over many years. The key observations are:

- groundwater flow is primarily in the Quaternary deposits and the upper part of the bedrock;
- the bedrock and Quaternary deposits are generally fully saturated to close to the ground surface;
- flow in the regional groundwater is generally towards the coast.

The first of these conclusions is supported by groundwater flow modelling carried out within the Nirex programme [79]. The Nirex models represented coupled groundwater flow, transport of heat and transport of salinity, and so accounted for the main processes that affect groundwater flow. It was found that a better match to the observed temperatures and temperature gradients was obtained from models in which the upper part of the sandstones (within about 70 m of the top of the sandstones) has significantly greater hydraulic conductivity than the lower part of the sandstones or lower rocks.

The second is based on observations of heads in boreholes, both those drilled for LLWR in the vicinity of the LLWR and those drilled further afield for the Environment Agency.

The third arises because the flow at depth at a location is related to the average topography in the vicinity. Thus, at depth, there is a general flow from the high ground inland towards the lower ground near the coast. This implies a lowering of the groundwater head when moving from inland towards the coast, and this is confirmed by measurement.

Further details of the hydrogeological data and model are presented in reference [7] [65].

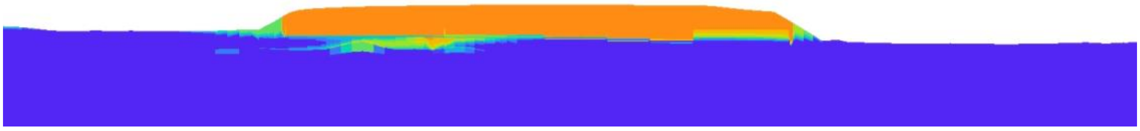
5.4.2.2 Numerical Modelling

Finite-element models of the flow in the region of interest were developed using ConnectFlow [7] [65]. The modelling was carried out on two scales: a relatively coarse grid regional-scale sub-model and a site-scale sub-model with finer elements. The boundary conditions for the site-scale model were obtained from a calculation with the regional model. The use of regional- and site-scale models in this way enables practicable calculations of the flow in the whole region, with a detailed representation of the region of primary interest in the immediate vicinity of the site.

The site-scale model includes all the key engineered components [13], but with simplified detail. Although the ConnectFlow model calculates water levels and flows in the repository these are not used directly in the assessment model. Rather, the results from the site-scale modelling were used as part of the calibration and checking of the Compartment Flow Model, see below. They were also used to inform the EPA and other pathway assessments.

Evolution of water saturation in the vaults, modelled using ConnectFlow, is illustrated in Figure 5.6 for the Reference Emissions Scenario. The model assumes the cap geomembrane is intact at all time snaps, so infiltration into the repository is low. The vault bases, drainage blankets and geosphere immediately below the repository are unsaturated throughout the assessment timeframe. The combined effects of sea-level rise and coastal erosion lead to no significant change in desaturation below the repository during the assessment timeframe.

2130 AD, Reference climate



2230 AD, Reference climate

2430 AD, Reference climate

3280AD, Reference climate

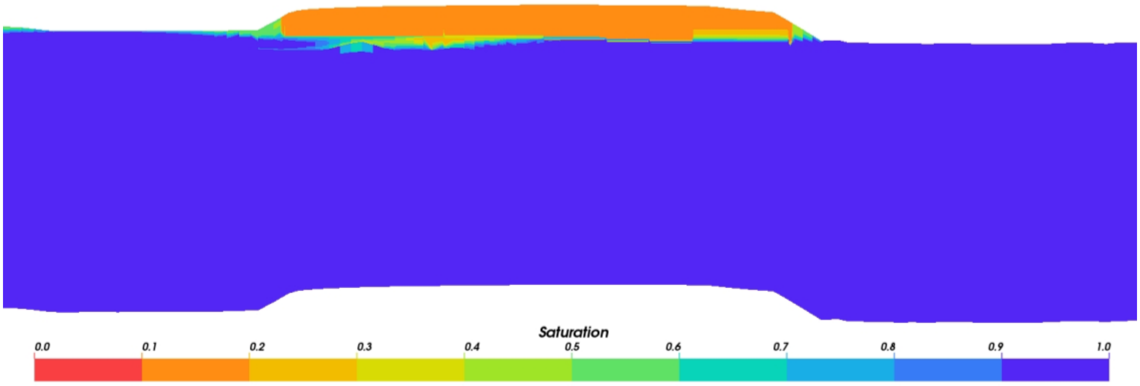


Figure 5.6: Evolution of water saturation on a slice through the long axis of the vaults for the Reference Emissions Scenario [65]

The primary outputs from the numerical modelling are the flow path areas, thicknesses and flow rates in the B3 unit from below the repository to the point of discharge. These outputs are fed into the assessment model, as described below.

Pathlines in the B3 unit calculated by the ConnectFlow model are shown in Figure 5.7 and Figure 5.8 [65]. Each pathline shows the direction of movement of a particle released into groundwater from a location in the repository until is discharges to surface water, i.e. at the coast. Most pathlines discharge at the outcrop of the B3 lithofacies unit, which is below mean low water.

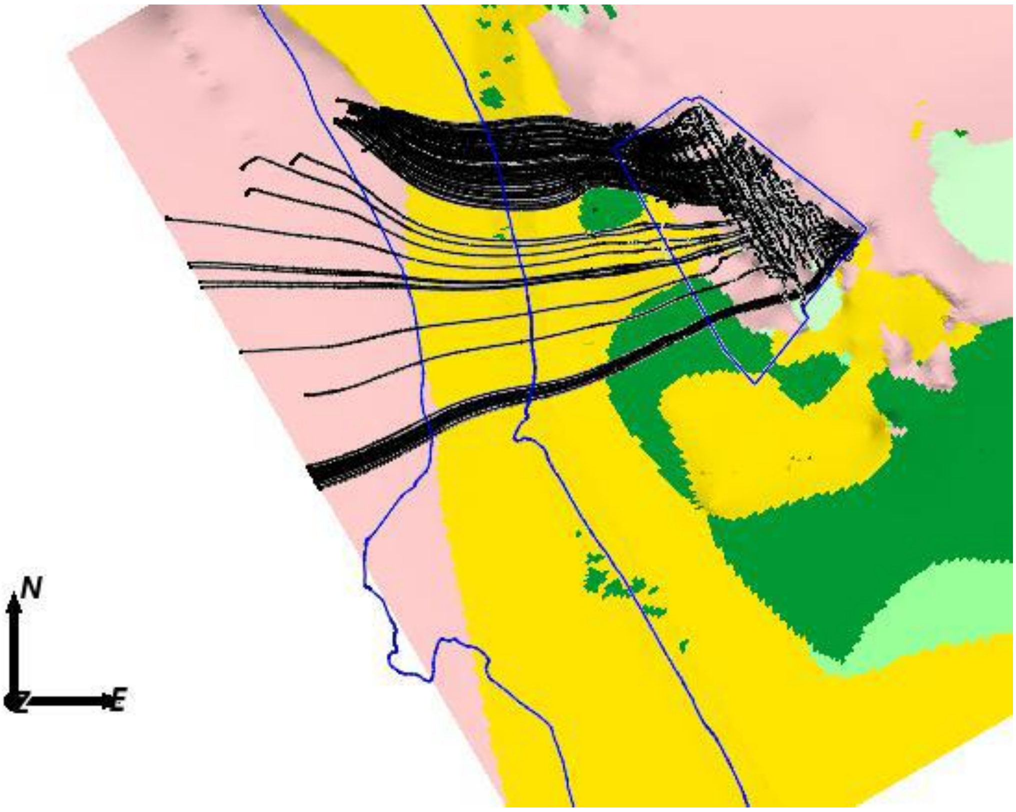


Figure 5.7 Plan view of present day pathlines from starting positions in the trenches shown over the lower lithofacies units (pink is sandstone, yellow is B3SS, green is D) for the 2025 hydrogeological model

Figure 5.8 Section view of pathlines from starting positions in the trenches shown passing through the lithofacies units (blue is B2, bright yellow is B3SS, light brown is B3), vertical exaggeration 5:1, for the 2025 hydrogeological model[10]

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[10] The geological and hydrogeological conceptual models divide B3 into B3SS and B3. B3 and B3SS are represented in the ConnectFlow model. B3SS has lower gravel content than B3 and more sand, and therefore typically lower hydraulic conductivity. B3 and B3SS are not differentiated in the assessment model.

5.4.3 Overview of the Assessment Model

The assessment model is implemented in GoldSim software. The model represents the whole lifetime of the repository, from 1950, prior to first disposals in 1959, until the repository starts to be disrupted by coastal erosion or is inundated. The model represents incremental disposal of waste, and calculates radioactive decay and ingrowth, radionuclide releases and transport, and the resulting amounts and concentrations of radionuclides in the repository, geosphere and biosphere (Figure 5.9) throughout the lifetime of the repository. Radionuclide concentrations in the geosphere and biosphere are used to calculate doses and risks to people and impacts on groundwater.

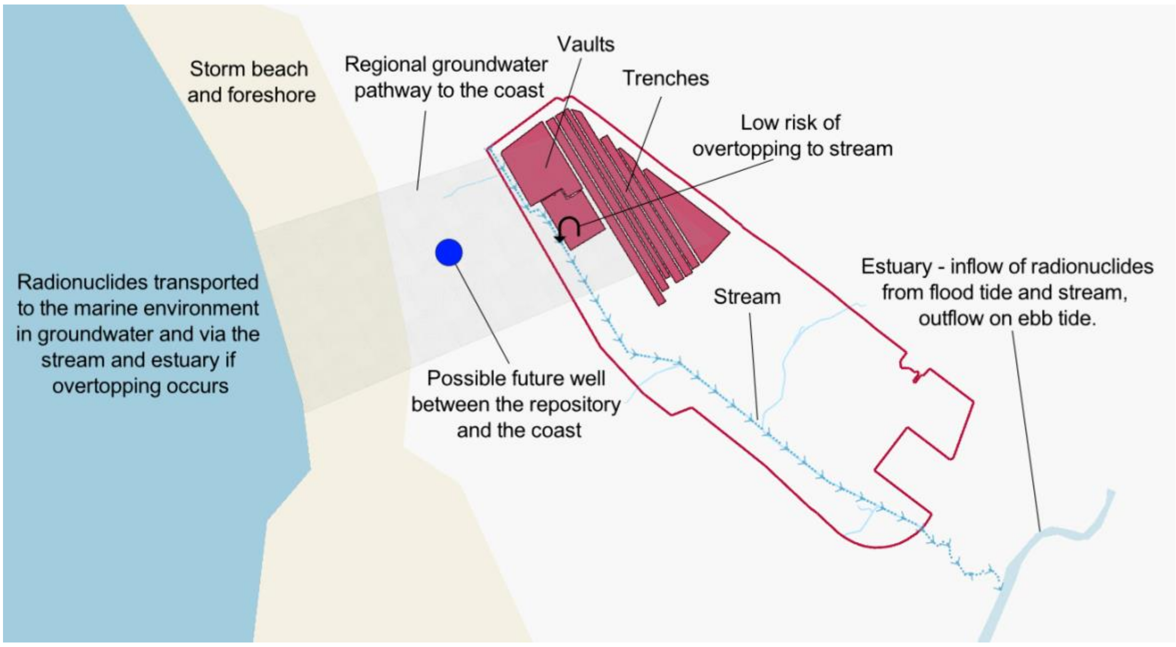

Figure 5.9: Simplified schematic of major features and flow paths in the groundwater assessment model

The assessment model represents the most important features, events and processes described in the conceptual model for potential impacts on groundwater and risks to people. The main features are represented by discretising the repository, geosphere and biosphere into volumes in which conditions, material properties, transport processes, and radionuclide concentrations are expected to be reasonably homogeneous and are treated as such in the assessment model.

Key processes included in the model are:

- time varying 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 and flows within the repository;
- release of contaminants from the waste to porewater subject to solubility limitation;

- advective transport of contaminants through the repository, subject to reversible linear sorption, and accounting for the effects of complexing agents on contaminant mobility;
- advective transport of contaminants in the B2 unit, where contaminants are subject to reversible linear sorption and complexation, which affects mobility;
- horizontal transport of contaminants in the B3 unit, where contaminants are subject to reversible linear sorption and complexation, which affects mobility.

Embedded in the GoldSim model is a compartment flow model (CFM). This is used to calculate water levels and flows in the repository and B2 unit. These water levels and flows are then used in the radionuclide release and near-field transport calculations. The compartment flow model calculates flows of water from the repository into deeper geosphere pathways and overtopping to surface streams if relevant conditions develop. Flow data for the deeper groundwater pathways (in the B3 unit) are provided by ConnectFlow, as described in Subsection 5.3.

5.4.4 Assessment Model Representation of the Near Field

5.4.4.1 Compartment Structure

The CFM discretises the near field and surrounding B2 unit into compartments and calculates water levels within the compartments and flows between the compartments. The assessment model uses the same compartment structure as the CFM for transport calculations. Additional compartments are needed to represent release processes, as described in the next subsection. The assessment model uses water levels and flows calculated by the CFM to compute the release and transport of radionuclides, both within and out of the repository. Calculated water levels in the repository are used to determine which wastes are saturated, and which are unsaturated. The approach is unchanged from the 2011 ESC.

The CFM represents the main engineered features of the repository (Figure 5.10 and Figure 5.11):

- The cap – an engineered barrier to infiltration.
- Cut-off wall – a low permeability wall that surrounds the repository. Note the barrier provided by the cut-off wall is represented in the CFM, but the wall is not represented using compartments.
- Vaults 8, 9 and future vaults – which have different designs and hydraulic properties, for example the base of Vault 9 is less permeable than either Vault 8 or the future vaults, and the vault walls vary in height and material.
- The trenches - existing waste disposal units with no engineered walls and a natural clay base. From Trench 3 onwards, bentonite was rotovated into the bases of the trenches where the natural clay was absent [80].

- Profiling – the profiling material lies between the top of the disposed wastes and the cap.
- B2 lithofacies unit – the repository is excavated into the B2 unit, which continues below the repository.

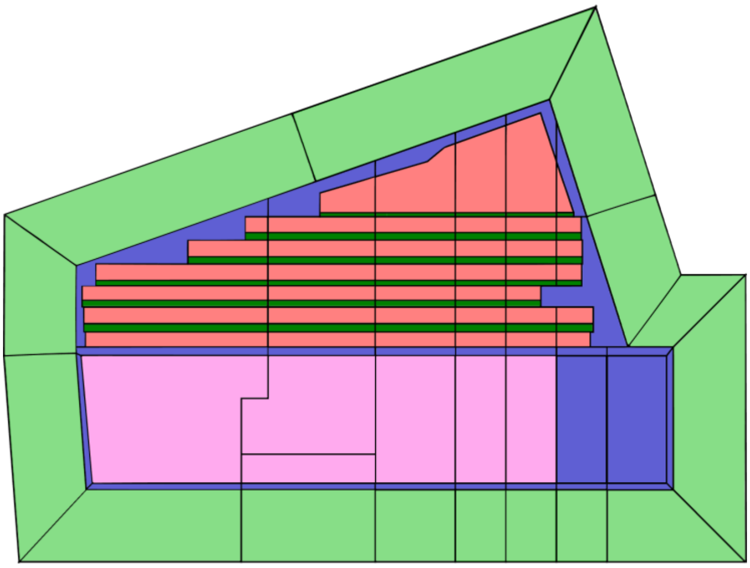

Figure 5.10: Discretisation of the repository and surrounding geology in the groundwater pathway compartment flow model and contaminant transport model [23]

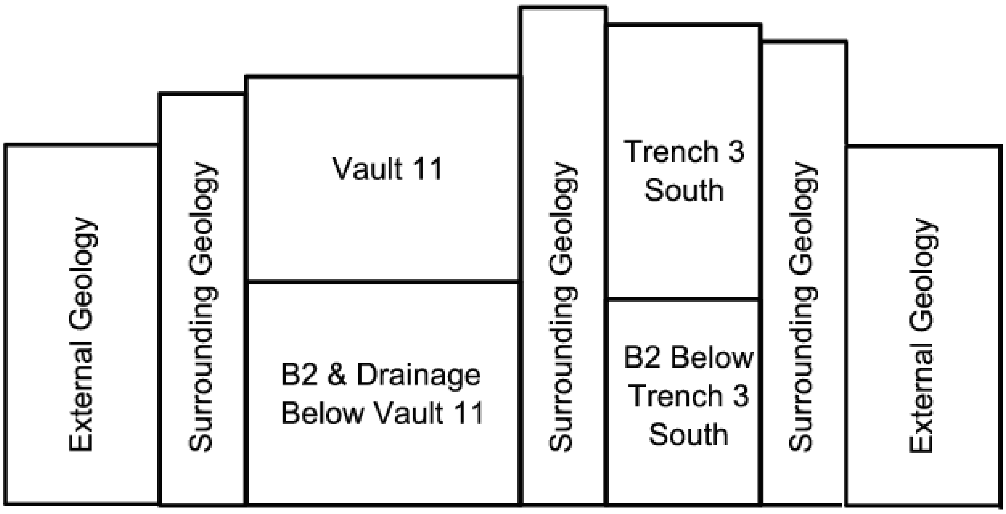

Figure 5.11: A schematic vertical west-east cross-section through the Compartment Flow Model at Vault 11 [23]. The External Geology compartments are outside the cut-off wall. The vertical dimensions are not to scale, and only Trench 3 has been included for clarity. Vaults and trenches have two compartments in each stack, elsewhere there is only one compartment per stack. Each compartment can have multiple material layers.

Representing the vaults individually captures the major spatial heterogeneities in their radionuclide inventories. Mixing of waters in the vaults, drainage blankets and groundwater pathway will reduce heterogeneities in radionuclide concentrations, so there is no benefit to representing vault disposals in more detail, even though more detailed inventory information is available for existing disposals.

In the assessment model, the radionuclide inventory in each trench is distributed so concentrations are uniform along its length. Post-PoA, flows are predominantly downwards through the bases of the trenches, so the groundwater pathlength to the coast is different for each trench. Each trench is discretised along its length into sections corresponding to the adjacent vaults (Figure 5.10). This is to support representation of strip capping and flow calculations using the CFM, rather than to calculate spatial heterogeneities in radionuclide releases to the groundwater pathway.

5.4.4.2 Near-field Processes

The assessment model represents the following near-field processes:

- release of radionuclides from saturated and unsaturated waste;
- transport of radionuclides in flowing waters;
- radionuclide sorption;
- solubility limitation of selected radionuclides;
- complexation with EDTA disposed in the trenches and vaults, and complexation with ISA generated from degradation of cellulose in the vault waste.

Material properties and radionuclide concentrations are assumed to be homogeneous in each compartment. Only advective transport is considered in the reference case. The reference case takes no credit for the containers, so advection is expected to control the peak radionuclide releases and fluxes. The fluxes of radionuclides from one compartment to another are equal to the water flow rate multiplied by the radionuclide concentrations in water in the source compartment.

Saturated and unsaturated wastes are represented using separate compartments. The amounts of waste that are saturated and unsaturated, and therefore which compartment the waste is associated with, and the sizes of the compartments, change as a function of the water levels calculated by the CFM.

All radionuclides are assumed to be immediately available on contact with water. This is a cautious simplifying assumption, as radionuclides in the matrix of waste items will only become available slowly, for example as the waste degrades.

In the saturated waste, water contacts the full pore volume. Therefore, all the radionuclide inventory is available for transport from the saturated waste. In the unsaturated waste, water only contacts a fraction of the pore volume (related to the degree of saturation). Therefore, only a fraction of the radionuclide inventory is available for transport from the unsaturated waste. The fraction of the radionuclide inventory that is available for transport from the unsaturated waste is represented in the assessment model by representing the unsaturated waste using two compartments. Radionuclides in the 'bound' compartment do not contact water and are not transported. Radionuclides in the 'unbound' compartment do contact water and are transported.

Radionuclides are transported from the unbound unsaturated waste compartments into the unbound saturated waste compartments. Radionuclides in the unbound saturated waste compartments are either transported laterally within the repository or downwards through the base of the repository.

The waste stacks and gaps between the waste stacks are not represented separately in the reference case, uniform flow model. The model does not take any credit for diversion of water around the waste by the containers. All the infiltration into the vaults flows through the waste, leaching contaminants.

Radionuclide sorption is modelled as linear, equilibrium, reversible sorption. In the trenches, sorption to soil materials is considered. In the vaults, sorption to grout is considered. Sorption to other materials (e.g. concrete, corrosion products) is neglected.

Solubility limitation is considered explicitly for isotopes of technetium, uranium and radioactive isotopes of elements that are also present as non-radiological contaminants [15], such as lead. All other isotopes are assumed to not be solubility limited. This simplifying assumption may overestimate fluxes of some radionuclides.

Complexation of radionuclides is represented by factors that reduce sorption and increase solubility (noting that solubility enhancement is only relevant for isotopes that are solubility limited). The factors depend on the concentration of the complexant and the radionuclide. Only some radionuclides form complexes.

EDTA has been detected in trench leachate. The amount of EDTA in the trenches is unknown. The assessment models assume EDTA is present in the trenches at the measured concentrations throughout the assessment timeframe. In the model there is no depletion of EDTA, so this is a cautious simplification, i.e. the models are expected to overestimate releases of complexed radionuclides to groundwater.

Estimates of the amount of EDTA disposed to the vaults and in future disposals are available [81]. Therefore, the concentration of EDTA in vault porewaters is calculated in the assessment models. The calculated concentrations decrease over time as EDTA is leached from the vaults. Therefore, the magnitude of solubility enhancement and sorption reduction decreases with time for the affected radionuclides.

The pH in the containers in the vaults is expected to be high enough for ISA to form from cellulosic waste. The high pH inside the containers will inhibit microbes that could degrade ISA [6]. The concentration of ISA in the vault porewaters has been estimated. Like EDTA in the trenches, the concentration of ISA in the vaults is assumed to be constant. The pH in the containers is expected to be high enough for ISA to form throughout the assessment timeframe, and low rates of cap infiltration will limit leaching of ISA. However, we do not take credit for the diminishing amount of cellulosic waste remaining and from which ISA could form. Therefore, this is a cautious approach. We also neglect the potential for ISA to degrade in the gaps between the waste stacks where the pH is lower than in the containers.

In the vaults, solubility enhancement and sorption reduction factors are chosen based on the higher of those for EDTA and ISA, i.e. the complexant expected to have the greatest impact on contaminant mobility. This is a reasonable approach as one complexant is likely to dominate, by virtue of being present in higher concentration or complexing a given element more strongly.

The reference case model does not represent the barrier to radionuclide release provided by the containers. The variant dual porosity model assumes infiltration is directed down the gaps between the waste stacks, and radionuclides diffuse through the grouted waste into the water flowing down through the gaps. The full area of the waste stacks is available for diffusion. As the containers would be largely intact [13], the area for diffusion would be much smaller than assumed in the model. Diffusion would be limited to the area of tears and damaged lids (in surcharged containers) and small corrosion penetrations (in all containers). Although this aspect of the dual porosity model is less cautious and more realistic than the reference case model, the model is still cautious. For example, we treat the gaps between the container stacks as a zero-concentration boundary. This overestimates the diffusive fluxes of radionuclides from the waste into the gaps.

The thickness of the grout barrier is variable, as shown schematically in Figure 5.2 and Figure 5.3. This is accounted for in our assessment models by selecting an effective diffusivity for the grouted wasteform that is much higher than the expected effective diffusivity of fresh, uncracked grout [82].

Chemical conditions (Eh, pH) in the trenches and vaults are assumed to remain constant throughout the assessment timeframe. Our near-field models [6] show this is a reasonable simplification.

If the CFM calculates that water levels in the repository rise sufficiently for overtopping to occur, then leachate is assumed to be released from the relevant trench or vault compartment(s). This approach is expected to substantially overestimate the radionuclide fluxes, as much of the overtopping water is expected to comprise clean water that has not contacted the waste (Figure 5.4).

5.4.5 Assessment Model Representation of the Geosphere

Radionuclide transport in the geosphere (B2 unit and B3 unit underlying the repository) is also calculated in the GoldSim groundwater assessment model.

B2 Unit

In the assessment model, the B2 unit below each repository, surrounding and external geology compartment (Figure 5.10 and Figure 5.11) is discretised into a stack of five vertical compartments. This discretisation results in an amount of numerical dispersion in the transport calculations, which reflects the amount of physical dispersion expected due to small scale heterogeneities in the B2 unit [53].

Water saturations in the underlying B2 unit, and flows into and within the underlying B2 unit, are calculated by the CFM. Flow and transport are predominantly downwards into the underlying B3 unit.

B3 Unit

The assessment model representation of the groundwater pathway is shown in Figure 5.12. This comprises a single pathway below the trenches, which then splits into six parallel paths from below each vault to the coast. Each of the groundwater paths from the western site boundary to the coast is divided into four sections (Figure 5.12). This enables variations in radionuclide concentrations along the paths to be modelled in response to evolving radionuclide releases from the repository and differences in the attenuation of each radionuclide by decay and sorption. The discretisation also enables coastal erosion of the groundwater paths to be represented.

Each of the 'boxes' in Figure 5.12 is a GoldSim 'pipe' element. The pipe elements calculate 1-D advection and dispersion of a 'packet' of contamination through the length represented by the pipe. A limitation of the pipe elements is that the advective velocity and dispersion of a 'packet' of contamination do not change if conditions evolve while the packet is travelling through the pipe. While this simplification introduces some bias into the transport calculations, we are confident it does not significantly affect the calculation results for the following reasons.

- Each pathway is represented by chains of pipes rather than a single pipe. As a packet of contamination exits one pipe and flows into the next, the transport calculations are updated to reflect the evolving conditions.
- The key radionuclides for the groundwater pathway sorb weakly. Therefore, their travel time through the pipes is short providing limited opportunity for conditions to change while they are travelling through a pipe.
- Strongly sorbing radionuclides are unlikely to reach the natural groundwater discharge location at the coast, except at the very end of the assessment timeframe immediately prior to disruption of the repository by coastal erosion.
- We use the expectation values of radionuclide concentrations in the area of contaminated groundwater to calculate risks associated with the use of water from abstraction wells [68]. This is further discussed below, in Subsection 5.4.7.1. The amounts of activity in the contaminated area have a bigger impact on the expectation values than the spatial distributions, because the expectation values depend on the area weighted average concentrations.

Overall, the potential impacts of the bias are expected to be smaller than the potential impacts of uncertainties.

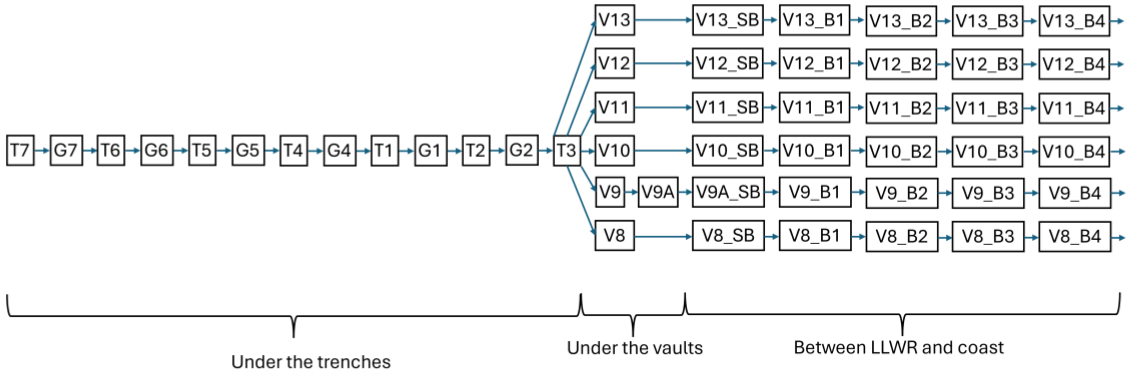

Figure 5.12: Assessment model discretisation of the groundwater pathway (in the B3 unit), each box is a GoldSim model pipe element [23]

The B3 unit and the regional groundwater are not included in the CFM. Data on flows in the B3 unit from below the repository to the coast are taken directly from ConnectFlow calculations (Figure 5.13).

The ConnectFlow model has been calibrated by adjusting the hydraulic conductivities of the lithofacies units until the calculated heads provide a good match against borehole data. The calibration process therefore derives average hydraulic conductivities over the length scales of the units represented in the groundwater flow model. The detailed representation of the geological units in the ConnectFlow model, and the calibration process, ensure that the model provides robust parameter values for input to the assessment model.

The ConnectFlow model has been run for several future time snaps. The effects of climate change, coastal erosion and evolution of the repository engineering on groundwater recharge, the location of the coast, and flows through the repository are included in each time snap.

ConnectFlow outputs pathway flow rates, cross-sectional areas, and vertical thicknesses for use in the assessment model. Pathway areas, vertical thicknesses and flow rates are output at five flow planes (F1 to F5 in Figure 5.13) for each time snap. They are then imported to the assessment model. The assessment model interpolates the flow rates, areas and thicknesses between the time snaps, to provide a description of continuous evolution of the groundwater pathway.

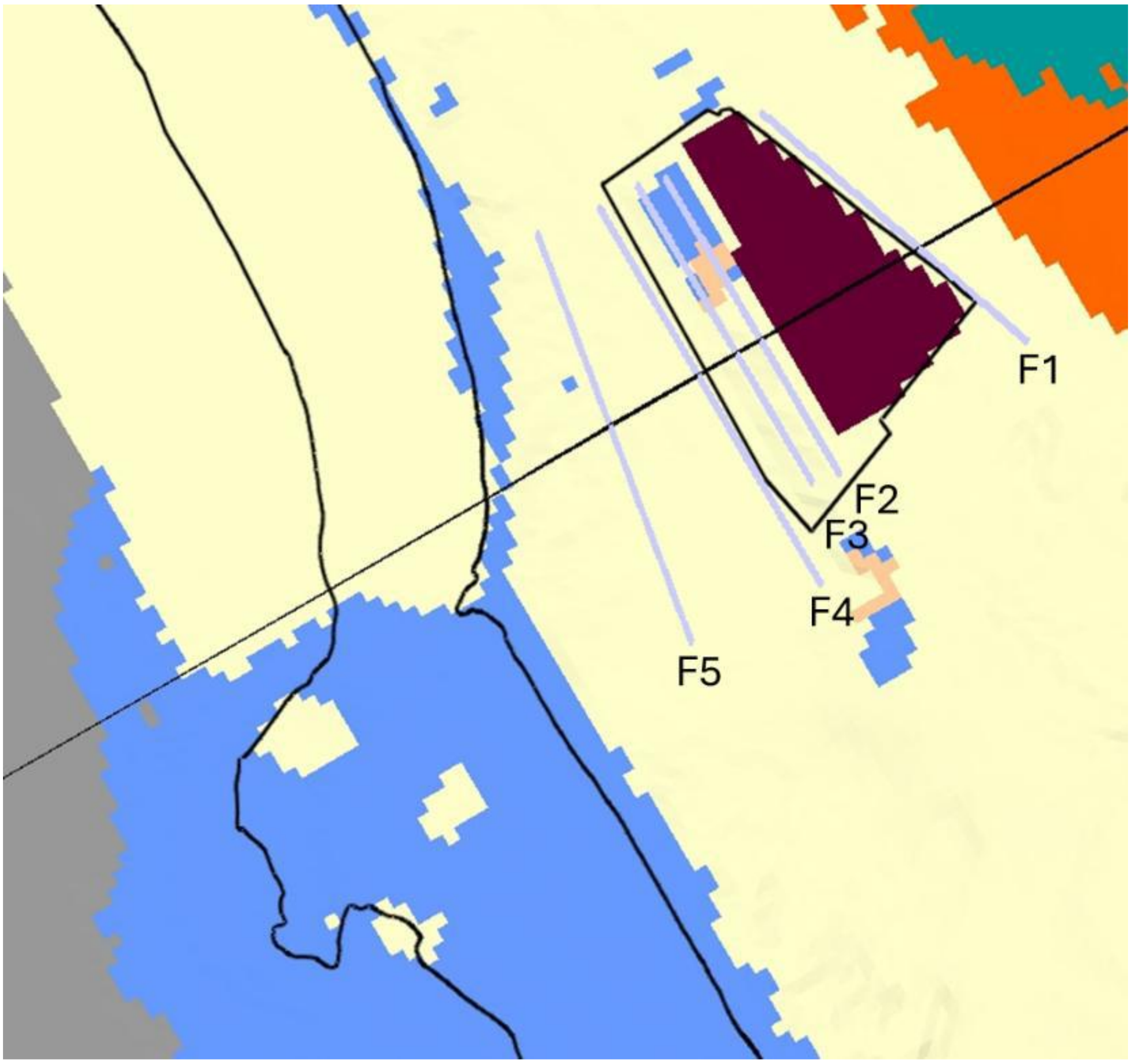

Figure 5.13: Flow planes used to output groundwater pathway flow rates, cross-sectional areas and vertical thicknesses from the groundwater flow model for use in the assessment model [65]

Chemical Processes

Radionuclides (and other contaminants) sorb onto minerals in the B2 and B3 units. Linear, equilibrium, reversible sorption is modelled, consistent with the approach used for the near field.

Average EDTA concentrations are calculated in the B2 and B3 units below the repository and are used to calculate SEFs and SRFs. EDTA concentrations are lower in the geosphere than the repository, due to dilution with groundwater, so the impacts of complexants are lower in the geosphere than the repository. ISA is assumed not to be present in the geosphere, as it is expected to be degraded by microbes.

Concentrations of complexants calculated for the B3 unit below the repository are applied throughout the pathways. Dilution of complexants along the pathways, which could reduce the amount of complexation, is not included in the assessment models.

5.4.6 Assessment Model Representation of System Evolution

5.4.6.1 PoA and Post-PoA

Radioactive decay and ingrowth are calculated in the assessment model for all compartments throughout the assessment timeframe. The model represents evolution of the repository from 1950, through disposal of the first waste in 1959 to Trench 1, to start of disruption by coastal erosion. Therefore, continuous evolution is modelled for the whole of the PoA (including the historic period) and the post-PoA. This is necessary because contaminant transport and behaviour in the post-PoA period will be influenced by processes taking place in the PoA.

Radionuclides are modelled to be disposed to each trench and vault compartment over the relevant operational period. Disposals to each trench and vault compartment are assumed to be constant over the operational period. Radioactive decay and ingrowth are calculated concurrent with disposals and in the period after disposals are complete. Radionuclides are assumed to be homogeneously distributed within each trench and vault. Spatial heterogeneity is further discussed in Subsection 10.7.)

Leaching from the trench waste during the PoA is represented in the assessment model. Infiltration into the trenches changes over time, reflecting the period prior to construction of the interim trench cap, construction of the interim trench cap and construction of the final cap as a series of strips. The CFM calculates the proportions of water that are captured by the trench drains and that drain through the bases of the trenches. These flows are then used to calculate the radionuclide fluxes. Leachate management is assumed to continue for the trenches until the end of the PoA (2242 AD).

A small amount of activity may be released from containers in the vaults during the PoA, for example tritium has been measured in waters ponded in the ullage spaces of containers with open grout ports in the lid. Rainwater runoff from the vaults, which contains any radionuclides released from the containers, is captured by the vault bases and directed into the leachate management system. Monitoring data show that the vault bases are effective at capturing rainwater and directing it into the leachate management system, for example radionuclides have not been detected in the Vault 9 leak detection system.

In our assessment model we assume the containers are effective at limiting rainwater contact with the waste, and the vault bases are effective at capturing leachate from open vaults and directing it into the leachate management system. This is implemented in the assessment model by setting the rainfall rate to zero, i.e. we do not explicitly represent releases from incident rainfall to the leachate management system in the assessment model. This is a cautious approach because is maximises the inventory remaining when the vault is capped.

During the PoA, there is expected to be negligible infiltration through the final cap, and therefore very little radionuclide release from infiltration into containers in closed vaults. The assessment model assumes leachate management continues for the vaults to the end of the PoA (2242 AD), so the infiltration rate for closed vaults is set to zero.

More details of the implementation in the assessment model are provided in reference [23].

5.4.6.2 Degradation of the Near Field

The engineered elements are constructed with characteristics according to their design specification. As time passes the various components will degrade, for example the cap geomembrane may become brittle and fail, and the vault bases and walls may degrade, and their hydraulic conductivity increase. More water will pass through the vault bases to access the drainage blankets, instead of potentially accessing the drainage blankets by overtopping the 1 m high future vault walls.

The degradation of engineered components is represented by the hydraulic conductivities set for the near field components in the CFM. The EPA [13] describes conceptual models of degradation of the engineering and provides time varying cap infiltration rates and engineering component hydraulic conductivities, which are used in the CFM.

5.4.6.3 Environmental Change

Groundwater flows and therefore releases of radionuclides will be influenced by climate change and coastal erosion as discussed in Subsection 5.4.2. The aspects of climate change most important to the groundwater pathway are:

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

The evolution of the site under environmental changes is described in reference [8], and the effects on the groundwater regime in reference [25] [65].

Projections of sea-level rise are described in Subsection 3.5.1. Sea-level rise will increase groundwater heads in the geosphere below the repository. Sea-level rise will increase the rate of coastal erosion, which will shorten the groundwater pathlength to the coast and affect regional groundwater heads in the geosphere below the repository. Coastal erosion will tend to reduce regional groundwater heads below the repository, somewhat offsetting the effects of sea-level rise.

If regional groundwater heads rise to the level of the vault drainage blankets, then the drainage capacity will be greatly reduced. This will cause water levels in the vaults to rise and may ultimately lead to discharges to surface soils and shallow groundwater. The stepped vault bases mean that drainage capacity will be reduced for Vault 12 first, followed by Vault 11, then Vault 10 and finally Vault 9a.

Changes to precipitation and temperature will impact HER, infiltration into the repository, and flows in the geosphere. The boundary conditions for groundwater modelling, for the Reference, High and Low Emissions Scenarios are described in Table 3.4.

The groundwater pathway reference case is based on the Reference Emissions Scenario. The High and Low Emissions Scenarios are explored as variant calculation cases.

A 'what-if' case, the Delayed Coastal Erosion case assumes no change in the natural environment. For the groundwater pathway, this case is represented by retaining the present pathway geometry (including pathlength) and boundary conditions, while representing the effects of near-field degradation using the cap infiltration rates and engineering component hydraulic conductivities developed by the EPA. The scenario is modelled up to 250,000 years after present.

5.4.7 Assessment Biosphere Model

The biosphere model covers the use of abstracted well waters, natural discharges to the marine environment, and discharges to the Drigg Stream. Radionuclides discharged to the marine environment and stream are subsequently transported into the estuary, so there are two pathways to the estuary. Radionuclides discharged to the stream are transported to the marine environment via the estuary, therefore here are also two pathways to the marine environment.

These four biosphere paths or subsystems have several common features. Each subsystem is modelled in GoldSim using a compartment approach in which the subsystems are discretised into well-mixed compartments. The compartments contain water and solid materials (e.g. suspended sediments). The transport of radionuclides between the compartments occurs due to movement of water and solid materials.

The presence of radionuclides in the biosphere leads to potential for radiation exposure. For each biosphere pathway we identify the people at greatest risk (PRPs) and calculate risks to them. The PRPs are based on local and national habits observed present day. People using well water are not currently present between the LLWR site and the coast, but abstraction of well water is observed from other locations along the coast.

The PRPs and exposure modes are described in detail in reference [23] and are summarised in the following subsections. In addition to protection of people, we consider protection of groundwater. Calculated risks for the well pathway are compared against the relevant criteria for protection of groundwater from radionuclides set out in guidance from the Environment Agency [20] [21].

5.4.7.1 Well Biosphere Pathway

The well biosphere pathway represents the abstraction and use of waters from a well that intersects the plume of radionuclides from the LLWR. The potential location, usage, characteristics and probability of a well drilled in the vicinity of the LLWR are discussed in references [83] and [84]. We assume the well provides water for an isolated dwelling constructed between the repository and the coast. We assume the PRP uses water from the well for drinking and to water crops and animals. Key components of the approach to calculating well risks are summarised below and described in more detail in reference [68]. The mathematical models and data are described in more detail in reference [23].

We calculate the expectation value of risk for comparison against the risk guidance level. The expectation value of risk accounts for two key uncertainties:

- whether a well exists within the contaminated area;
- uncertainty in the location of the well within the contaminated area.

If a well exists within the contaminated area, it could be located anywhere in that area. The location of the well could affect the radionuclide concentrations in the abstracted groundwater and therefore the calculated doses and risks. We take this into account when calculating the radiological risks.

To do this, we discretise the plume into small elements (Figure 5.14). Within each element the radionuclide concentrations are effectively constant. For each element, we calculate the risk which would be incurred if a well were to be present in that element. Assuming a well exists somewhere in the plume, the probability of a well existing in a particular element is equal to the area of the element divided by the area of plume. It follows that the expectation value of risk is equal to the area weighted average risk for the plume. This is equal to the risk calculated using the area weighted average radionuclide concentrations, i.e. the expectation values of the radionuclide concentrations.

The contaminated area is defined from our detailed groundwater flow modelling results (Subsection 5.4.2). It excludes areas peripheral to the groundwater flow paths in which radionuclides might be present in low concentrations due to dispersion, i.e. we assume all radionuclides released from the repository, and ingrown within the contaminated area, are concentrated within the main plume. This maximises the expectation values of radionuclide concentrations. However, as discussed later in this subsection, it turns out that radiological risks are not sensitive to definition of the width of the contaminated area perpendicular to the groundwater flow paths.

There are small scale heterogeneities in the properties of the B3 unit within the volumes represented by each of the small elements in our groundwater flow model. These will result in areas with higher and lower radionuclide concentrations, with radionuclides likely concentrated in more permeable sediments in which there is more groundwater flow. It is not possible to characterise these small-scale heterogeneities in full or represent them in the assessment model. However, we have investigated their potential impacts on the expectation values of radionuclides concentrations and concluded their potential impacts are smaller than the potential impacts of other uncertainties we do explore with the assessment model. This is further described in Subsection 10.8.

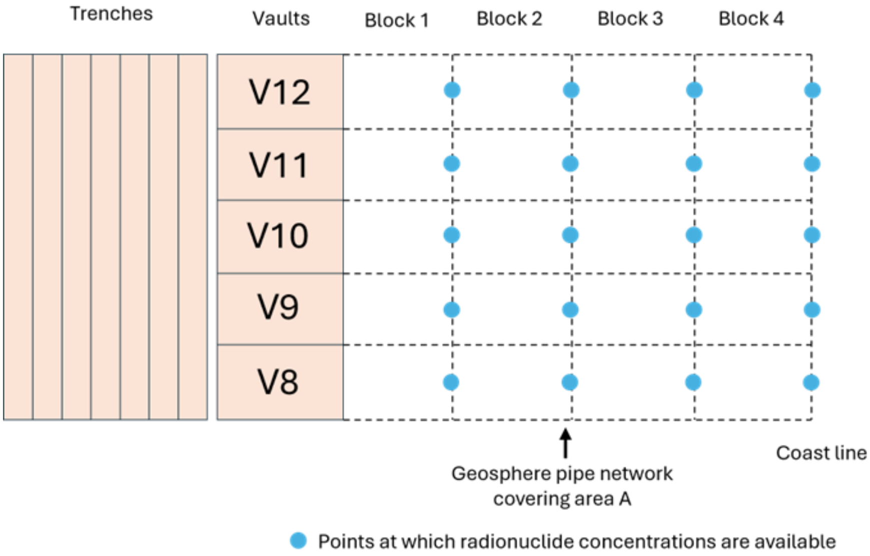

Figure 5.14: Locations where groundwater radionuclide concentrations are output for calculation of the expectation values (i.e. the area weighted concentrations)

It is not certain a well would exist within the contaminated area. We also take this into account when calculating the radiological risk. Therefore, the radiological risk also includes the annualised probability a well exists within the contaminated area. The annualised probability a well exists within the contaminated area is equal to the spatial density of wells (km^{-2}) multiplied by the size of the contaminated area (km^2). This relationship is true because the spatial density of wells is low [65]. The contaminated area decreases over time as the coast erodes, so the annualised probability a well exists within the contaminated area decreases over time.

Best estimate and upper and lower limits of the spatial density of wells were derived considering the present-day spatial density of isolated dwellings along the Cumbrian coastal strip, and local data on licensed and unlicensed boreholes and the present-day use of the Cumbrian coast [84].

As radiological risk depends on the expectation values of radionuclide concentrations in the contaminated area and the annualised probability a well exists within the area, radiological risks are not sensitive to definition of the width of the contaminated area perpendicular to the groundwater flow paths. For given radionuclide fluxes from the repository, and therefore amounts of radionuclides in the contaminated area, as the contaminated area increases the expectation values of the radionuclide concentrations decrease. However, the probability a well exists within the contaminated area increases, and vice versa. These two effects cancel out in the calculation of radiological risk.

5.4.7.2 Marine Biosphere Pathway

The marine biosphere path is concerned with the transport of radionuclides from the LLWR facility in regional groundwater (in lithofacies unit B3) and subsequent discharge at the coast. Cautiously discharges are assumed to be to the inter-tidal region rather than to below

low water (to the local coastal waters), which increases potential doses through occupancy of the coast. Following discharge, the radionuclides will be mixed and dispersed into local coastal waters and sediments through tidal action.

The marine biosphere path PRP is assumed to spend time on the intertidal zone and to consume marine food products, in particular fish, molluscs, crustaceans and seaweed. The inadvertent ingestion of seawater and beach sediments is also considered. It is assumed that marine foodstuffs are taken entirely from local coastal waters and that their radionuclide concentrations are in equilibrium with the concentrations in local coastal waters.

5.4.7.3 Stream Biosphere Pathway

There is a low probability that radionuclides could discharge into the Drigg Stream. The primary contaminated medium is stream water, with stream-bank soils also becoming contaminated. Suspended sediment is not included in the assessment model. For discharges to the Drigg Stream, the people at greatest risk are farming individuals making use of land adjacent to the streams.

Our assessment model for the stream biosphere pathway includes the following very cautious aspects.

- Most of the water entering the stream pathway is unlikely to have contacted the waste (Figure 5.4). However, as discussed previously, in our assessment model, all water that overtops the repository and enters the stream is taken from compartments which represent the waste and immediately adjacent natural sediments. This is expected to overestimate the radionuclide concentrations in the water and fluxes to the stream.
- Some of the water overtopping the cut-off wall may infiltrate the ground adjacent to the cap, transporting of some of the associated radionuclides into the deeper regional groundwater system. However, in our assessment model all the water and associated radionuclides are assumed to discharge to the stream.
- We cautiously assume porewater radionuclide concentrations in the stream bank soils are the same as in the stream water. This is expected to overestimate radionuclide uptake by grazing animals (cows).

5.4.7.4 Estuary Pathway

Dissolved radionuclides that discharge to the marine environment, and the stream, would subsequently enter the tidal section of the River Irt and the Ravenglass Estuary. Consideration of previous modelling studies [85] established that a simple model of the estuary is appropriate. Only the Irt arm of the estuary needs to be represented. The model includes a representation of the sediment column and represents flooding of neighbouring estuary pastures.

The PRP for the Estuary Pathway comprises individuals (e.g. fishermen) that spend a proportion of their time on the estuary pasture and salt marsh regions, and who catch and consume seafood products obtained either from estuary waters or sediments

5.5 Assessment Calculation Results

5.5.1 Assessment Cases

Risks are calculated for a reference case and variant cases. The reference case is based on the Reference Emissions Scenario, reference engineering evolution case described in our EPA [13] and best estimate parameter values. The assessment calculations for this pathway (and all other pathways) use the Stage 2 forward inventory (Subsection 3.4.1). The reference case is evaluated deterministically and probabilistically.

Uncertainties and biases in the assessment model are explored in variant cases. These include:

- uncertainty in the evolution of the system (resulting from factors external to the system), including the High and Low Emissions Scenarios and the 'what-if' delayed coastal erosion case;
- conceptual model uncertainty, including uncertainty in the evolution and performance of the engineered barriers, extent of complexation in the geosphere, and colloidal transport;
- data uncertainty, including flow rates in the B3 unit and the inventory of complexants;
- assessment model bias, comprising representation of the barriers provided by the wasteform.

Additional variant cases are used to explore the risks if only LLW is disposed, and the importance of chemical barriers in the near field (sorption and solubility limitation) and geosphere (sorption).

Risks are calculated for all four biosphere pathways and associated PRPs. Post-PoA releases to the stream biosphere pathway only occur in two of the cases.

- Around 6% of realisations of the probabilistic reference case. These realisations are associated with poor cap performance and higher rates of infiltration into the repository than expected.
- The deterministic early cap degradation and poor cap performance case. This is a low likelihood case with close to lower bound cap performance. The cap degrades faster (i.e. earlier) than in the reference case, and infiltration is always higher (i.e. poorer cap performance) than in the reference case.

Releases to the stream biosphere pathway only occur in extreme situations, so releases to the stream biosphere pathway are not likely to occur. Releases to the estuary via the stream are also unlikely to occur.

Comprehensive results for the groundwater assessment calculations are presented in reference [23]. When reviewing the Phase 2 results, we identified that the solubility limits for uranium in the trenches recommended in reference [86] for use in the groundwater assessment significantly overestimate the solubility of uranium in the trench environment [87]. Therefore, calculations undertaken using those solubility limits result in calculated concentrations of uranium (and ingrown progeny) in groundwater that are too high. As radionuclides disposed in the trenches dominate the calculated risks [23], the risks calculated using these solubility limits are too high.

In response, we revised the uranium solubility limits [87] and re-evaluated the deterministic and probabilistic reference case using the revised limits. The revised results are presented in the final version of [23] and in the following subsections.

All other assessment cases considered in reference [23] (except for a vault-only case, for which trench solubility is irrelevant) have not been re-evaluated. We consider this to be a proportionate approach because for all cases considered, except the 'what-if' delayed coastal erosion case, assessed impacts from uranium and ingrown progeny are below the GRA risk guidance level. The delayed coastal erosion case is not credible as it assumes the coast never erodes, while the coast is currently eroding in a regime of approximately stable sea-level.

The following subsections include the reference case calculation results and a selection of the results from other cases to illustrate the impact of alternative assumptions and uncertainties.

5.5.2 Results Overview

For all emissions scenarios and all bar one calculation case (a low likelihood case with early degradation of the cap and poor cap performance), the peak risk occurs for the well biosphere pathway. The peak expectation value of risk in the probabilistic reference case is also for the well biosphere pathway. In all the cases, including the low likelihood case with early degradation of the cap and poor cap performance, the peak risk is below the risk guidance level.

In the case with early degradation of the cap and poor cap performance, the peak risk is around a factor of three higher for the stream pathway than the well pathway. The peak risk for the stream pathway is $3.5\ 10^{-7}\ y^{-1}$, i.e. below the risk guidance level. The peak risk for the well pathway includes the probability a well exists in the contaminated area, while the peak risk for the stream pathway is a conditional risk, assuming cap performance close to the lower bound. We cannot calculate the probability of early degradation of the cap and poor cap performance to include in the calculation of risks for this case. However, the probability is substantially less than one, so the peak risk for the stream pathway will be substantially less than the conditional risk of $3.5\ 10^{-7}\ y^{-1}$. In addition, our assessment model for the stream pathway is cautious as explained in Subsection 5.4.7.3, which also leads to an overestimation of risk.

5.5.3 Reference Case (Deterministic)

Peak risks for the groundwater pathway are given in Table 5.1. Peak risks are for the well biosphere pathway. Peak risks for the well biosphere pathway are calculated from the expectation values of radionuclide concentrations in the contaminated area and include the annualised probability a well exists in the contaminated area.

The post-PoA risks are very low because the cap performance is very good throughout the assessment timeframe. In this case, the cap geomembrane is assumed to remain intact throughout the assessment timeframe, consistent with the substantial body of evidence described in our EPA [13]. Therefore, once the final cap is installed, infiltration into the repository is very low and there is little leaching of radionuclides, even though the model does not take credit for the containers.

For the well biosphere pathway, risks are dominated by I-129. I-129 is long-lived, sorbs weakly and is not solubility limited. Tc-99 and Cl-36 are the next most significant radionuclides, although the impacts from these radionuclides are several orders of magnitude lower than those from I-129.

Most of the I-129 inventory is associated with potential future disposals of ILW [4]. However, the peak risk is from I-129 released from the trenches before they are fully covered by the final cap. Therefore, the peak risk from I-129 occurs at the start of the post-PoA period, and the risks decrease thereafter. Risks from Tc-99 and Cl-36 are also dominated by releases from the trenches prior to final capping. Peak impacts for these radionuclides occur at the beginning of the PoA (Figure 5.15).

We have sampled for I-129 in leachate from the marine holding tank, probe holes in the interim trench cap, the trench drains, vault leachate and in some boreholes. However, I-129 is not included in the routine monitoring programme. The last sampling for I-129 was in 2015. Many of the samples were below the limit of detection. The highest concentration above the limit of detection was 1.33 Bq l^{-1} in probe hole P7.1 (I-129 was below the limit of detection in some probe holes including the adjacent probe hole P7.2). The highest concentrations measured elsewhere, including in groundwater, were around 0.1 to 0.2 Bq l^{-1}. Many measurements were below the limit of detection, which varies between approximately 0.01 Bq l^{-1} to 0.08 Bq l^{-1}.

The peak expectation value of the I-129 concentration in the contaminated area calculated by the assessment model is 0.004 Bq l^{-1}present day. This is broadly consistent with the measurements considering the dilution expected between the waste and the B3 unit.

Risks due to releases from the vaults are much lower than from the trenches due to effective leachate capture during the PoA and then very low infiltration through the final cap. This is demonstrated by the vault only case presented in reference [23]. Risks from vaults would be even lower if credit was taken for the barrier provided by the containers, and protection of the containers provided by the CPUs and shielded modules. As the peak risks are dominated by the trenches, the peak risks do not change if LLW and ILW are disposed to the vaults in the future, or only LLW.

Short-lived radionuclides, including tritium, decay to negligible levels before the end of the PoA. Longer-lived radionuclides that sorb strongly do not result in significant risk because they are released from the repository very slowly and transported slowly within the groundwater pathway. Radionuclides that sorb strongly may not enter the groundwater pathway between the repository and the coast before the repository starts to be disrupted by coastal erosion, or they might be retained close to the repository.

Table 5.1: Peak risks for the groundwater pathway (deterministic) reference case

Pathway	Peak risk (y^{-1})	Time of peak	Key radionuclides
Well	2.3 10^{-9}	2250	I-129
Marine	8.7 10^{-12}	2250	I-129
Stream	No releases		
Estuary	3.1 10^{-13}	2250	I-129, Tc-99 (and C-14 at later times)

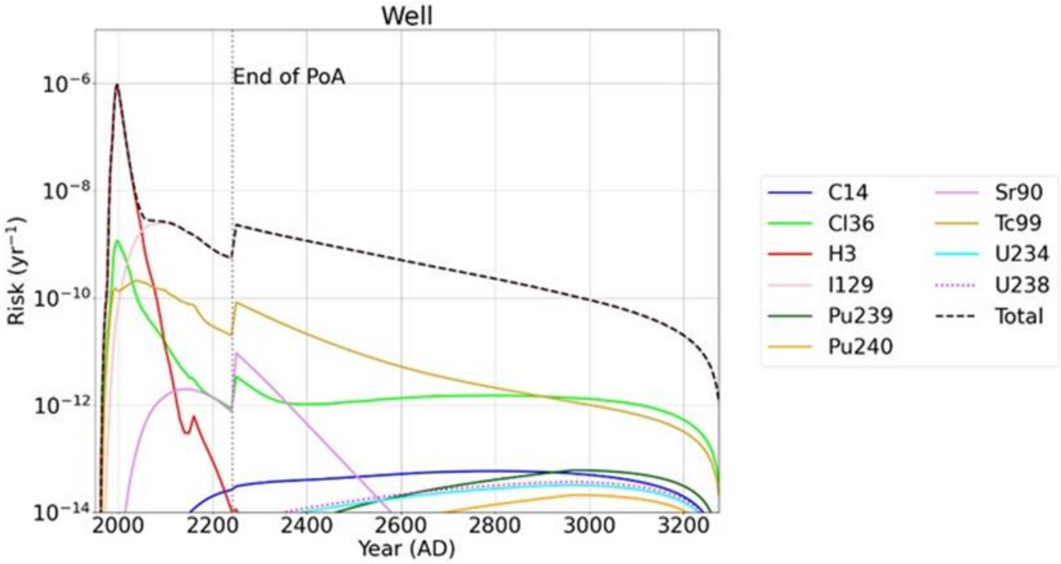

Figure 5.15: Well biosphere pathway risks for the deterministic reference case

Risks from the estuary biosphere pathway are much lower than risks from the marine biosphere pathway because only a small proportion of radionuclides transported to the marine environment are subsequently transported into and retained within the estuary.

Confidence in the calculated risks is built by some simple qualitative arguments:

- The very low infiltration rates into the repository are expected to result in low rates of radionuclide release, consistent with the model results.

- Our near-field work [6] shows that while the cap geomembrane is intact, much of the infiltration into the repository should be consumed through anaerobic corrosion of metal waste (in the trenches and vaults) and metal containers (in the vaults). This process is not included in the assessment model, so the model will tend to overestimate the amount of water available to leach radionuclides. This will be most significant at early times when the cap is performing at its best.
- The assessment model includes cautious assumptions, for example neglecting the barrier provided by the containers.
- The assessment model does not take credit for the expected reduction in future releases from the trenches when the interim cap over the southern area of the trenches is replaced, including releases of I-129, Tc-99 and Cl-36.
- The modelled dilution as the small amount of water draining through the base of the repository mixes with a much larger volume of groundwater flowing through the B3 unit is underpinned by the results of a detailed groundwater flow model. The groundwater flow model provides a good match against measured time average and transient groundwater heads.
- The assessment model has been audited against FEP lists to build confidence all potentially important FEPs are included.
- The assessment model has been developed iteratively, starting prior to the 2011 ESC, with thorough checking at each iteration.

Confidence in the calculation results is also built by comparing the calculated well risks against results from a much simpler (scoping) model. This is also implemented in GoldSim and is described in reference [23].

The scoping model represents release and transport from a trench or vault using seven well-mixed compartments. One compartment represents the trench or vault. Three compartments represent the B2 lithofacies unit underlying the trench or vault, and three additional compartments represent transport in the B3 lithofacies unit. All compartments are assumed to be fully water saturated and solubility limitation is neglected. The model includes a simple representation of repository evolution, and the associated change in water flows.

Lateral flows in the repository are neglected, with only vertical flows due to infiltration through the cap being considered. However, the vertical flow rates through a trench or vault and its underlying B2 lithofacies unit can be different to represent lateral flow through and under the cut-off wall into the B2 unit. These lateral inflows significantly contribute to flow in the B2 unit under the repository because infiltration through the cap is very low. Therefore, flow rates (m y^{-1}) were derived from the CFM in the main assessment model. The flow rates from the CFM were processed to match the simplified evolution included in the scoping model.

The results from the scoping model for each trench and vault are combined to give well pathway risks for the whole repository (Figure 5.16). The scoping model provides a good match to the main assessment model, building confidence in the main assessment model. The slightly higher well pathway risks calculated by the scoping model than the main assessment model are not unexpected given the simplifications included in the scoping model, for example the scoping model assumes all the waste is water saturated and therefore overestimates water contact with the waste.

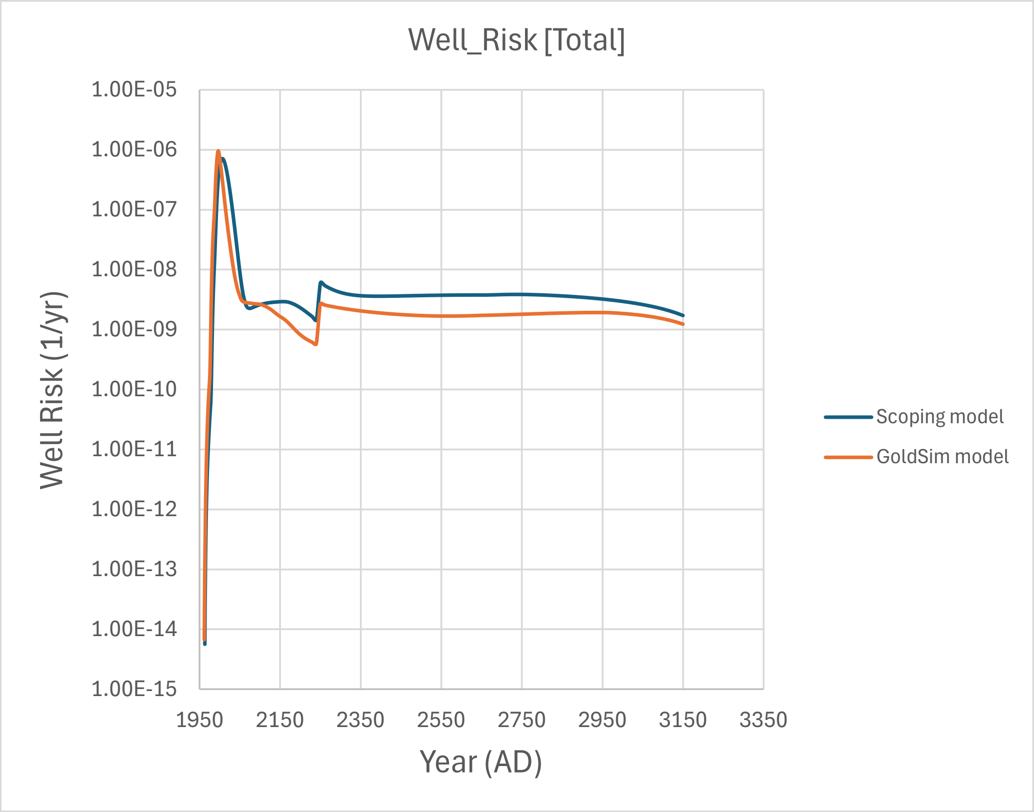

Figure 5.16: Well pathway risks calculated using the scoping model and the assessment model

5.5.4 Reference Case (Probabilistic)

The parameters treated as uncertain in the probabilistic reference case are:

- the radionuclide inventory (in particular, the activity per trench or vault);
- infiltration rates through the cap, and the time of onset of cap degradation;
- parameters relating to the engineered barriers (e.g. hydraulic conductivity);
- solubility limits and sorption parameters for the near field, B2 and B3 lithofacies units;
- the strength of complexation with EDTA and ISA in the trenches and vaults;
- groundwater flows;
- the probability a well exists between the LLWR and the coast;
- the stream sediment loading.

PDFs were developed to describe the uncertainty in each of these parameters. The assessment model was run for 1,000 realisations, independently randomly sampling each of these PDFs for each realisation. We found that the CFM model did not converge for around 5% of the 1,000 iterations. Radionuclide release and transport calculations were not undertaken for these realisations but were undertaken for all the converged realisations of the CFM. We also checked convergence of the statistical results generated by the release and transport calculations.

We have investigated realisations where the CFM did not converge. The CFM represents a complex, coupled, non-linear system, so it is not possible to understand the reason(s) for non-convergence in every realisation that did not converge. However, our investigation indicates there is not a consistent pattern in non-convergence in the CFM that would lead to bias in the assessment results.

Convergence of the release and transport results was checked using two approaches.

1) Running the model for different numbers of realisations to determine the number of realisations needed for the mean well pathway risks to stabilise.
2) Comparing statistics of the of the total pathway risk as a function of time from the first 500 realisations and the second 500 realisations. If the two sets of results are consistent with each other, and with the results from all realisations, then it can be asserted that the results have converged.

Around 350 realisations were found to be sufficient for the mean well pathway risks to stabilise. The statistics from the first 500 realisations and second 500 realisations were found to be similar for all biosphere pathways. There were some differences at the statistical limits, driven by a small number of realisations with low probability behaviours. The low probability, high consequence realisations do have some impact on the mean. However, overall, the results of the convergence studies suggest that, in most instances, 1,000 realisations will provide a reasonable estimate of the mean and median risks for the pathways.

In the probabilistic case, the mean risks are higher than the median risks for all biosphere pathways. This is expected given the shapes of the log-triangular PDFs used to describe uncertainty in some of the parameter values. For example, in the log-triangular PDF for inventory uncertainty, the mean is around a factor of 1.5 higher than the median. The differences between the means and medians of these factors compound in the calculations resulting in an expectation value (mean) of risk that is substantially greater than the median.

The peak median risk for the well biosphere pathway is similar to the deterministic reference case, which is expected. In some of the realisations of the probabilistic case the cap geomembrane fails within the assessment timeframe, resulting in a significant increase in infiltration (but still typically only a small portion of HER). The increase in infiltration leads to higher radionuclide fluxes from the repository and at later times, and an associated increase in well and marine biosphere pathway risks, compared with realisations where the cap geomembrane does not fail. This effect is illustrated in Figure 5.17, by the order of magnitude increase in mean risks from I-129 starting at around 2500 AD.

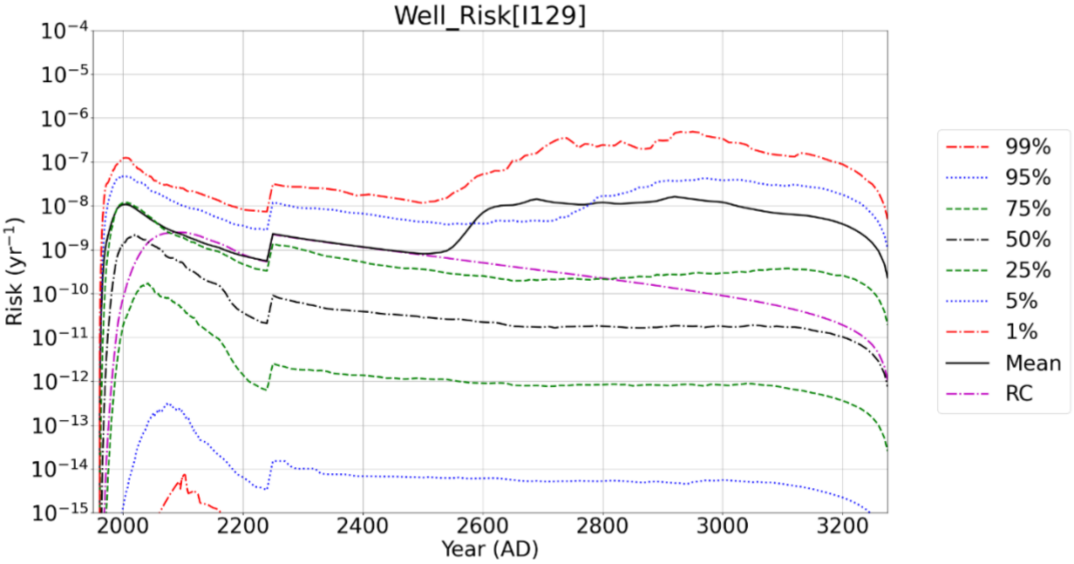

Figure 5.17: Well pathway risks from I-129 in the probabilistic reference case

The mean risks in the probabilistic reference case for the well biosphere pathway are higher than in the deterministic reference case (RC in Figure 5.17) after 2500 AD. Similarly, at the end of the assessment timeframe, the mean risk from I-129 for the marine pathway is around two orders of magnitude larger than in the deterministic reference case.

In the deterministic reference case, I-129 is the key radionuclide for risks. In the mean of the probabilistic reference case, and in the highest percentile cases, Th-232 and its ingrown progeny Ra-228 and Th-228 are more important than I-129. The probabilistic calculation includes parameter uncertainties that equally affect release and transport of I-129, Ra-228, Th-228 and Th-232 (e.g. cap infiltration, groundwater flow rates), and some additional parameter uncertainties that only affect Ra-228, th-228 and Th-232 (e.g. sorption distribution coefficients[11], sorption reduction factors, the amount of EDTA in the vaults). Extreme values of the common parameter uncertainties combined with extreme values for the additional parameter uncertainties for Th-232 and Ra-228 result in Th-232 and Ra-228 becoming more important than I-129 in the highest percentile cases.

Th-232 and Ra-228 are the most important radionuclides in the highest risk realisation. This realisation has a low sorption in the trenches and B2 for Th-232, high sorption of Th-232 in B3 and low sorption of Ra-228 in B3. Consequently, Th-232 readily migrates into the B3 unit where it migrates slowly. Ra-228 ingrown from Th-232 in the B3 unit rapidly migrates through the B3 unit. Th-232 and Th-228 are the most important radionuclides in the second highest risk realisation. This has low sorption of thorium in B2 and B3, so thorium is mobile in the geosphere.

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[11] Iodine only sorbs very weakly onto the B3 unit, so uncertainty in the sorption distribution coefficient has limited impact on its behaviour.

The increase in infiltration following failure of the cap geomembrane is sufficient in a small number of realisations (which have high percentile infiltration rates) to result in water levels in the repository rising to the level where overtopping to the perimeter of the cap occurs. Radionuclides are released to the stream and subsequently enter the estuary. Radionuclides also enter the estuary from the marine environment, as occurs in realisations where water levels in the repository are too low for overtopping to occur.

There is no risk for the stream pathway in the deterministic reference case. The peak mean risk for the stream pathway in this probabilistic case is 1.7 10^{-7} y^{-1} and occurs at 2940, as overtopping only occurs following failure of the cap geomembrane. This risk includes the probability that there is a stream pathway. The assessment model for the stream pathway is simple and includes multiple cautious assumptions, so the risk is expected to be lower than calculated.

The mean risk in the estuary biosphere pathway is around five orders of magnitude larger than in the deterministic reference case by the end of the assessment timeframe. In the deterministic case radionuclides only enter the estuary via groundwater discharges at the coast. In this probabilistic case, radionuclides are also released to the stream in a small number of realisations. These radionuclides subsequently enter the estuary.

Risks from the deterministic and probabilistic reference cases are compared in Table 5.2. The expectation value of risk (i.e. the mean) is below the risk guidance level for all biosphere pathways. This is one line of evidence that groundwater would be protected from the radioactivity in the repository [20] [21]. Other lines of evidence are provided by our other assessments.

Table 5.2: Comparison of risks for the deterministic and probabilistic reference cases

Pathway	Case	Peak risk (y^{-1})	Time of peak	Key radionuclides
Well	Deterministic	2.3 10^{-9}	2250	I-129
	Probabilistic median	1.9 10^{-9}	2250	I-129
	Probabilistic mean	4.2 10^{-7}	2250	Th-232, Ra-228, Th-228
Marine	Deterministic	8.7 10^{-12}	2250	I-129
	Probabilistic median	9.8 10^{-11}	3275	I-129, Ra-226
	Probabilistic mean	3.4 10^{-8}	3275	Th-228, Ra-228
Stream	Deterministic	Historic releases to the stream are included in the model but do not result in post-PoA risks. No releases to the stream post-PoA.		
	Probabilistic median	No releases to the stream pathway.		
	Probabilistic mean	1.7 10^{-7}	2940	Cl-36, Cs-135, Po-210
Estuary	Deterministic	3.1 10^{-13}	2250	I-129, Tc-99 (and C-14 at later times)
	Probabilistic median	4.3 10^{-12}	3275	Po-210, Pb-210, I-129, C-14
	Probabilistic mean	2.4 10^{-8}	2920	Po-210, Pb-210

The results from the probabilistic case start to show the importance of cap performance. Comparing the well pathway risks from the deterministic reference case against the mean (i.e. expectation value) of risks from the probabilistic reference case, shows the potential for failure of the cap geomembrane within the assessment timeframe contributes to higher risks than the deterministic reference case. However, the risks are still below the risk guidance level, and use of log-triangular PDFs, including for the inventory, also contributes to the increase.

In a small number of realisations of the probabilistic model, the cap geomembrane fails early enough, and infiltration is high enough, to result in discharges to the stream. Risks from the stream biosphere pathway are higher than from the marine biosphere pathway. This is expected and is consistent with the aim of the engineering to direct infiltration through the cap to the regional groundwater system, away from surface water and shallow groundwater [10]. However, the expectation value of risk for the stream biosphere pathway is still around a factor of five below the risk guidance level. The impacts of early degradation of the cap and poor cap performance have been explored in more detail in a deterministic case.

5.5.5 Variant case - Early Degradation of the Cap and Poor Cap Performance

This is a pessimistic case. It assumes the 5th percentile time for onset of degradation of the cap geomembrane (i.e. early onset of degradation) and the 95th percentile for the infiltration rate (i.e. high infiltration). As discussed in Subsection5.5.1, this calculation uses a uranium solubility limit for the trenches that is too high. This leads to the identification of U-234 and U-238 as key contributors to assessed impacts in this case. Even with early degradation of the cap geomembrane, a high rate of infiltration, and high uranium solubility, risks are below the risk guidance level for all biosphere pathways.

As described previously for the probabilistic reference case, our assessment model for the stream biosphere pathway is simple and includes multiple cautious assumptions. Therefore, the risk for the stream biosphere pathway is expected to be lower than calculated (Table 5.3).

Table 5.3: Comparison of risks for the deterministic reference case and a case with early failure of the cap geomembrane and poor cap performance

Pathway	Case	Peak risk (y^{-1})	Time of peak	Key radionuclides*
Well	Reference	2.3 10^{-9}	2250	I-129
	Poor cap	1.1 10^{-7}	3120	U-238, U-234, Cl-36, Tc-99, I-129
Marine	Reference	8.7 10^{-12}	2250	I-129
	Poor cap	3.1 10^{-8}	3275	U-238, U-235, U-234, I-129
Stream	Reference	No releases		
	Poor cap	3.5 10^{-7}	2770	Cl-36
Estuary	Reference	3.1 10^{-13}	2250	I-129, Tc-99 (and C-14 at later times)
	Poor cap	9.0 10^{-10}	3275	Pu-239, C-14, Pu-240, Tc-99

* Contributions from U-234 and U-238 are substantially overestimated in the poor cap performance case.

5.5.6 Other Variant Cases

Reference [23] qualitatively and quantitatively assess a wider range of variant cases. Results from the cases assessed quantitatively are summarised in Table 5.4. The most important uncertainty is the performance of the cap. Poor cap performance has a greater impact on calculated risks for the well biosphere pathway than the other uncertainties, although the peak well risk with poor cap performance is still around an order of magnitude below the risk guidance level.

Again, as discussed in Subsection5.5.1, all assessment cases (other than the reference case and the vault-only case) use an inappropriately high uranium solubility limit in the trenches. The results in Table 5.4 (particularly the comparison between the reference case and LLW only case results) indicate that the high uranium solubility limit has only a very minor effect on the calculated risks.

Table 5.4: Summary of variant calculation case results

Category	Case description	Peak risk pathway	Peak post-PoA risk (y^{-1})	Comments
Reference case (deterministic)	Best estimate conceptual model and parameter values.	Well	2.3 10^{-9}	Well risks are dominated by I-129 released from the trenches during the PoA. Peak risks are at the start of the post-PoA period.
	Only LLW disposed to the vaults is included in the assessment model.	Well	4.1 10^{-11}	Well risks from vault LLW are considerably lower than from the trenches.
	Only LLW is included in the assessment model (trenches and vaults).	Well	2.3 10^{-9}	Well risks are dominated by releases from the trenches.
Conceptual model uncertainty	Model includes diversion of water down the gaps between vault waste stacks. Radionuclides diffuse out of the containers into water flowing down the gaps. Ignore the presence of the containers as a barrier to diffusion - radionuclides can diffuse through the full area of the waste stacks.	Well	3.8 10^{-8}	Well risks are dominated by Cl-36, followed by I-129 and Tc-99. Risks are overestimated as the simple model implementation results in radionuclide concentrations in the gaps between waste stacks exceeding those in the waste.
	Assume there is no complexation in the geosphere.	Well	2.7 10^{-9}	Reference case EDTA concentrations in the geosphere are too low for complexation. Therefore, results identical to the reference case.

Category	Case description	Peak risk pathway	Peak post-PoA risk (y^{-1})	Comments
Evolution uncertainty	As the reference case, but with coastal erosion assumed to occur according to the High Emissions Scenario.	Well	2.3 10^{-9}	Peak risk from I-129 at 2250. Negligible change to the groundwater pathway compared with reference case during the PoA and start of the post-PoA period.
	As the reference case, but with coastal erosion assumed to occur according to the Low Emissions Scenario.	Well	2.6 10^{-9}	Peak risk from I-129 at 2250. Negligible change to the groundwater pathway compared with reference case during the PoA and start of the post-PoA period.
Engineering evolution and performance uncertainty	Early failure of the cap geomembrane and poor cap performance.	Stream	3.5 10^{-7}	Water levels in the repository rise to the point where overtopping occurs and water is released to streams. Peak risk from Cl-36 at 2770. Peak risk from the well pathway is 1.1 10^{-7} y^{-1}.
	Faster degradation of the vault bases and walls (95th percentile values of hydraulic conductivity).	Well	2.7 10^{-9}	Post-PoA well risks are dominated by I-129, U-238 and U-234 released from the trenches during the PoA. No significant impact on releases from the vaults.

Category	Case description	Peak risk pathway	Peak post-PoA risk (y^{-1})	Comments
Parameter uncertainties	Complexants inventory enhanced by an order of magnitude.	Well	2.8 10^{-9}	I-129 does not form complexes with ISA or EDTA. There is no increase in complexation of uranium in the trenches. Increased complexation of other radionuclides only has a small impact on total risk compared with the reference case.
	Groundwater flow rates in B3 increased by three standard deviations.	Well	1.9 10^{-10}	Greater dilution than the reference case (this is more important than faster transport of radionuclides that sorb).
	Groundwater flow rates in B3 decreased by three standard deviations.	Well	3.9 10^{-8}	Lower dilution than the reference case. Notably slower transport of radionuclides at sorb, such as U-234 and U-238.
Barrier performance and other sensitivity cases	Zero sorption and unlimited solubility in the trenches and vaults. No change to sorption in the geosphere.	Well	5.1 10^{-9}	This is not a realistic case but is undertaken to understand the contribution of the barrier to overall performance. Peak risks are due to U-234 and U-238 as they are no longer solubility limited.

Category	Case description	Peak risk pathway	Peak post-PoA risk (y^{-1})	Comments
	No sorption in the geosphere. No change to sorption in the near field.	Well	2.0 10^{-6}	This is not a realistic case but is undertaken to understand the contribution of the barrier to overall performance. Peak risk is due to Ra-228 ingrown from Th-232 disposed to the trenches.

5.6 Summary of Groundwater Results and Assessment

We use detailed models of the near field, groundwater flow, and the outputs of our EPA to identify the most important FEPs and implement appropriately simplified representations in our assessment models. We explore uncertainties, biases, and the roles of individual barriers using our assessment models to understand the key controls on risks and build confidence in the assessment results. Exploration of biases includes alternative model representations of the system, involving different assumptions. For example, in the reference case we cautiously take no credit for the containers and assume all infiltration through the cap flows through the waste (the uniform flow model). We have developed a more realistic, but more complex, dual porosity model which takes credit for flow of water down the gaps between the container stacks, instead of through the waste, but does not take credit for the containers as a barrier to radionuclide release.

Once we have identified the key FEPs from the assessment model results, we implement these in simpler insight models to help confirm the main assessment model results and build confidence.

For all biosphere pathways, the best estimate (deterministic reference case) risks to people are orders of magnitude below the risk guidance level. Best estimate (deterministic reference case) risks are highest for the well pathway. At $2.3\ 10^{-9}\ y^{-1}$, the peak risk from the deterministic reference case is around three orders of magnitude below the risk guidance level. This result also provides one line of evidence that groundwater would be protected from radionuclides in the waste [20] [21]. Other lines of evidence are provided by our other assessments.

The expectation values of risks (i.e. the means) from the probabilistic reference case are also below the risk guidance level. The highest risk is for the well biosphere pathway at $4.2\ 10^{-7}\ y^{-1}$. The expectation value of risk for the well biosphere pathway is dominated by a relatively small number of realisations and is very similar to the 95th percentile.

For all biosphere pathways the peak expectation value of risk is much higher than the peak median risk. The peak median risks are more similar to the peak risks from the deterministic reference case. This is expected given the shapes of the log-triangular PDFs used to describe uncertainty in some of the parameter values. For example, in the log triangular PDF for inventory uncertainty the mean is around a factor of 1.5 higher than the median. The differences between the means and medians of these factors compound in the calculations resulting in an expectation value (mean) of risk that is substantially greater than the median.

The probabilistic reference case includes the potential for failure of the cap geomembrane, leading to higher infiltration into the repository and a higher expectation value of risk than the deterministic reference case where the cap geomembrane does not fail. Although this aspect of the probabilistic reference case is more realistic than the deterministic reference case, we consider that overall the mean risks from the probabilistic reference case are cautious.

The deterministic reference case results show that risks for the well pathway are dominated by releases of I-129 from the trenches before they are covered by the final cap. The calculated releases from the trenches for the time period c.2030 to the end of the PoA are arguably cautious as they do not include replacement of the interim trench cap over the southern area of the trenches. As the peak post-PoA risks are due to releases from the trenches during the PoA, the peak post-PoA risks may be overestimated.

Peak post-PoA risks are negligibly reduced if ILW is not disposed. This is because peak post-PoA risks are due to releases from the trenches during the PoA.

The key barrier is the cap, and the most important uncertainty is its performance. The probabilistic case includes uncertainty in the cap performance, and we have evaluated a low likelihood deterministic case with early failure of the cap geomembrane and poor cap performance. Poor cap performance leads to releases to the stream pathway. This only occurs in a small percentage of realisations of the probabilistic reference case, in which the cap performs poorly, similar to the deterministic early cap failure and poor cap performance case.

The expectation value of risk for the stream pathway in the probabilistic reference case is 1.7 10^{-7} y^{-1}. This is the mean over realisations in which there is a stream pathway and realisations in which there is not a stream pathway, so it includes the probability there is a stream pathway. A pathway to the streams develops in 6% of the realisations.

In the deterministic early failure of the cap geomembrane and poor cap performance case, the peak stream pathway risk is 3.5 10^{-7} y^{-1}. This is a conditional risk because it assumes early failure of the cap geomembrane and poor cap performance.

In the unlikely case of poor cap performance, the calculated risks are below the risk guidance level, despite the cautious assumptions in our models. For example, our models do not take credit for the barrier provided by the containers in the vaults, nor protection of the containers by CPUs and shielded modules. Although the long-term performance of future stronger containers for LLW, new containers for ILW and shielded modules is uncertain, this is a notable cautious bias in our assessment models. Our assessment model for the stream pathway also includes several cautious assumptions (Subsection 5.4.7.3).

The results of the poor cap performance case and probabilistic reference case confirm the basis for the optimised post-PoA design: to minimise water contact with the waste, then passively direct any waters that contact the waste to the regional groundwater system. Transport through the regional groundwater system provides the following safety functions:

- radionuclides are isolated from the biosphere for as long as possible;
- strongly sorbing radionuclides are contained in the geosphere;
- dilution and dispersion of all radionuclides.

6 Assessment of the Gas Pathway

This section presents the background, approach to assessment, models and data, case selection, and assessment calculation results for the gas pathway. A complete description of these elements is provided in reference [26]. The final subsection provides an assessment of the results against the regulatory guidance.

6.1 Assessment Scope

The scope of the gas assessment is to assess the potential risks associated with post-PoA releases of radioactive gases from the LLWR, consistent with Requirement R6 of the GRA [19]. Radioactive gases could be released through the cap or through a gas vent. The most significant impacts would be to receptors on the cap, i.e. residents on the cap or those using the cap for agriculture, who could be present after the end of the PoA. Releases of gas during the PoA are considered insofar as they affect post-PoA releases (impacts during the PoA are considered separately in the PoA assessment [14]). The period of interest lasts until disruption of the repository by coastal erosion - risks from radioactive gases during erosion of the repository are discussed qualitatively (see Section 11).

C-14 and Rn-222 are identified as the only two radionuclides with the potential to give rise to radiological exposure resulting from gaseous releases from the LLWR in the post-PoA period. Other gaseous radionuclides can be excluded for the following reasons:

- Tritium (H-3) has a half-life of 12.3 years and will have decayed to low levels by the end of the PoA.
- Thoron (Rn-220) has a very short half-life (56 seconds) and will decay to negligible levels before it is released from the repository.
- Chlorine and iodine are very soluble in water, so volatile radioactive gases such as Cl-36 and I-129 are only likely to be generated in very small quantities, and the cap will help to retain them in the repository.

This is consistent with the approach undertaken in the 2011 ESC [88].

6.2 Previous Assessments and Environment Agency Comments

We developed separate assessments of the potential impacts from radon gas and C-14-bearing gases[12] in support of the 2011 ESC [89] [90]. These assessments only considered the post-PoA period. The potential impacts from radon and C-14 gas during the PoA were assessed separately, using simpler, cautious models [91]. The human intrusion assessment [88] also considered the potential post-PoA doses from radon associated with inadvertently

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[12] C-14 bearing gases considered in the gas pathway assessment are principally C-14 bearing methane (in the near field) and C-14 bearing CO_2 (in the biosphere). C-14 bearing gas is referred to in shorthand as C-14 gas for the remainder of the report.

intruding into the waste, e.g. excavation of waste and then construction of a house on ground contaminated by the waste.

An updated, less cautious and more realistic assessment of the potential post-PoA impacts of C-14 was developed in 2013 [40]. This formed the basis for updated C-14 capacity calculations, and the current site radiological capacity for C-14. This included a revised model for C-14 uptake in the biosphere, as published in [92].

The Environment Agency's review of the 2011 ESC identified FIs that needed to be addressed and made recommendations for potential improvements. We have taken these recommendations into account when developing the gas pathway assessment. A full description of how they have been considered is provided in the *'Addressing Regulatory Requirements and Guidance'* report [18].

6.3 Approach in this Assessment

The assessment of radioactive gas in support of the 2026 ESC considers the potential impacts from C-14-bearing gas and radon. Doses from radon are dominated by inhalation, while doses from C-14 are dominated by foodchain uptake through plants. People can be exposed to radon, C-14 gas or both gases, depending on the situation. We have assessed the potential risks from situations where people are exposed to either radon or C-14 gas, and situations where people are exposed to both gases.

We have not yet decided whether the cap vent [5] will be closed or left open at the end of the PoA. Also, the gas management strategy and vent design have not been optimised. Therefore, cases with the vent closed and the vent left open are assessed for both C-14 and radon. The results will inform the decision about whether the vent will be closed at the end of the PoA and will inform optimisation of the gas management strategy and vent design, including options [93] with more than one vent.

Carbon-14

The C-14 gas assessment approach builds on that developed for the 2013 assessment [40]. However, there have been substantial updates and improvements to the C-14 gas conceptual and assessment models. These reflect updates to the illustrative inventory, repository design and understanding of the repository evolution and performance, and updates to understanding of releases of C-14 from key waste types.

Key developments to the conceptual model for C-14 gas compared with the 2013 update are described below.

- Recognition that updated understanding of the performance of the cap geomembrane will result in lower long-term infiltration into the near field and less leaching of the waste and grout. This will result in higher pH conditions in the containers throughout the assessment timeframe. The pH in the containers will typically be too high for significant microbial activity, including microbial processes that generate C-14 gas.

- The higher pH conditions in the containers will lead to much lower metal corrosion rates, and slower (abiotic) release of C-14 gas from irradiated metals congruent with corrosion.
- Once the geomembrane in the cap fails, release of C-14-bearing species in solution from existing containers, and committed containers of the same design, may be different to releases from planned new stronger containers for future disposals.
- Updates to the availability, rates and forms of C-14 release from key waste types including graphite and irradiated steels based on the results of experiments, national and international studies (e.g. [26]).
- Updates to the range of waste types included in the conceptual model, associated with the potential to dispose ILW.
- Updated understanding of the form and rate of release of C-14 from secondary waste slags produced during metal treatment.

Generation and release of C-14 gas is a continuous process, sometimes starting before the waste is packaged or disposed (e.g. releases from graphite), continuing throughout the PoA and post-PoA. The amount of C-14 released during the PoA affects the amount remaining and released post-PoA. We have therefore updated our assessment modelling approach to calculate releases for the whole lifetime of the repository, starting in 1959 and continuing until the repository starts to be disrupted by coastal erosion. C-14 gas fluxes are passed to the PoA assessment [14], to calculate potential doses from C-14 gas during the PoA. Post-PoA risks from C-14 gas are assessed here.

Radon

Radon has a short half-life of about 3.8 days. The transport time of radon gas from the waste to the cap surface (by advection and diffusion) is therefore an important factor in the assessment, since this affects the proportion of radon that will have decayed before being released.

Changes to the anticipated long-term engineering (cap) performance have necessitated substantial revision of the radon conceptual and assessment models compared with the 2011 ESC.

It is now recognised that the geomembrane in the cap should remain intact for hundreds of years to several thousand years [13]. While the cap geomembrane is intact it will be a barrier to diffusive radon gas release but also bulk gas release. With the cap vent(s) closed, a small gas overpressure could develop in the repository, with bulk gas focused through defects (small holes) in the cap geomembrane to discharge at the cap surface, increasing bulk gas velocities compared with the situation where there is no geomembrane. Advection of radon by bulk gas is therefore expected to be an important transport process. After the cap geomembrane has degraded, bulk gas velocities will substantially decrease, so diffusion could become an important or dominant transport process.

In the 2011 ESC, transport of radon was considered to be diffusion dominated because the cap geomembrane was expected to be less durable, and degrade more quickly, than is now expected.

6.4 Assessment Models and Data

6.4.1 Conceptual Models

6.4.1.1 C-14 Gas

In the trenches, C-14 is assumed to be present as surface contamination on a wide range of waste types. Microbes will degrade organic waste generating CO_2 and CH_4 gases including in C-14-bearing CO_2 and CH_4. H_2 will be generated by anaerobic corrosion of metals. Microbes will reduce CO_2 using H_2 forming bulk and C-14-bearing CH_4.

C-14 present as matrix contamination in steels will be released congruent with corrosion of the steel. As the steels in the trenches are generally thin, and corrosion rates are moderate under the near neutral trench pH conditions, most of any matrix contamination is likely to be released over the assessment timeframe. C-14 may be released in methane gas and in inorganic and organic forms to waters. Microbes can metabolise C-14 in inorganic and organic forms to methane.

Most of the C-14 inventory in the vaults is associated with matrix contamination (e.g. graphite and steels), but there is also surface contamination of C-14. Biogeochemical conditions in the vaults are significantly different to the trenches. At closure, the pH in the waste containers is expected to be around pH 12.5 [6]. This pH is typically too high for microbial activity, although some activity in niches on the surfaces of cellulosic waste in super-compacted pucks cannot be ruled out with confidence (e.g. by analogy with reference [94]).

C-14 is released from the vault wastes at different rates and in different forms, depending on the waste type. C-14 is released to gas or water in inorganic and organic forms. C-14 released in inorganic forms will react with cement minerals in the grout and become trapped in carbonate minerals in the containers. While the cap geomembrane is intact, infiltration into the repository is expected to be very low [13]. There will be no flows of water through the containers and waste, or down the sides of the container stacks, to establish transport pathways for radionuclides in solution.

Once the cap starts to degrade, infiltration into the repository starts to increase. Water is expected to flow through the stacks of existing and committed containers (Figure 5.2), transporting C-14 in solution out of containers. Future stronger containers will mostly be intact and C-14 in solution can diffuse out of those containers where the containers are locally perforated (Figure 5.3). Outside the containers the pH is expected to be in the range pH 8 to pH 10. A cautious approach is taken to microbial activity outside containers, and all organic C-14 (in the form of small organic molecules) is assumed to be metabolised to CH_4.

The travel time of C-14 through the repository and cap is expected to be less than one year, so in the context of the assessment, C-14 gas is released as soon as it is generated with negligible attenuation by radioactive decay. The reference case assumption is that the cap vent(s) will be closed at the end of the PoA. Post-PoA, C-14 gas will migrate through the cap, finding routes through defects (small holes) in the geomembrane. A variant case assumes the cap vent(s) will be left open post-PoA.

C-14 will be taken up by plants grown in topsoil directly above the cap, by photosynthesis and transpiration. Exposure to C-14 gas can occur by inhalation and consumption of contaminated foodstuffs. The latter is the dominant exposure pathway. Doses are calculated to a smallholder who is assumed to live on the cap, consume foodstuffs from a kitchen garden and keep goats for milk production. Assuming the receptor is a smallholder is cautious because it maximises carbon intake from the land and therefore doses from C-14.

6.4.1.2 Radon

Radon gas is generated by radioactive decay of radium (Ra-226). However, only a fraction of the radon generated is released to the gas phase. Some remains trapped in mineral grains where it decays to radionuclides that are not gases.

Radon gas will diffuse through the waste, profile and cap material. It will also be advected by bulk gas. The travel time through the waste, profile and cap is likely to be substantially longer than the half-life of radon, resulting in significant attenuation.

The relative importance of advection and dispersion for radon will depend on the condition of the geomembrane in the cap, which will be a barrier to gas movement. Gas will only be able to pass through defects (small holes) in the geomembrane. This has two effects compared with the situation where there is no geomembrane. The travel time will tend to increase because the gas pathlength is longer, and the area for diffusive release is very small. Conversely, retarding release of bulk gas will cause a small degree of pressure rise in the repository, until bulk gas is able to flow sufficiently quickly through the defects that the bulk gas release rate matches the generation rate. Advection of radon through defects by bulk gas will significantly reduce the travel time compared with diffusive transport.

Once antioxidants in the geomembrane are depleted, and oxidative induction has occurred, defect prevalence will increase and stress tears will develop, until the geomembrane completely fails around 150 years later [13]. At some point during this 150-year period there will be enough tears that the geomembrane will cease to be a barrier to gas release. The spatial pattern of release and travel time will be significantly different with the geomembrane intact and degraded, so both situations are considered.

The conceptual model of gas migration with the geomembrane intact is shown in Figure 6.1. Most of the radon flux will tend to be sourced from close to the defect, as gas velocity increases towards the defect and the pathlength to the defect decreases, reducing the travel time to the defect. As distance from the defect increases, the proportion of radon that reaches the defect before it decays will decrease.

Gas is expected to rise vertically through the profile material and then migrate laterally within the gas collection layer (Figure 6.1). The BES has high water saturation and low permeability. Gas is expected to migrate through an area of the BES below the defect in the geomembrane, and along the interface between the top of the BES and the geomembrane to the defect. This is the reverse of the flow path considered for water [95]. There will be some lateral spreading of gas in the cap layers above the defect in the geomembrane. However, once equilibrium gas pressures have developed in these cap layers the amount of lateral spreading is expected to be limited.

Conditions in the repository will be dry. As gas pressure builds up the repository, some bulk and radon gas may migrate under or through the cut off wall [73]. Our reference assumption for assessment calculations is that all bulk gas, and associated radon, migrates to the surface of the cap. This is expected to be an overestimate of the bulk and radon gas fluxes to the surface of the cap. This assumption is cautious in the context of the radon pathway because it maximises bulk gas velocities through the cap, minimising the time for decay of radon in the cap. It also reduces the pathlength and travel time for most waste.

If the vent(s) is(are) left open after the end of the PoA, gas pressures in the repository will be lower and most of the bulk gas is expected to discharge through the cap vent(s). The amount of bulk gas that discharges through small holes in the geomembrane is expected to be much lower. Only radon within a 'capture zone' of each vent would contribute to the radon flux out of the vent. Any radon outside of the capture zone would decay to negligible levels before reaching the vent, due to the travel time.

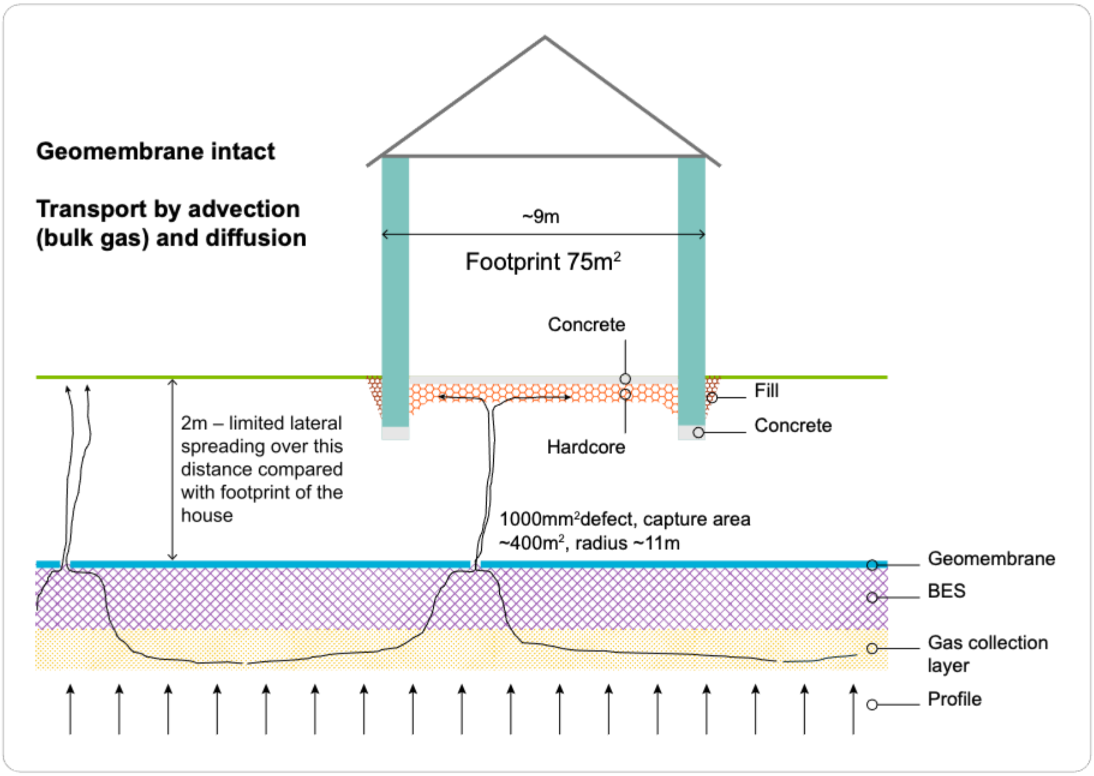

Figure 6.1: Conceptual model of gas migration with the geomembrane intact.

Once the geomembrane has fully degraded, bulk and radon gas are expected to migrate vertically through the cap by diffusion and advection (Figure 6.2). Diffusion will be a much more significant transport process because gas can flow through the full area of the cap, so the advective velocity will be much lower.

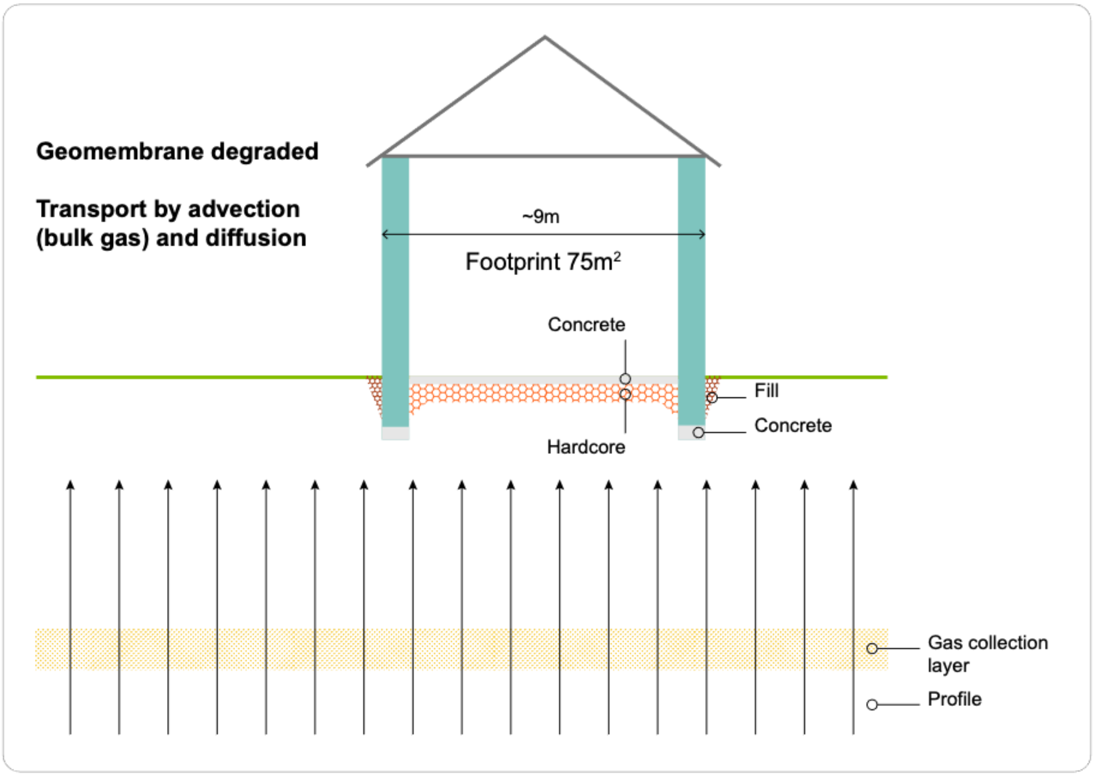

Figure 6.2: Conceptual model of gas migration with degraded geomembrane.

Exposure to radon gas is through inhalation. This can occur inside buildings and outside. If radon gas is present beneath a building, it can enter the building through foundations, floors and cavity walls. Cautiously, all bulk gas and associated radon immediately beneath a building is assumed to enter the building. There will be ventilation of radon to the air outside the building.

6.4.2 Overview of the Assessment Models

The important features, events and processes differ for C-14-bearing gas and radon gas. Therefore, we have developed separate assessment models for each gas. Features, events and processes common to both models are treated consistently, and data common to both models are also consistent. Therefore, risks to people that could be exposed to radon and C-14-bearing gas can be calculated by adding the risks from each model to give the total risk for the gas pathway. An overview of each model is provided below.

6.4.2.1 C-14 Gas

The assessment model for C-14-bearing gas is implemented in GoldSim (Subsection 2.10). The model calculates C-14-bearing gas generation and release from the trenches and each individual vault, and potential doses to a smallholder on the cap from plant uptake of C-14 and ingestion of contaminated plants and animal products. Releases during the PoA affect the amount of C-14 remaining at the end of the PoA, and C-14 gas fluxes post-PoA. Therefore, the model represents the whole lifetime of the repository, from first disposals in 1959 until the repository starts to be disrupted by coastal erosion. Calculated C-14 gas fluxes during the PoA are passed to the PoA assessment [14] for calculation of potential doses during the PoA.

Only a small proportion of the total C-14 inventory is in the trenches, so the trenches are treated as a single source area. The C-14 gas fluxes from each vault will be different, reflecting differences in the total C-14 inventory in the vault, the types of waste C-14 is present in, and differences in the biogeochemical conditions in containers of different waste types and between the vaults. The area of a smallholding is comparable to the area of a vault, and spatial variations in the inventory in Vault 9 and the future vaults are unknown. Therefore, it is not necessary to calculate fluxes from sub-areas of the vaults. As risk depends on the area weighted average flux over the whole cap, uncertainty in the spatial distribution of the inventory in Vault 9 and the future vaults will only have a limited effect on the calculated risk.

The only exception is that we calculate C-14 gas fluxes from the shielded modules, as there is potential for C-14 gas release from the modules to be higher than from other areas of the vaults due to concentration of the ILW inventory in these modules. We calculate doses assuming the whole area of a smallholding is underlain by shielded modules. This is a cautious assumption for the Reference Inventory. The Reference Inventory is not expected to contain sufficient volume of waste requiring disposal in shielded modules for the whole area of a smallholding is underlain by shielded modules.

C-14 gas release from the trenches is calculated using the Simple Model Of Gas Generation (SMOGG)[13] and the gas fluxes are input to the GoldSim model. C-14 gas release from the vaults and shielded modules is calculated in the GoldSim model.

For each vault and shielded module, the waste is divided into several types (Figure 6.3, Figure 6.4). These reflect differences in the form of C-14 in the waste, the release processes and rates. Each waste type is represented by a model compartment. Further model compartments represent the containers[14] and vault outside the containers. Transfers are used to represent release of C-14 gas and C-14 in solution from each waste type, and the subsequent transport and fate of C-14 in solution in the repository. The latter includes incorporation of C-14 in solution into gas by microbial activity. Ultimately C-14 is either released in gas, trapped in carbonate minerals, or released in water (as an authorised discharge during the PoA, or to groundwater via the vault bases and passive drains post-PoA).

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[13] This is a spreadsheet model, originally developed to calculate bulk and radioactive gas generation from waste that might be disposed to a deep Geological Disposal Facility (GDF) [141]. It has also been applied to other assessments, such as those for individual waste streams.

[14] In the model the containers are represented by two compartments: container and container porewater. For simplicity this is not shown in the figures. The container porewater compartment is used to represent dissolved C-14 that is retained in porewater inside the container until infiltration into the repository increases to a level where aqueous transport out of the containers is possible.

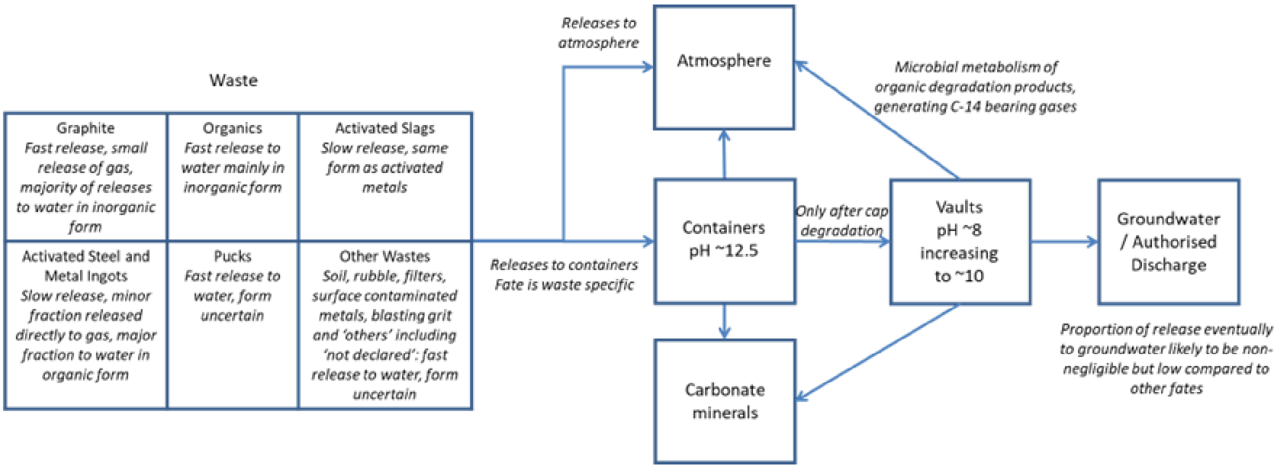

Figure 6.3: **Model compartments and transfers used to represent release and fate of C-14 in an LLW vault**

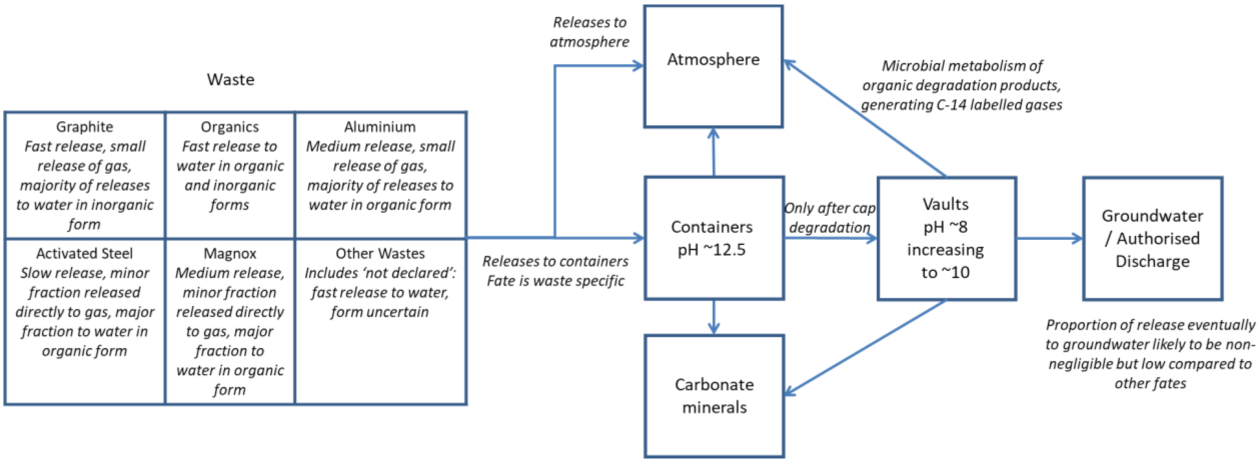

Figure 6.4: **Model compartments and transfers used to represent release and fate of C-14 in a ILW vault or shielded module**

C-14 can be released from the waste in the form of gas (carbon dioxide, methane, or carbon monoxide), and to solution in inorganic (carbonate and bicarbonate ions) and organic forms (small organic molecules). The proportions depend on the waste type and the biogeochemical conditions in the vault containers.

The assessment model also calculates diffusive and advective transport of aqueous C-14 from the containers into the vaults. Release and transport depend on the rate of infiltration into the repository, and the container type and condition. The model includes time varying infiltration, representing degradation of the cap and increase in infiltration with time, and separate models for releases from each waste type in each container type.

In the containers and vaults, C-14-bearing carbonate and bicarbonate ions will mainly react with cement minerals (e.g. in grout) forming carbonate minerals. In the vaults, C-14-bearing small organic molecules may be metabolised by microbes forming C-14 gas.

The C-14-bearing gas is released to atmosphere through defects (small holes) in the cap geomembrane or through a gas vent. The half-life of C-14 is 5700 years, so there is negligible decay of C-14 while it migrates through the cap or vent. C-14-bearing methane and carbon monoxide released from the waste are expected to be oxidised to $^{14}CO_2$ in the cap soil.

The biosphere model represents uptake of C-14 in plants grown in soil on the cap and ingestion of plants by animals and humans. It is a specific activity equilibrium model based on reference [96], and the calculations are undertaken in GoldSim. Compartments and transfers are not needed to model transport of C-14 in the biosphere or to calculate concentrations in the environment or in foodstuffs. Two routes are modelled for uptake of C-14 into plants: photosynthesis and transpiration. For uptake via photosynthesis, the specific activity of C-14 in the biomass of the plant is equated to the specific activity within the plant canopy atmosphere. For transpiration, the concentration in plants is calculated as the concentration of C-14 in the soil water multiplied by a transpiration ratio. The specific activity of C-14 in animals is taken to be equal to the specific activity of C-14 in the plants they consume.

6.4.2.2 Radon

The assessment model for radon is also implemented in GoldSim. The model represents generation of radon from the decay of Ra-226 in the trenches and vaults, transport of radon through the cap, and potential doses to a house occupant on the cap from inhalation of radon gas. The main steps in the calculations are:

1) Calculate decay of Ra-226 and emanation of Rn-222 in different parts of the repository.
2) For each part of the repository, calculate transport and concurrent decay of radon to the surface of the cap by advection and diffusion.
3) For each part of the repository, calculate the concentration of radon in a house above a gas pathway, and then the dose to an occupant.
4) Calculate the expectation value of dose for the whole repository (i.e. the area weighted dose over the cap) and then calculate the risk.

The spatial variation in doses from radon is expected to be greater than the spatial variation in doses from C-14-bearing gas, reflecting the smaller area of a house compared with a smallholding. Also, there are significant amounts of Ra-226 (the parent of radon gas) disposed in certain parts of the trenches. Therefore, the radon assessment model uses the same assessment model grid, and gridded inventory, as the coastal erosion and human intrusion models (Figure 3.12). This represents the major spatial variations in the existing disposals of Ra-226.

Spatial variations in the inventory of Ra-226 within Vault 9 and the future vaults are not known. Therefore, the Ra-226 inventory in Vault 9 and the future vaults is assumed to be evenly distributed within each vault. Vault 9 is divided into waste in existing and committed containers and future stronger containers. Shielded modules in Vault 10 and Vault 11 are also represented explicitly. As risk depends on the area weighted average flux over the whole cap, uncertainty in the spatial distribution of the inventory in Vault 9 and the future vaults will only have a limited effect on the calculated risk.

The radon assessment uses analytical models, with the calculations undertaken in GoldSim. Compartments and transfers are not needed to model transport of radon from the waste into houses, or to calculate radon concentrations in houses.

The model uses an emanation fraction approach to calculate the proportion of radon emanated from the wastes, based on monitoring data from the trenches. The model then calculates advective and diffusive transport of radon through the final cap, using bulk gas generation rates calculated using SMOGG. Our models do not take credit for the high density of radon, and its tendency to sink, which will tend to reduce radon fluxes through the cap. Radon can be released either through defects (small holes) in the intact cap geomembrane, through the degraded geomembrane, or through a gas vent. The half-life of radon is 3.8 days, so there is significant radioactive decay of radon as it migrates through the waste, profile fill, cap and vent. Concentrations of radon in a house above the cap are then calculated and used to calculate potential doses to a house occupant.

Two approaches are used to calculate radon fluxes into a house:

- an analytical model of radon transport into a house through the foundations, and loss from the house by ventilation;
- an empirical ratio between the concentration of radon in soils and radon in houses.

The first approach can be applied when the cap geomembrane is intact or degraded. The second approach is only relevant when the geomembrane is degraded, and radon releases are more uniform. Although there are large variations in the radon concentrations measured in houses compared with the radon concentrations measured in soils, the best estimate empirical ratio and the analytical model give similar results for the situation where the geomembrane is degraded.

Potential exposure to radon gas in a basement dug into the gas collection layer of the cap is considered as a human intrusion event (see Section 8). The radon model is used to calculate radon fluxes into the gas collection layer, and attenuation by the house foundations and floors, which are then used in the human intrusion assessment to calculate radon concentrations in a basement.

6.4.3 Carbon-14 Inventory and Discretisation

The C-14 assessment model represents C-14 gas releases from the trenches and vaults. There is little C-14 in the trenches compared with the vaults, so detailed spatial representation of the C-14 inventory in the trenches is not important. The trenches are therefore combined into a single source. Spatial variations in the profile thickness over the vaults are not important for the C-14 gas assessment, due to the long half-life of C-14 which means there is no significant decay while migrating through the profile fill. Therefore, spatial variations in the profile thickness do not need to be represented in the assessment model.

The vaults are discretised into individual vaults - this level of discretisation is appropriate given the cap area occupied by the receptor (discussed further in Subsection 6.4.6).

The disposed and projected future C-14 inventory (see Subsection 3.4) is assigned to each vault. Disposals are assumed to occur linearly between the start and end date of disposals to each vault. The inventory is allocated to different waste material types which reflect the release mechanisms, rates and form(s) of release:

- LLW and ILW Graphite - irradiated graphite wastes such as graphite sleeves from fuel elements, and reactor core blocks.
- LLW and ILW activated steel - C-14 is present in the matrix of the metal as an activation product. All steels are treated as mild steel in the assessment model (this is cautious as mild steel corrodes faster than stainless steel). Small amounts of C-14 associated with other metals (nickel, copper, uranium, iron) are grouped together with steels.
- ILW aluminium - C-14 is present in the matrix of the metal as an activation product.
- ILW magnox metal - C-14 is present in the matrix of the metal as an activation product.
- LLW and ILW organics - organic wastes that have not been treated, e.g. ion exchange resins.
- LLW pucks - soft organic wastes, placed into thin wall metal drums and super-compacted.
- Other LLW and ILW - a wide range of other waste types where C-14 is present as surface contamination, including surface contaminated metals and concrete, or in fine granular material, such as soils and blasting grit from metal treatment. It is estimated that 60% of the C-14 inventory in metal LLW is associated with metallic treatment streams (filters and blasting grit) so is assigned to this category. This is considered as an uncertainty in variant calculation cases.

Table 6.1 and Table 6.2 summarise the C-14 inventory per waste material type for each vault, for LLW and ILW respectively.

Table 6.1: C-14 inventory per material type for LLW in the vaults for the reference case (TBq)

	Graphite	Steel	Organics	Pucks	Other
Vault 8	1.0 10^{-1}	5.1 10^{-2}	1.7 10^{-1}	9.4 10^{-2}	7.6 10^{-2}
V9 E&C	4.0 10^{-1}	1.8 10^{-2}	1.4 10^{-1}	2.7 10^{-2}	4.1 10^{-1}
V9 Future	6.8 10^{-1}	2.5 10^{-1}	2.3 10^{-1}	0	1.4 10^{+0}
Vault 10	1.3 10^{+0}	4.0 10^{+0}	6.5 10^{-6}	0	6.4 10^{+0}
V10 SM	0	0	0	0	0
Vault 11	5.9 10^{-1}	6.6 10^{-1}	0	0	1.2 10^{+0}
V11 SM	0	0	0	0	0
Vault 12	7.7 10^{-1}	1.3 10^{-1}	0	0	6.4 10^{-1}

Table 6.2: C-14 inventory per material type for ILW in the vaults for the reference case (TBq)

	Graphite	Steel	Organics	Magnox	Al	Other
Vault 8	0	0	0	0	0	0
V9 E&C	0	0	0	0	0	0
V9 Future	3.0 10^{+1}	1.7 10^{+0}	1.8 10^{-6}	1.0 10^{-1}	3.1 10^{-6}	4.6 10^{-1}
Vault 10	6.8 10^{-1}	3.6 10^{+0}	3.1 10^{-7}	4.0 10^{-2}	5.2 10^{-7}	5.2 10^{-2}
V10 SM	2.0 10^{+1}	3.6 10^{+0}	1.4 10^{-6}	1.6 10^{-2}	2.4 10^{-6}	4.2 10^{-1}
Vault 11	0	4.6 10^{+0}	0	0	0	3.8 10^{-1}
V11 SM	0	7.6 10^{+0}	0	0	0	2.6 10^{-1}
Vault 12	0	6.8 10^{+0}	0	0	0	0

6.4.4 Carbon-14 bearing Gas Generation

C-14 can be present as surface contamination and as matrix contamination. C-14 present as surface contamination may be released immediately on contact with water. C-14 present as matrix contamination (e.g. in graphite or steels) may only be released slowly, either through migrating to the surface of the waste, or concurrent with degradation of the waste.

C-14 gas generation rates from the trenches are calculated using NWS' Simple Model of Gas Generation (SMOGG) [97].

Gas generation and migration from the vaults is modelled in GoldSim. Table 6.3 summarises the assessment models and data for release of C-14 from each waste type from the vaults. The data are substantiated in reference [26] and underpinning references.

Table 6.3: Summary of C-14 release models for each waste type

Material	Release Rate	Form of Release
Graphite	5% of inventory available for release. 95% is 'locked up' in the graphite lattice. Release rate 1.0 10^{-2} y^{-1}.	1% direct to gas 30% to water (organic) 69% to water (inorganic)
LLW and ILW Plate Steels	Congruent with corrosion. Reference corrosion rate 0.005 μm y^{-1}, contaminated thickness 10 mm.	10% direct to gas 90% to water (organic)
ILW magnox metal	Congruent with corrosion. Reference corrosion rate 0.1 μm y^{-1}, contaminated thickness 2 mm.	25% direct to gas 75% to water (organic)
ILW aluminium	Congruent with corrosion. Reference corrosion rate 1 μm y^{-1}, contaminated thickness 2 mm.	2% direct to gas 98% to water (organic)
LLW and ILW organics	Release rate 3.16 10^{-2} y^{-1} (representing release over 100 y).	80% to water (organic) 20% to water (inorganic)
LLW pucks	Release rate 3.16 10^{-2} y^{-1} (representing release over 100 y).	100% to water (uncertain)
'Other' LLW and ILW	Release rate 3.16 10^{-2} y^{-1} (representing release over 100 y).	100% to water (uncertain)

6.4.5 Carbon-14 Bearing Gas Migration and Fate

As described in Table 6.3, a proportion of the C-14 is released directly from the wastes as gas, a proportion is released to solution in organic form, and a proportion is released to solution in inorganic form.

C-14 released from the waste in inorganic form will largely react with cement minerals in the grout, forming carbonate. The exception is C-14 released from graphite, where there is expected to be voidage which is not fully saturated and some graphite which is not directly in contact with grout, so some undissolved inorganic carbon may form. Experts elicited that 5% of the inorganic carbon is released as gas, with the remainder forming carbonate.

The pH inside the containers is expected to be too high for significant microbial activity (around pH 12.5 [6]), so the C-14 will remain 'locked up' in carbonate minerals for the assessment timeframe. The exception is pucks, where microbial niches with lower pH may occur. GRM results calculate the fate of C-14 for this type of system [40], giving the fate of C-14 as: 45% released as gas, 45% incorporated into carbonate, and the remaining 10% remains in solution.

C-14 gas released from the containers is transported rapidly through the vaults to atmosphere (using an arbitrary fast rate of 100 y^{-1}).

C-14 released from the waste in organic form is expected to remain in solution in the container until an aqueous (advective or diffusive) transport pathway out of the container is established. In the reference case, the cap geomembrane remains intact for the full assessment timeframe and there is no release of C-14 in solution from the containers.

Geomembrane degraded - release from containers

Variant cases consider failure of the cap geomembrane before the disruption of the site by coastal erosion.

For the existing and committed wastes disposed in Vault 8 and Vault 9, once the cap geomembrane fails, infiltration through the cap initiates flow down the stacks (Figure 5.2) and C-14 in solution is leached from the containers. The mathematical model assumes there is no retardation of aqueous C-14 by sorption. This is a reasonable assumption because some small organic molecules will not sorb significantly onto cement materials, and inorganic carbon will be at its equilibrium concentration. There could be some retardation by isotopic exchange with carbonate minerals, but this is assumed to be small and is neglected.

For future wastes disposed in Vaults 9 to 12, once the cap geomembrane fails, C-14 in solution begins to diffuse out of containers into a thin film of water in the gaps between container stacks (Figure 5.3). The HHISOs and ILW boxes have different geometries, so diffusive releases are calculated separately for each container type.

A simple model is used to calculate the diffusive fluxes for each container type. The model has two compartments (Figure 6.5) which represent water saturated grouted waste and water in the gaps between the container stacks. The area for diffusive release is equal to the area of the container expected to be perforated by corrosion. We compared the diffusive fluxes calculated using this simple model against the fluxes calculated using a finite elements model with much finer discretisation. The differences between the models were as follows.

- The simple model underestimates the flux out of the container for the first few years, associated with release of aqueous C-14 that is close to the defect. The assessment model is an acceptable simplification because diffusive releases are not expected to start from all containers at the same time.
- The simple model overestimates long-term fluxes out of the container, because it does not represent the geometric resistance of diffusing into a small perforation in the container (for a given perforated area, the geometrical resistance decreases as the number of perforations increases). Therefore, the simple model is cautious.

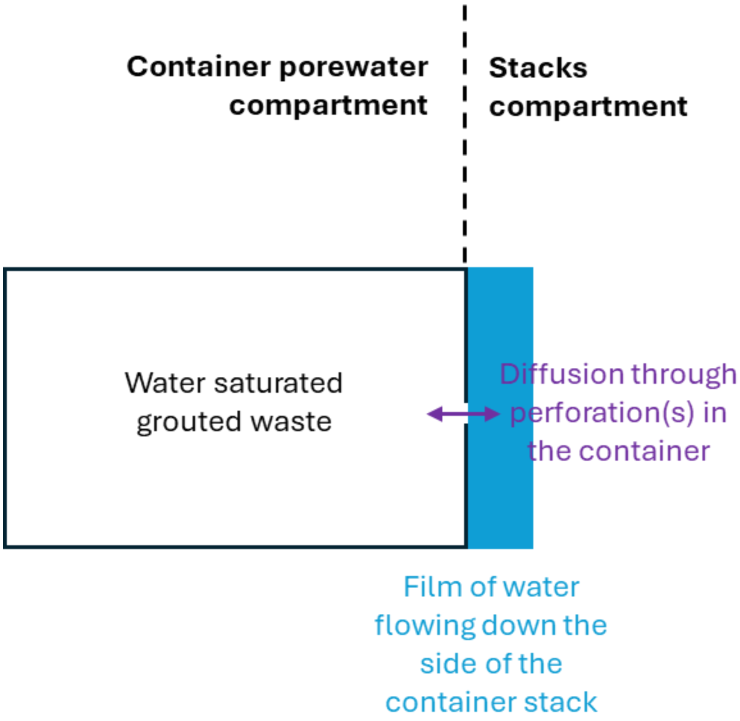

Figure 6.5: Simple 'two compartment' diffusion model

The cap geomembrane will not fail instantaneously over the entire cap. It is considered in the model that each strip of the cap begins to fail after 1,000 years from its initial emplacement. This is consistent with the earliest geomembrane degraded infiltration rates described in the EPA [95]. As the cap geomembrane gradually degrades, the number of containers that encounter the infiltrating water increases. This is represented in a simplified way by linearly increasing the total surface area available for diffusion from zero (before failure of the cap geomembrane) to a maximum value (diffusion from all containers) over 100 years, representing the period of degradation of the cap geomembrane. For the leaching release from existing container types, the model cautiously represents an immediate step change in the infiltration rate at the initial failure time of each cap strip.

Once outside containers, the pH conditions are low enough for microbial activity [6]. The model represents microbial metabolism of small organics in all the vaults as a rapid release of C-14 to gas (at a rate of 100 y^{-1}).

6.4.6 Carbon-14 Release to Biosphere and Exposures

The assessment model calculates risks to the people at greatest risk from exposure to C-14-bearing gas (PRPs, Subsection 2.13.2). Exposure to C-14 gas can occur by inhalation and consumption of foodstuffs. The foodchain pathway is much more significant than the inhalation pathway [40] [90] [98]. Therefore, the PRP chosen is a smallholder with a cautious but plausible habit set, which makes use of the cap area for agriculture and kitchen gardening. This is consistent with the reference PEG C chosen in the 2011 and 2013 assessments [90]. Assuming the receptor is a smallholder is cautious as it maximises carbon intake from the cap and therefore maximises the potential doses from C-14 [90].

In the reference case, the C-14 gas is released from the vaults and trenches through defects in the geomembrane across the cap. Releases and doses will vary with location on the cap. Given that a smallholding could be located anywhere on the cap, it is appropriate to use area-weighted average gas fluxes and therefore area-weighted average doses to assess risks. A variant case considers releases through an open vent at the crest of the cap.

The model considers that all $^{14}CH_4$ will be oxidised to $^{14}CO_2$ in the cap soils, so all the C-14 will be in the form of CO_2 once it is released from the soil surface. This is realistic for well-drained agricultural soils, and cautious for shallow or poorly drained soils. It might also be cautious for situations where large fluxes of methane are discharged through a small area, e.g. through an open vent while the trench wastes are degrading. However, in this case, the microbial populations responsible for methane oxidation in soils are likely to be stimulated by the high methane flux, increasing the oxidation rate.

A mathematical model is used to calculate concentrations of C-14 in plants grown in topsoil directly above the cap, due to uptake by photosynthesis and transpiration. A specific activity model is then used to calculate the uptake of C-14 by animals consuming these plants, by assuming that the specific activity of C-14 in animal products is equal to the specific activity of C-14 in plants consumed by the animal.

Only the ingestion pathway is included in the assessment models. Doses are calculated to a smallholder who is assumed to live on the cap, consume foodstuffs from a kitchen garden and keep goats for milk production. Doses to adults, children and infants are considered for the reference case, using age-specific food intakes and ingestion dose coefficients. This is done to demonstrate that differences in the doses to the range of age groups are small (within a factor of 2) compared to other uncertainties, so only adults are considered in the variant cases.

6.4.7 Radon Gas Inventory, Discretisation and Generation

Radon gas is generated by radioactive decay of radium (Ra-226). Ra-226 is primarily associated with relatively small volume waste streams, such as those containing Ra-226 sealed sources and radium paint. Therefore, the distribution of Ra-226 in the trenches and vaults is spatially heterogeneous (Figure 6.6).

The forward (future) inventory for the 2011 ESC included a substantial amount of Ra-226 in the form of luminised paint in waste stream 7S01. It has since been confirmed that the radium inventory in stream 7S01 was incorrect, so this radium will not be disposed to the LLWR and is not included in the inventory for the 2026 ESC.

Location	**Ra-226 (TBq)**
Trenches	0.29
Vault 8	0.09
Vault 9/9A	0.05
Vault 10	0.0003
Vault 10 ES1	0.0001
Vault 11	0
Vault 11 ES1	0
Vault 12	0

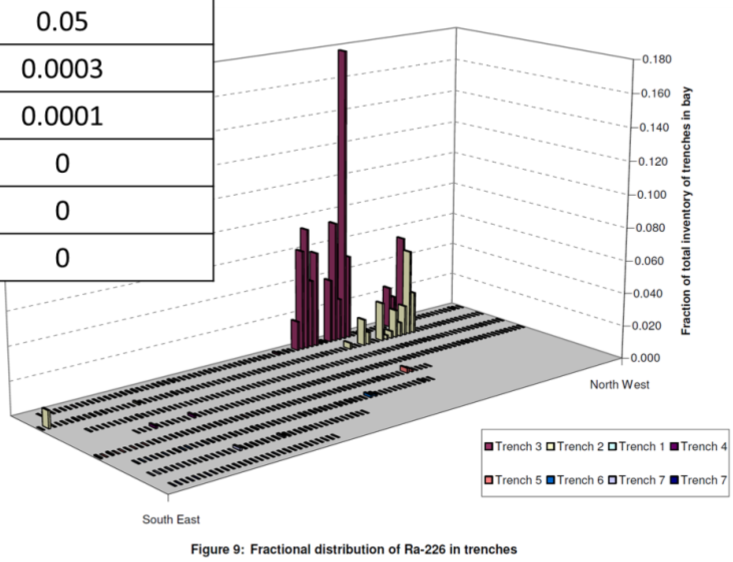

Figure 6.6: Ra-226 in the trenches (plot of spatial distribution in the trenches from [62])

Ra-226 is also ingrown in the repository from radionuclides including U-238, U-234, Th-230, etc. However, the amount of ingrowth during the assessment timeframe is small. Disposed Ra-226 is the main source of radon.

The radon assessment uses a discretised grid to represent the heterogeneity in the Ra-226 inventory in the trenches and vaults and variations in profile thickness across the cap (Figure 3.12). The grid is consistent with the discretisation used in the coastal erosion and human intrusion assessments. It reflects key repository features and the spatial distribution of the anticipated key radionuclides for doses from these pathways.

Decay of the Ra-226 inventory is calculated for the elapsed time from the final date of disposals to each vault or trench.

An emanation fraction of 15% is used for the trenches, based on radon monitoring data from probes in the interim trench cap [26] which gave a range of 7% to 15%. The lower bound of 7% is used for the ISO containers in the vaults, as the fraction of radon released from the grouted ISO containers is expected to be lower than the fraction released from un-containerised trench wastes.

6.4.8 Radon Gas Migration

Radon gas will diffuse through the waste, profile and cap material. It will also be advected by bulk gas. Bulk gas generation rates were calculated using the SMOGG model [99]. Bulk gas generation from the trenches is much greater than the bulk gas generation from the vaults.

The travel time through the waste, profile and cap is likely to be substantially longer than the half-life of radon, resulting in significant attenuation. The travel time through the waste is excluded from the assessment calculations as it is different for each layer of containers. This is a simplifying assumption. It also cautious as it results in underestimation of radioactive decay of radon from lower-level containers and overestimation of the fluxes. The degree of caution is uncertain as emanation fractions are estimated based on trench monitoring data, which include an element of transport and decay between source and monitoring point. Side calculations are used to explore the significance of this bias in the assessment calculations (covered in Subsection 6.6.1).

The thickness of the cap is 3 m everywhere, but the thickness of the profile material varies across the repository (Figure 6.7).

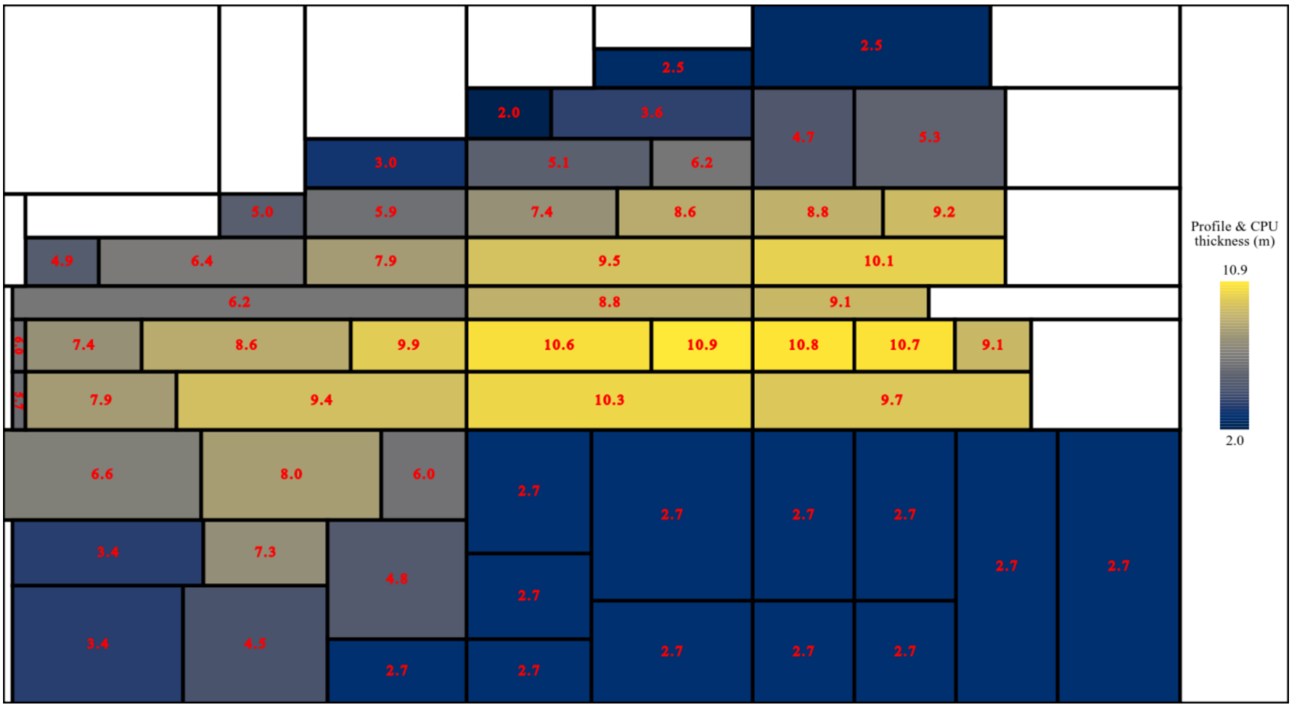

Figure 6.7: Average thickness of profile material over the waste for each model grid cell [26]

Analytical models have been developed for the 2026 ESC and implemented in GoldSim to calculate transport of radon by diffusion and advection with bulk gas, and decay of radon. These models are described in detail in reference [26]. As analytical models are used, compartments and transfers between compartments are not needed to calculate transport of radon gas. A side calculation using a compartmental model gave very similar results to the analytical model [26]. The two models have different biases, so this builds confidence in the assessment results.

In the reference case, advective velocities are calculated for each model grid cell (Figure 3.12) using local bulk gas generation rates, with no lateral spreading. This is a cautious approach as it minimises the pathlength for transport of radon and maximises the transport velocity. Variant cases consider the impact of lateral gas spreading.

6.4.9 Exposure to Radon Gas

Radon gas could accumulate in houses (or other buildings) constructed on the cap. The footprint of a detached house on the cap is expected to be around 75 m^2. Exposure to radon gas is through inhalation.

People can also be exposed to radon gas outdoors. However, occupancy outdoors is expected to be significantly less than indoors, and radon is rapidly diluted and dispersed. Therefore, doses outdoors are significantly lower than indoors. Indoor occupancy is assumed to be 90% of the year.

The dose factor for Rn-222 and its progeny in secular equilibrium is 20 nSv per (Bq hr m^{-3}) [57]. The dose factor relates the Rn-222 unit equilibrium equivalent concentration (EEC) in air to the effective dose rate. The EEC is calculated using an equilibrium factor to calculate the equivalent concentration of Rn-222 with its short-lived progeny in secular equilibrium with it and has a typical value of 0.4 [57]. There is no distinction made by age, with it being implicitly assumed that the smaller tissue and organ masses appropriate at younger ages are approximately compensated by the smaller breathing rates at those ages. Therefore, the same dose factor is applicable to adults and children.

Geomembrane intact

The gas plume above a defect is expected to be narrow and is considered in the model to spread out in hardcore placed as part of the foundations of the house. Therefore, a house may capture all gas being released through a defect. It is possible some gas would be released around the perimeter of the house, if it is not able to freely enter the house, and from effects such as atmospheric pumping. However, these effects are difficult to quantify, so cautiously all the gas is conceptualised to enter the house. Radon concentrations in the house are calculated by modelling advection-diffusion through the foundation layers into the house atmosphere.

If the cap vent(s) is(are) left open post-PoA, little gas is expected to be released through defects, so radon would only enter houses built over the vent(s).

Geomembrane degraded

The radon concentration in a house can be calculated using two approaches: using an empirical relationship between radon concentrations measured in soils and radon concentrations measured in houses (taken to be a factor of 1.21 10^{-3} [26]); or by modelling advection-diffusion through the foundations and floor. The two approaches give similar results.

6.4.10 Risk Calculations

In the gas assessment, a dosimetric approach is used to calculate conditional radiological risk from dose. This gives slightly higher risks than an epidemiological approach.

Some of the potential exposure situations considered in the gas assessment are likely to occur (probability close to one) while others are less likely (probability much less than one). Our calculations of radiological risk include the probability of the exposure situation.

Ordnance Survey map data have been used to calculate the spatial density of buildings on the West Cumbrian coast. The results are fed into a statistical analysis to calculate the annualised probabilities of different exposure situations. The approach used is analogous to that used in the groundwater pathway assessment to calculate the annual probability a well exists in the area of contaminated groundwater (Subsection 5.4.7.1). The people living in a house on the cap and consuming foodstuffs from a smallholding on the cap could be the same people, so the combined risks from radon and C-14 gas are also assessed.

With the cap vent closed, the relevant probability is that of a smallholding anywhere on the cap, or the probability of a house on the cap, noting that only some houses would be over defects (small holes) in the geomembrane. Risks are averaged over the cap area as it is not known where the settlement would be located. With the cap vent open, the relevant probability is that of a house or smallholding over the vent - with or without the kitchen garden also being located over the vent.

The probabilities of the different exposure situations are summarised in Table 6.4. While it is likely that there would be a house on the cap, and moderately likely there would be a smallholding on the cap, some of the potential situations of interest with an open cap vent are unlikely. Note that these probabilities assume there is no preference for occupancy of a particular area of the cap. This is consistent with the major area of the cap being gently sloping (1:25 slope).

Table 6.4: Summary of the probabilities of the exposure situations for gas assessment

Exposure situation	Probability
House on the cap	$8.3\ 10^{-1}$
Smallholding on the cap	$3.4\ 10^{-1}$
House on the vent	$6.7\ 10^{-3}$
Smallholding on the vent	$1.1\ 10^{-2}$
Smallholding on the vent and house on the vent (the kitchen garden is also assumed to be on the vent)	$6.6\ 10^{-4}$
Smallholding on the vent and kitchen garden on the vent (the house is assumed not to be on the vent)	$7.5\ 10^{-4}$
Smallholding on the vent and house not on the vent and kitchen garden not on the vent	$1.0\ 10^{-2}$

6.5 Carbon-14 – Results

Subsection 6.5.1 defines the reference case and alternative assessment cases. Subsection 6.5.2 describes the results of the reference case at a level intended to illustrate the functioning of the model and provide reference case results. Subsection 6.5.3 describes the results of the alternative assessment cases more briefly and compares the results to the reference case results. Fuller presentation of the results is given in reference [26].

6.5.1 Assessment Calculation Cases

A reference calculation case has been developed which provides an assessment of impacts from C-14-bearing gas. This case uses best estimate parameter values but takes a cautious approach where the most realistic representation cannot be robustly quantified (Subsection 2.5.1).

The reference case is evaluated deterministically. Deterministic variant calculation cases area then used to explore the key uncertainties identified in the model and data. Table 6.5 lists the variant cases and describes the changes compared with the reference case. The uncertainties can be grouped under the headings of engineering performance uncertainty, conceptual model uncertainty, parameter uncertainty, and 'what-if' situations.

Table 6.5: Calculation cases for C-14 gas assessment

Case	Description
Reference case	
Reference	The pH in the waste containers is too high for microbial activity, except in niches on the surfaces of cellulosic wastes in pucks. The cap geomembrane remains intact for the full assessment timeframe, so there is no release of C-14 in solution from the containers. Risks include the probability (likelihood) of a smallholding on the cap.
Engineering performance uncertainty cases	
Early failure of the cap geomembrane	There is significant probability the cap geomembrane would fail within the assessment timeframe. The geomembrane in each strip of the cap fails at 1,000 years after placement, initiating flow of water outside the waste stacks. pH outside the waste stacks is low enough for microbial activity, and microbes are assumed to metabolise all C-14 in small organic molecules in all vaults.
Conceptual model uncertainty cases	
Cap vent is not closed	Cap mushroom vent remains open post-PoA to manage bulk gas pressures. Risks include the probability (likelihood) of a smallholding/house/kitchen garden on the vent.
No SM ILW	As reference case, but no ILW inventory that is suitable for disposal in shielded modules.
No ILW	As reference case, but LLW only.
Metal melting	30% of the committed and future LLW steel inventory is present in metal ingots instead of blasting grit and filters ('other').

Case	Description
Slags	30% of the committed and future LLW steel inventory is present in slags instead of blasting grit and filters ('other').
Duration of institutional control	The duration of the period is increased from 100 years to 300 years or decreased to 0 years. The latter provides an upper bound on the sensitivity of the potential impacts to reducing the duration of the period of institutional control.
Hold-up of C-14 gas in the vaults	There is a delay associated with the time to migrate to the cap vent or through defects in the geomembrane.
No microbial activity in pucks	No microbial metabolism of C-14 in niches of cellulosic pucks in vault containers.
Parameter sensitivity cases	
Metal corrosion rates	Higher than expected steel, magnox and aluminium corrosion rates.
Higher proportion of C-14 released directly to methane from steels	Explores sensitivity to the proportion of C-14 released directly to methane from steels. Increased from 10% in the reference case to 25%.
Release rate from organics, pucks and 'other'	Assumes the release rate is three times slower than the reference case so releases continue further into the post-PoA period.
Higher plant C-14 uptake	Parameter sensitivity case for uptake of C-14 into plants. Maximum values are chosen for all relevant parameters.

Case	Description
Low likelihood 'what-if' cases	
Faster steel and magnox corrosion	pH in the containers is significantly lower than expected (pH 10.5 to 11), resulting in significantly faster steel and magnox corrosion (but slower aluminium corrosion).
Microbial metabolism of C-14 in solution	As above, plus methanogens are active in containers and metabolising (100% of) small organic molecules to methane. 5% of inorganic carbon is also assumed to be microbially metabolised forming methane.
Flow down the waste stacks	As above, plus the cap geomembrane in each strip of the cap fails at 1,000 years after placement. The containers and protection units are also assumed to fail at this time, allowing flow down through the waste stacks.
Microbes resource limited	As the 'Early failure of the cap geomembrane' case, but assuming that Vaults 10-12 contain insufficient organic material to support microbial populations, so C-14 in small organics molecules is not metabolised to gas in the vaults.
No containers	As the 'Early failure of the cap geomembrane' case, but no credit is taken for the presence of containers for future wastes, so a larger surface area is available for diffusion from the wasteform.
All releases from 'other' are to gas	This is explored as a mathematically possible bounding case, assuming release over a 300-year period.

6.5.2 Analysis of the Reference Case

The key features of the reference case model for C-14 are:

- C-14 gas is generated abiotically as graphite waste contacts water and as matrix contaminated metals corrode.
- The pH in waste containers (pH ~12.5) is too high for microbial activity, except in niches on the surfaces of cellulosic wastes in pucks. Microbes can metabolise cellulosic waste in pucks generating C-14 gas.
- The geomembrane remains intact for the full assessment timeframe, until the latest plausible disruption of the repository by coastal erosion (1,750 y after present), so there is no release of C-14 in solution from the waste containers.
- Risks include the probability (likelihood) of a smallholding on the cap.
- The model runs from 1959, the year of first disposals to the LLWR, and represents gradual emplacement of wastes using recorded and anticipated future inventories, with a constant rate of disposal to each trench or vault between the dates of opening and closure of each trench or vault. Radioactive decay is modelled from the time of disposal.

Migration and flux of C-14

The time-dependent activity distribution of C-14 in Vault 10 is shown in Figure 6.8. The activity distributions in other vaults are similar, although the total activities and timings are affected by the different initial inventories and disposal times.

There is an initial rapid release of C-14 from the wastes over the first ~100 y after disposal, from C-14 in organics, pucks, 'other' and graphite. The proportion of C-14 inventory in metals releases much more slowly; by the end of the assessment timeframe, only 0.2% of the C-14 in steels has been released and 36% of the C-14 in magnox metal. Note that 95% of the C-14 inventory in graphite remains in the waste; only the 5% of the inventory available for release is included in Figure 6.8.

C-14 in the wastes is released directly to atmosphere (for a proportion of the inventory in graphite and metals) or to solution in the containers, where it remains in the container porewater or locked up in carbonate minerals. Figure 6.8 shows that there is a very similar proportion of C-14 that remains in carbonate or in the container porewater. There is no aqueous release from the containers and the pH in the containers is too high for microbial metabolism except for microbial niches on cellulosic waste in pucks.

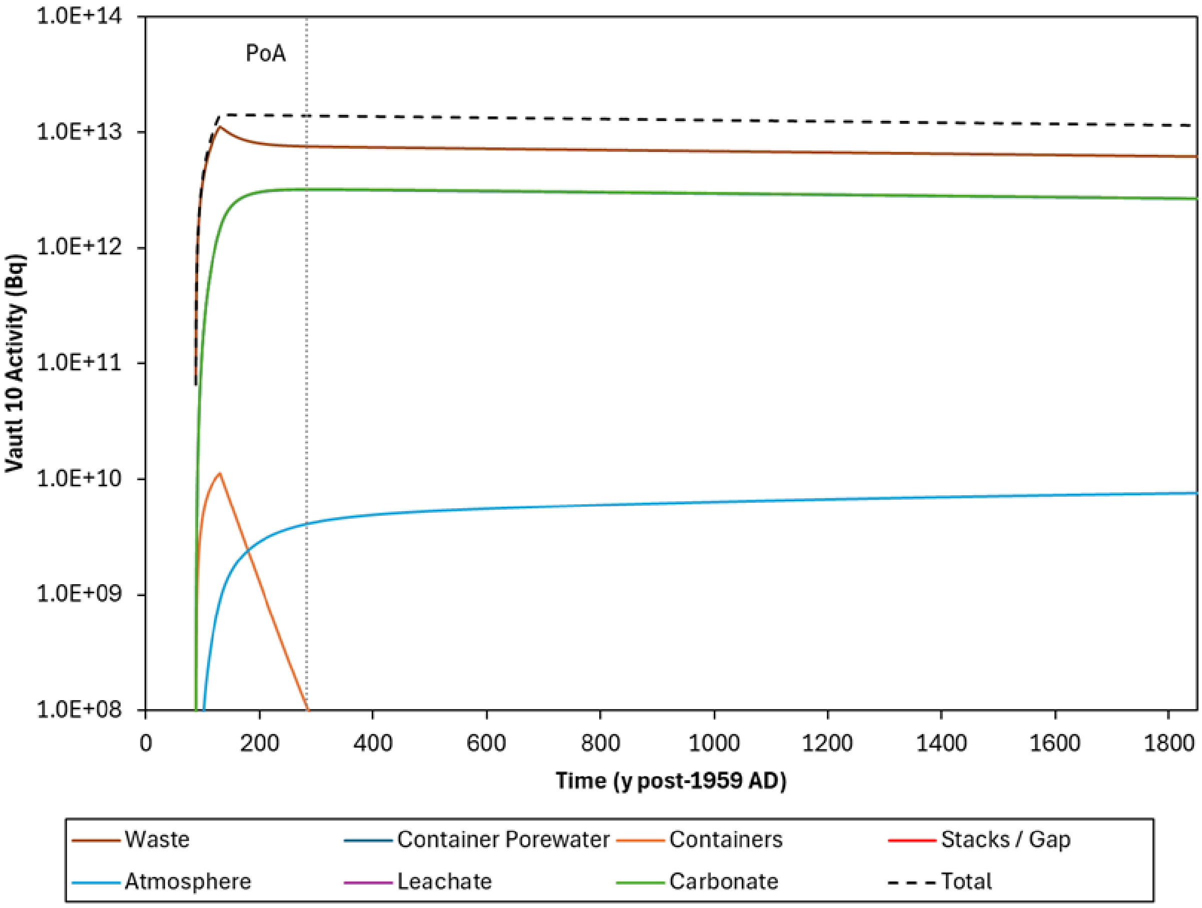

Figure 6.8: Activity distribution of C-14 in Vault 10. The end of the PoA is marked at 283 y (100 y after the end of disposals). Reference case.

Figure 6.9 shows the total C-14 gas flux to atmosphere from the trenches and from Vaults 8 to 12 (including shielded modules). The gas flux from the trenches is a time series based on SMOGG, starting in 1963 (end of disposals to Trench 1). The peak gas flux from the trenches arises in 1985, during the PoA.

The peak gas flux from the vaults also occurs during the PoA and is primarily due to C-14 released from pucks and graphite. At the end of the PoA, the main contributor to the C-14 gas flux is graphite. The longer-term release of C-14 is controlled by the slower release of C-14 from magnox and steels, congruent with corrosion. C-14 released from aluminium is a very small contributor to the total gas flux, due to the relatively small initial inventory of C-14 in aluminium. In the reference case, there is no release of C-14 gas from 'other' or organic wastes in the vaults; all C-14 released from these wastes remains in solution in containers in the form of small organics.

The highest flux per unit area arises from the shielded module in Vault 10, where there is a high concentration of C-14 in graphite (Figure 6.10). The next highest peak flux is from the future disposals to Vault 9, which also have a large graphite inventory.

The C-14 gas fluxes per unit inventory from graphite and irradiated steel wastes have decreased significantly compared with the 2013 assessment. This is due to slower steel corrosion rates, improved understanding of the form of release from steels, and improved understanding of the availability and rate of release from graphite. The slower corrosion rates reflect updates to the near-field conceptual model (pH in the containers) and significantly updated information on steel corrosion rates under different conditions. Improved understanding of the form of C-14 release from irradiated steels comes from the EC CAST project. Improved understanding of the availability, rate and form of C-14 release from graphite comes from experiments on reactor core graphite, and reviews undertaken in the UK and internationally over the last decade.

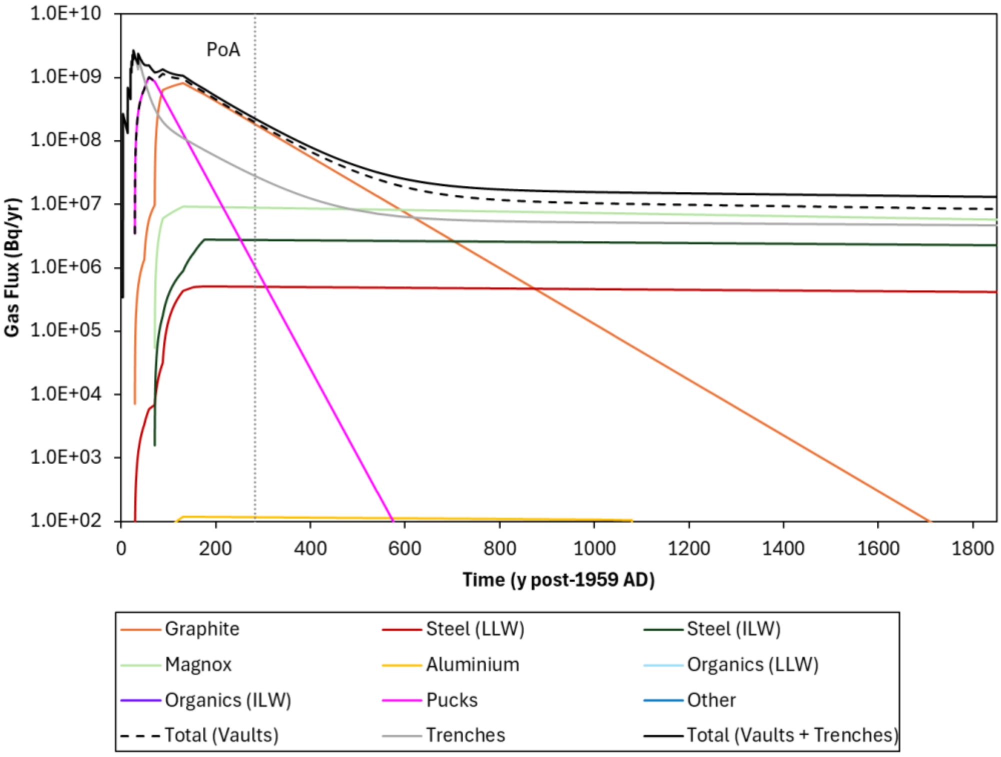

Figure 6.9: **Total C-14 gas flux from the trenches and all vaults (by waste type). Reference case.**

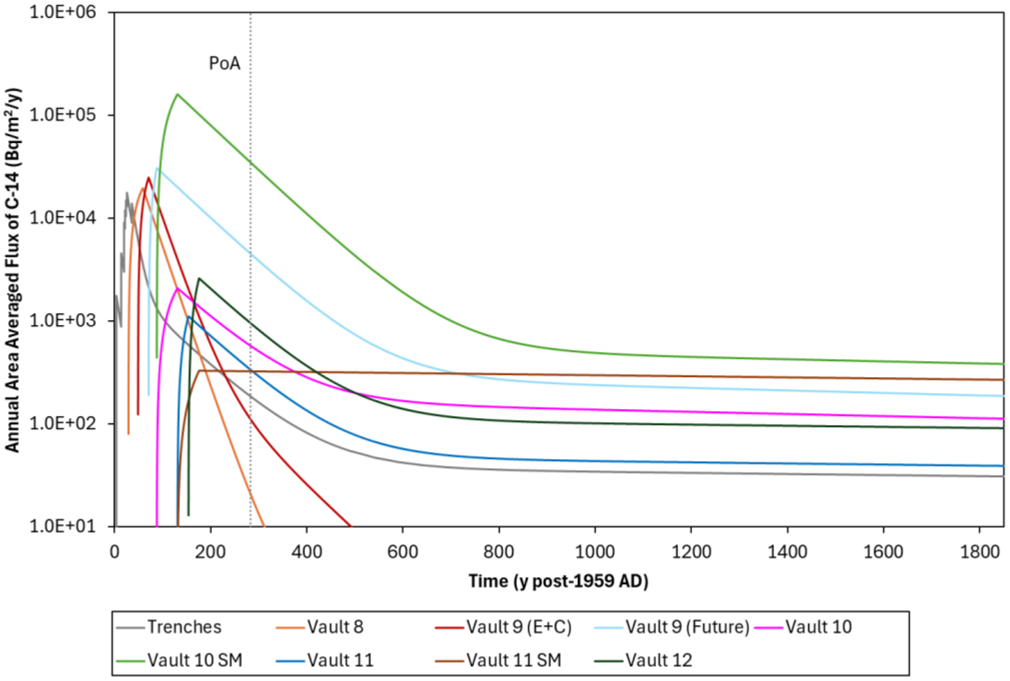

Figure 6.10: Total area-averaged C-14 gas flux from the trenches and each vault (over all waste types). Reference case.

Annual effective doses and risks

The annual effective dose from ingestion to the adult smallholder from releases from all vaults (due to the gas flux averaged over the area of Vaults 8 to 12) is shown in Figure 6.11. The peak dose occurs immediately after the end of the PoA and is primarily due to C-14 released from graphite, with C-14 released from magnox and steels dominating the longer-term doses.

Figure 6.12 shows the annual effective dose from releases from the whole repository. Contributions from the vaults and trenches are shown. The highest contribution is from the vaults.

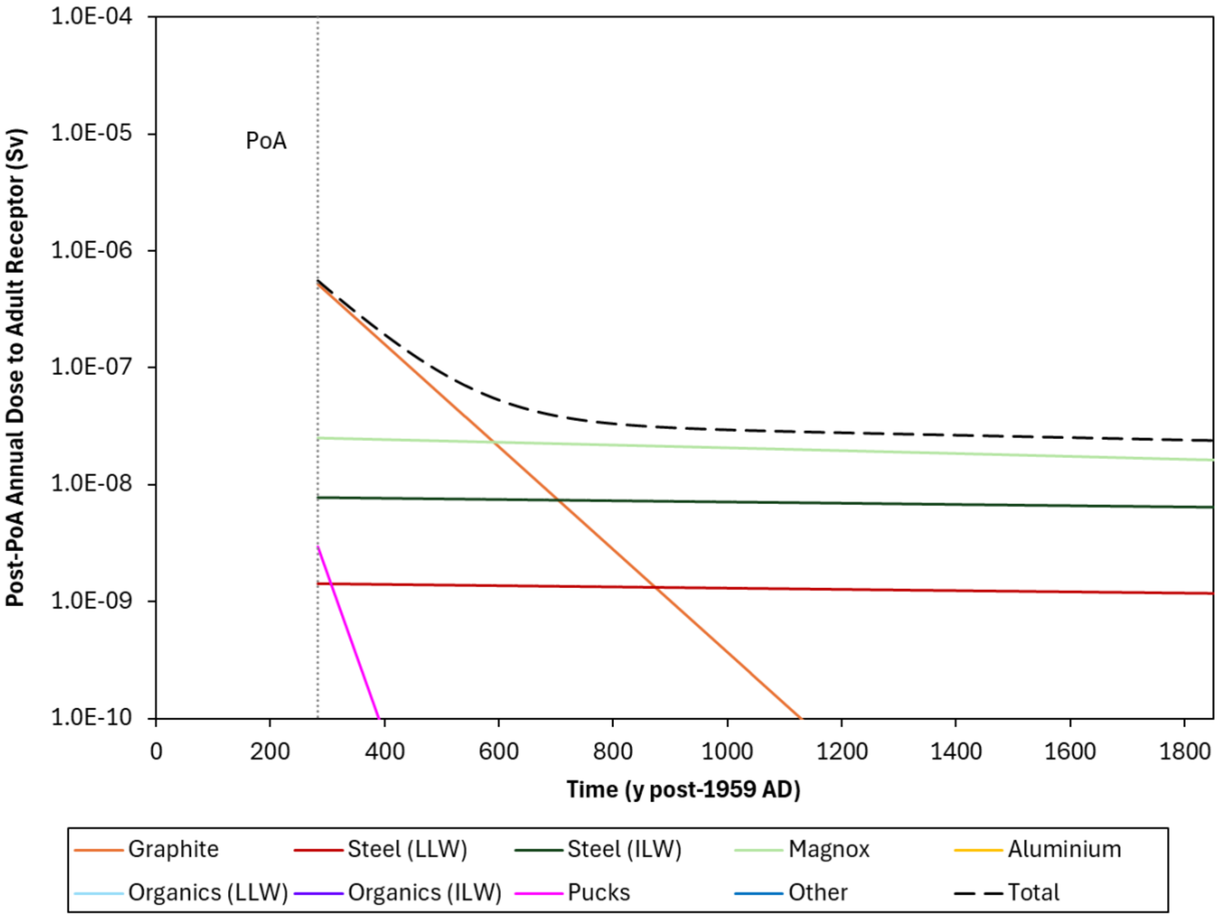

Figure 6.11: **Calculated annual effective dose for area-averaged flux from all vaults. Reference case.**

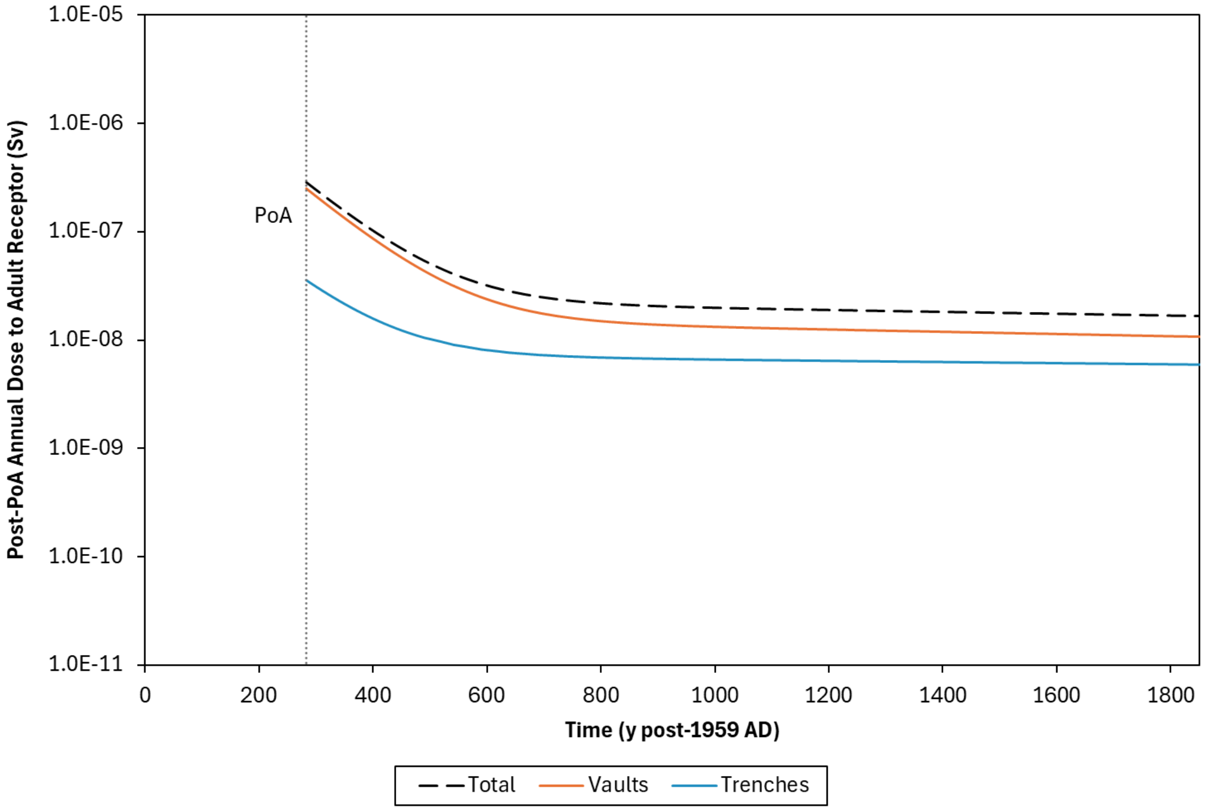

Figure 6.12: Calculated annual effective doses from area-averaged flux from the repository, showing the contributions of the trenches and vaults. Reference case.

Doses to children and infants consuming foodstuffs from the smallholding were also calculated. Doses to children are 5% higher than those to adults, and doses to infants are almost a factor of 2 higher than those to adults. Therefore, the differences are small compared with other uncertainties, and only doses to adults are considered in the variant cases.

Figure 6.13 summarises the calculated peak doses to adults. The peak annual dose from the area-averaged flux over the whole repository cap is 0.29 μSv.

The doses averaged over the whole repository are used to calculate risks to an adult smallholder on the cap. These risks take account of the probability that a smallholding is located on the cap, 0.34 (calculated from data describing the spatial density of settlements on the West Cumbrian coast).

Table 6.6 summarises the peak risk from each waste type. The total peak risk is 5.5 10^{-9} y^{-1} and occurs at the end of the PoA. This is a factor of almost 200 below the regulatory risk guidance level.

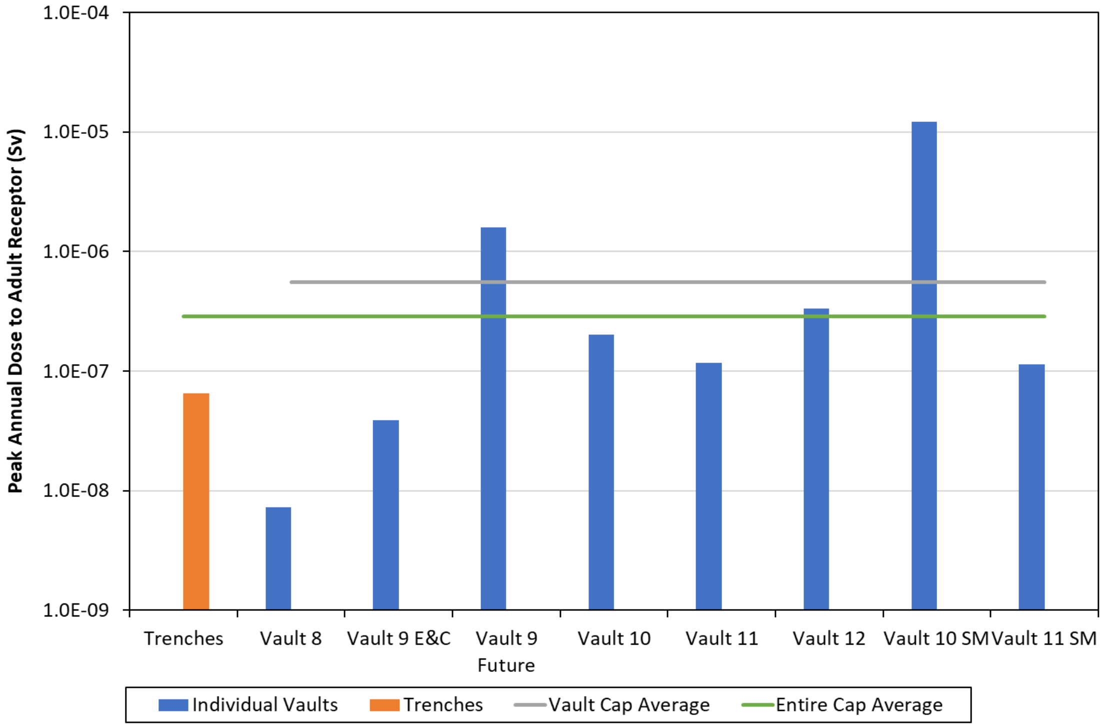

Figure 6.13: **Calculated peak effective doses for area-averaged flux from the trenches, each vault, all vaults and the whole cap area. Reference case.**

Table 6.6: **Peak risk (y^{-1}) for area-average flux from the whole repository, by waste type. Reference case.**

Waste type	Peak risk (y^{-1})	Waste type	Peak risk (y^{-1})
Graphite	$4.5\ 10^{-9}$	Organics (LLW)	$0.0\ 10^{0}$
Steel (LLW)	$1.2\ 10^{-11}$	Organics (ILW)	$0.0\ 10^{0}$
Steel (ILW)	$6.7\ 10^{-11}$	Pucks	$2.6\ 10^{-11}$
Magnox	$2.2\ 10^{-10}$	Other	$0.0\ 10^{0}$
Aluminium	$2.9\ 10^{-15}$	**Total**	**$5.5\ 10^{-9}$**

6.5.3 Alternative Cases

Results for the alternative cases, as identified in Table 6.5, are more fully presented in reference [26]. Here we present results from the early geomembrane failure case and cap vent open case and summarise results for the other cases.

Early geomembrane failure case

There is a substantial probability the cap geomembrane would be intact when the repository starts to be disrupted by coastal erosion [13]. However, there is a non-trivial probability that the cap geomembrane would degrade before the repository starts to be disrupted by coastal erosion. In this calculation case, each strip of the cap geomembrane is assumed to fail 1,000 years after placement. C-14 has a half-life of 5,700 years so assuming a cap geomembrane failure 500 years earlier or later than this value would not be expected to affect results significantly.

The key difference from the reference case is that failure of the cap geomembrane leads to increased infiltration into the vaults, and release of C-14 in solution from the containers into the vaults. Release from existing and committed containers is by advection - is it cautiously assumed that there is a step increase in infiltration at the start of geomembrane failure for each cap strip, which is instantly seen by all containers. Release from future stronger containers, and new containers for ILW, is by diffusion - it is assumed that the diffusive releases increase over a period of 100 years from the initial failure of each strip of the geomembrane. 3% of the area of the container sides is available for diffusive release (the basis for this value is described in reference [26]).

C-14-bearing small organic molecules released from the containers are microbially metabolised to C-14 gas.

Figure 6.14 shows the total C-14 gas flux to the atmosphere from Vaults 8-12 (including shielded modules). The behaviour is identical to the reference case until the cap geomembrane begins to fail. The peak flux now occurs immediately after the start of geomembrane failure, at 3038. There is an increase in flux from all waste types after geomembrane failure, as all have some proportion of C-14 released to small organics in solution, which are now metabolised to methane and released to atmosphere. The main contributors to the peak flux are now C-14 released from 'other' and organic (LLW) waste types.

The peak fluxes from existing and committed containers are likely to be an overestimate, as they do not take credit for the barrier to leaching provided by the containers. The highest flux per unit area in this case is from the existing and committed wastes in Vault 9.

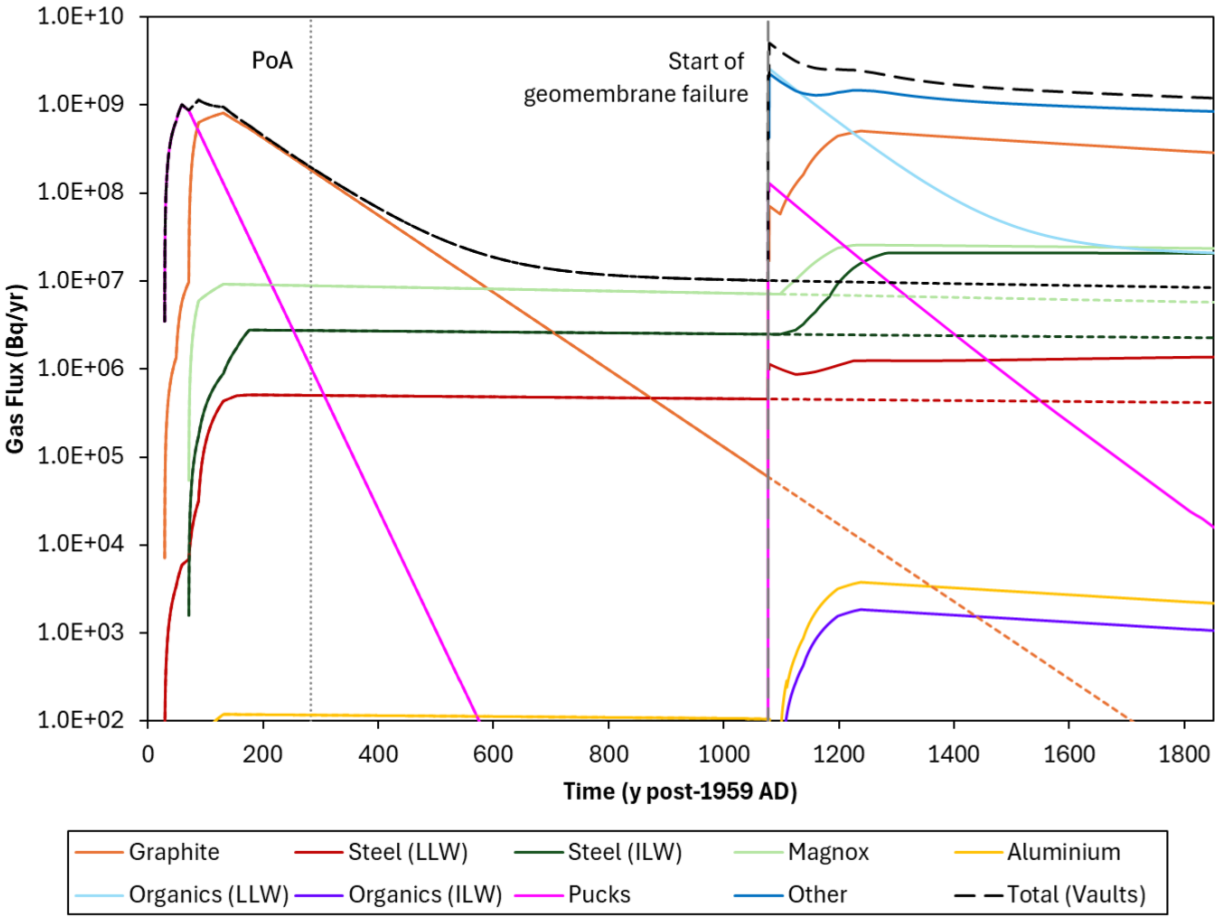

Figure 6.14: Total C-14 gas flux from all vaults (by waste type). Geomembrane failure case (solid lines) compared with reference case (dashed lines).

Figure 6.15 shows the effective dose from releases from the whole repository, averaged over the area of the whole repository. After geomembrane failure, the risk is completely dominated by releases from the vaults. The peak dose from the area-averaged flux over the whole repository cap is 6.5 μSv.

The doses averaged over the whole repository are used to calculate risks to an adult smallholder on the cap. As in the reference case, these risks take account of the probability that a smallholding is located on the cap (0.34). The peak risk is 1.2 10^{-7} y^{-1} and occurs at the end of the PoA. This is a factor of 8 below the regulatory risk guidance level, but a factor of 20 higher than the reference case.

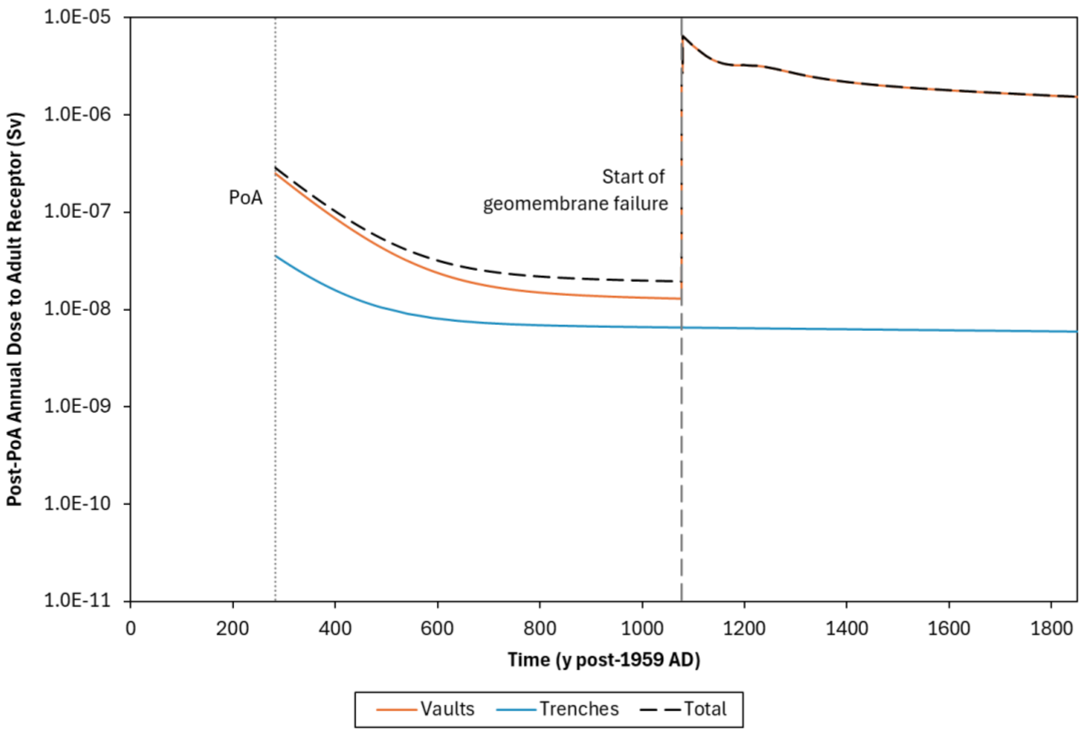

Figure 6.15: Calculated annual effective doses for area-averaged flux from the repository, showing the contributions of the trenches and vaults. Geomembrane failure case (solid lines) compared with reference case (dashed lines).

Table 6.7: Peak risk (y^{-1}) for area-average flux from the whole repository, by waste type. Geomembrane Failure Case.

Waste type	Peak risk (y^{-1})	Waste type	Peak risk (y^{-1})
Graphite	$1.2\ 10^{-8}$	Organics (LLW)	$6.3\ 10^{-8}$
Steel (LLW)	$3.3\ 10^{-11}$	Organics (ILW)	$4.5\ 10^{-14}$
Steel (ILW)	$5.1\ 10^{-10}$	Pucks	$3.2\ 10^{-9}$
Magnox	$6.3\ 10^{-10}$	Other	$5.6\ 10^{-8}$
Aluminium	$9.2\ 10^{-14}$	**Total**	**$1.2\ 10^{-7}$**

Cap vent is not closed case

The cap design includes a single 'mushroom' vent at the crest of the cap (Figure 6.16) [5]. In the 2011 ESC it was anticipated that by the end of the PoA, bulk gas generation would decrease to a level where the vent could be sealed, without risk of significant pressurisation of the repository leading to damage to the cap. Updated understanding of bulk gas generation, the cap performance and properties mean that it is now more uncertain whether the vent will be closed [26]. The vent (or vents) might need to be left open post-PoA to passively manage gas pressures in the repository. Therefore, cases with the vent closed and the vent left open are assessed. Final decisions on the vent design and whether the vent will be closed before the end of active institutional control will be made later.

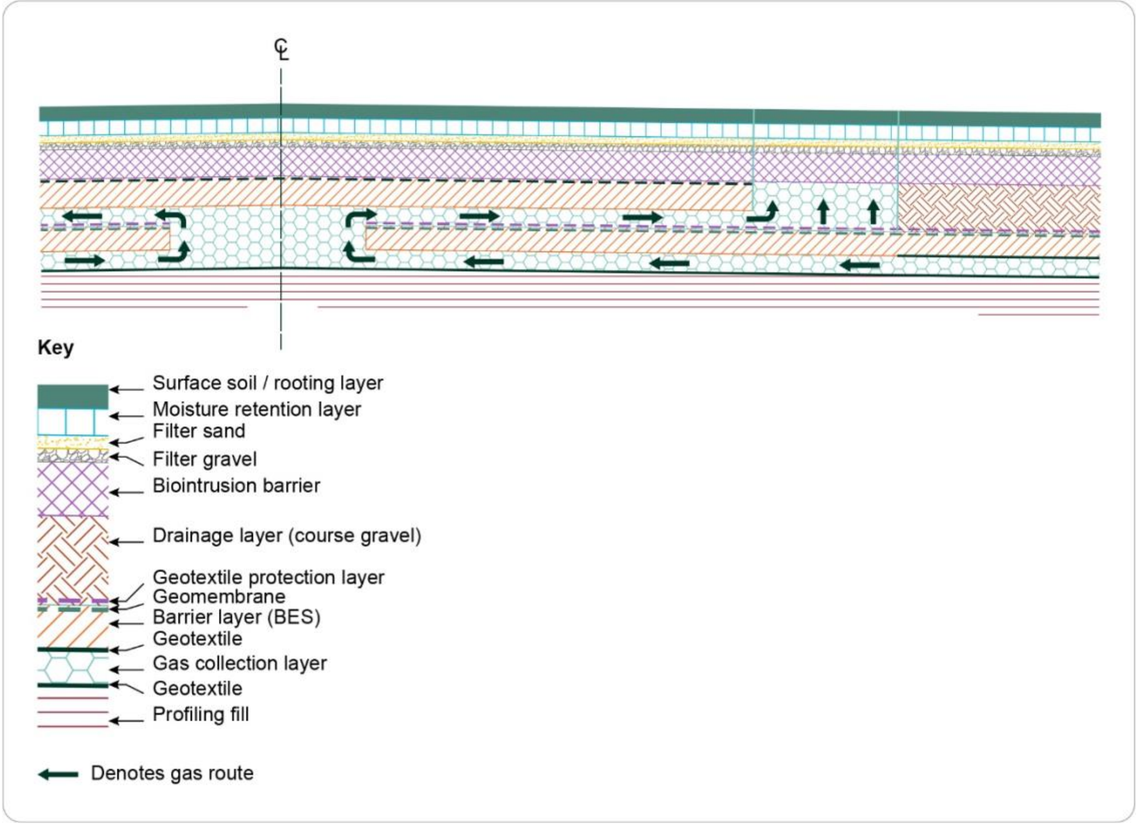

Figure 6.16: Cross-section through the 'mushroom' vent concept [5]. Note figure originates from the 2011 ESC and therefore does not include recent refinements to the cap layers.

In this variant case, the cap mushroom vent (Figure 6.16) remains open post-PoA to manage bulk gas pressures. The flux of C-14 to atmosphere is unchanged from the reference case, but the flux is now assumed to all be released through the vent rather than through defects (small holes) in the geomembrane.

Doses are calculated to an adult smallholder, where the vent is located fully beneath the smallholding. Doses are calculated for a case where the kitchen garden is located maximally above the vent and for a case where the kitchen garden is not located above the vent. Doses are a factor of 4 times higher when the kitchen garden is located above the vent (peak annual dose 25 μSv compared with 5.8 μSv), due to the additional contamination of fruit and vegetables grown in the garden. Doses are significantly higher than in the reference case, since the gas flux from all vaults and trenches is released into the vent, rather than distributed across the cap.

The doses are used to calculate risks to an adult smallholder on the cap. Risks are calculated for three exposure situations, each with different probabilities (Table 6.4): the smallholding, house and kitchen garden are on the vent (probability 6.6 10^{-4}, for combination with radon gas risks); the smallholding and kitchen garden are on the vent but the house is not on the vent (probability 7.5 10^{-4}); and the smallholding is on the vent but the house and kitchen garden are not on the vent (probability 1.0 10^{-2}). Table 6.8 summarises the peak risks in each situation.

The calculated doses are highest when the kitchen garden is on the vent and lowest when the kitchen garden is not on the vent. However, the probability of the kitchen garden being on the vent is considerably lower than the probability of the kitchen garden not being on the vent (Table 6.4). As radiological risk is calculated as the product of the dose, risk coefficient and the probability of the situation, the risks are highest when the kitchen garden not on the vent. The peak risk is 3.4 10^{-9} y^{-1}, a factor of almost 300 below the regulatory risk guidance level.

Table 6.8: Peak risks (y^{-1}) from the whole repository, and by waste type. Cap vent not closed case, with smallholding over vent.

Waste type	Kitchen Garden and House over Vent	Kitchen Garden over Vent, House not over Vent	Kitchen Garden and House not over Vent
Graphite	7.6 10^{-10}	8.6 10^{-10}	2.8 10^{-9}
Steel (LLW)	2.1 10^{-12}	2.4 10^{-12}	7.6 10^{-12}
Steel (ILW)	1.1 10^{-11}	1.3 10^{-11}	4.2 10^{-11}
Magnox	3.7 10^{-11}	4.2 10^{-11}	1.3 10^{-10}
Aluminium	4.9 10^{-16}	5.5 10^{-16}	1.8 10^{-15}
Organics (LLW)	0.0 10^{+0}	0.0 10^{+0}	0.0 10^{+0}
Organics (ILW)	0.0 10^{+0}	0.0 10^{+0}	0.0 10^{+0}
Pucks	4.3 10^{-12}	4.9 10^{-12}	1.6 10^{-11}
Other	0.0 10^{+0}	0.0 10^{+0}	0.0 10^{+0}
Total	9.3 10^{-10}	1.1 10^{-9}	3.4 10^{-9}

Other alternative cases

Peak annual doses for the reference and all alternative cases are shown in Table 6.9. The vent not closed case results are not repeated here as the risks (probabilities) are different compared to the other cases, so the doses are not readily comparable.

The peak dose from vault wastes reduces by a factor of 10 if only LLW is considered in the assessment.

The only cases with significantly increased doses (greater than a factor of 5) compared to the reference case are the early geomembrane failure case (discussed in the previous subsection), the higher plant uptake case, and some 'what-if' cases.

In the higher plant uptake case, all plant uptake parameters are varied together, so the sensitivity to individual parameters would be much smaller. The parameterisation of the biosphere model is already cautious (e.g. the generic effective diffusivity assumed for soils in the C-14 biosphere model). Therefore, it is unlikely doses could be a factor of 20 higher than the reference case due to biosphere model parameter uncertainty.

Table 6.9: Peak annual doses for reference and alternative cases (µSv). Area-average doses across the whole cap area. Area-average doses across the vaults presented in brackets.

Case	Peak dose (µSv)	Case	Peak dose (µSv)
Reference	0.29 (0.56)	Higher steel to methane	0.29 (0.57)
Early geomembrane failure	6.5 (14)	Slower pucks release	0.34 (0.69)
No shielded modules ILW	0.18 (0.33)	Higher plant uptake	5.3 (10)
No ILW	0.06 (0.05)	'What-if' faster corrosion	0.40 (0.81)
Metal melting	0.29 (0.56)	'What-if' microbial metabolism	4.2 (9.4)
Slags	0.62 (1.3)	'What-if' flow down stacks	4.2 (9.4)
Extended institutional control	0.06 (0.10)	'What-if' microbial resource limitation	6.5 (14)
No institutional control	0.78 (1.5)	'What-if' no containers	32 (72)
Faster corrosion	0.36 (0.73)	'What-if' other to gas	27 (60)

6.6 Radon - Results

Subsection 6.6.1 defines the reference case and alternative assessment cases. Subsection 6.6.2 describes the results of the reference case at a level intended to illustrate the functioning of the model and provide reference case results. Subsection 6.6.3 describes the results of the alternative assessment cases more briefly and compares the results to the reference case. Fuller presentation of the results is given in reference [26].

6.6.1 Assessment Calculation Cases

A reference calculation case has been developed which provides an assessment of impacts from radon gas. This case uses best estimate conceptual model and parameter values but takes a cautious approach where the most realistic representation cannot be robustly quantified (Subsection 2.5.1).

The reference case is evaluated deterministically. Deterministic variant calculation cases area then used to explore the key uncertainties identified in the model and data. Table 6.5 lists the variant cases and describes the changes compared with the reference case. The uncertainties can be grouped under the headings of conceptual model uncertainty, assessment model uncertainty, parameter sensitivity and uncertainty, and 'what-if' situations.

Table 6.10: Calculation cases for radon gas assessment

Case	Description
Reference case	
Reference	Two parallel calculations: geomembrane intact (1) and geomembrane degraded (2). Risks are calculated for houses on the cap.
Conceptual model uncertainty cases	
Radial flow in the profile fill above the vaults	The profile fill above the vaults will be granular and have similar properties to the gas collection layer. As reference case (1) but with radial flow of gas in the profile fill and gas collection layer above the vaults.
Radial flow in the profile fill above the trenches	As reference case (1) but with radial flow of gas in the profile fill and gas collection layer above the trenches. This is a lower likelihood case as most of the trench profile material will have cohesive properties and therefore is expected to have low permeability compared with the gas collection layer.
Cap vent not closed	Cap mushroom vent remains open post-PoA to manage bulk gas pressures. Risks are calculated for houses on the vent.
Alternative vent designs	Alternative gas management approaches and vent designs are assessed qualitatively and semi-quantitatively.
Building design	Assumes a modern building design, with more resistive layers in the foundations and lower ventilation rate.
Diffusion only through house foundations	Bulk gas is assumed to migrate across the foundations and discharge to the atmosphere at the perimeter of the house rather than entering the house. Radon only enters the house by diffusing through the foundations.

Case	Description
Duration of institutional control	No institutional control period - doses are calculated immediately after final capping. Provides an upper bound on the sensitivity of potential impacts to the period of institutional control.
Assessment model uncertainty cases	
Localised gas spreading	As reference case (1) but bulk gas is assumed to spread laterally from the release area over the full area of the corresponding model grid cell.
Gas pressure equilibration	As above, but gas pressures and flow velocities in the gas collection layer are assumed to equilibrate over the whole area of the cap, so the bulk gas velocity is the same everywhere.
Ventilation-rate-based calculation	Use ventilation-rate-based calculations (turnover rates) to calculate indoor radon concentrations, compared with the empirical relationship used in reference case (2).
Parameter sensitivity and uncertainty cases	
Barrier functions	a) Profile thickness of trenches varied from 2 to 10 m. b) Profile thickness of vaults varied from 2 to 8 m. c) Degraded BES - has the properties of the drainage layer. d) Moisture retention layer replaced with hardcore below house. e) No bulk gas. Investigates importance of diffusion only. f) Gas spreading over nominal 1 m^2 area above defects.

Case	Description
Parameter uncertainties	a) Cap geomembrane failure time: 500, 1,000 and 1,750 y. 500 is the earliest time for complete degradation of the geomembrane. b) Rn-222 emanation fractions for trenches and vaults. c) Value of the empirical ratio of Rn-222 concentrations in indoor air to soil atmosphere. d) Effective dose for unit concentration of Rn-222 in air.
Ra-226 hotspots	Qualitative assessment of the potential impacts from small hotspots of Ra-226.
Low likelihood 'what-if' cases	
No attenuation	Mathematically possible but physically impossible bounding case assuming no attenuation in the cap or house foundations.
Cap erosion	Maximum plausible extent of cap erosion, assuming the top three layers of the cap (surface soil, moisture retention, filter sand) have eroded by the start time of the assessment. Far beyond the maximum described by the EPA [13].

6.6.2 Analysis of the Reference Case

The key features of the reference case for radon are:

- Geomembrane intact:
  - Advection-diffusion through defects (small holes) in the cap geomembrane. Bulk and radon gas migrate vertically through the profile file into the gas collection layer. Advection in the gas collection layer is then radially towards the holes in the geomembrane.
  - Use the 'local' bulk gas generation rate. Exclude lateral spreading and mixing of gas in the profile fill and gas collection layer.
  - Narrow plume from a hole through the overlying cap layers; spreads over the area of a house in the foundation layers. Advection-diffusion through foundation layers into house atmosphere.
  - Note that only some houses will be over small holes in the geomembrane, so there will only be a radon pathway to some houses.
- Geomembrane degraded, only applicable from 500 y after cap emplacement (the shortest expected timeframe to degradation of the geomembrane):
  - Advection-diffusion vertically to below the foundations of a house. Use 'local' bulk gas generation rate and ignore lateral spreading of gas.
    - Advection-diffusion through foundation layers to house atmosphere; or
    - Empirical relationship used to calculate radon gas concentration in the house from the radon gas concentration in the soil atmosphere.
- Historic house design is assumed. A variant calculation case confirms that this is likely to have similar or higher radon concentrations to a modern house design. Risks are calculated for houses on the cap and houses associated with smallholdings on the cap (the latter is relevant to calculating combined risks from radon and C-14 gas).
- Outdoor radon concentrations are calculated using a simple dispersion model.
- Decay of the Ra-226 inventory is calculated for the elapsed time since the final year of disposal to each vault or trench. The start of the assessment is taken to be the end of the PoA, 100 y after final Vault 12 capping, i.e. at 2242 AD.

Migration and flux of radon

While the geomembrane is intact, bulk gas and radon are assumed to migrate vertically through the profile fill and then radially in the gas collection layer towards a defect. The bulk gas generation rate for each of the trenches and vaults is shown in Figure 6.17 and Figure 6.18. These are proportioned volumetrically to each grid cell. The bulk gas generation is mostly constant until 1,000 y in the trenches when mild steel waste fully corrodes, and 750 in the vaults when aluminium waste fully corrodes.

Figure 6.19 shows the flux of radon through the key parts of the pathway in model grid cell 1T (Figure 3.12) for the situation when the geomembrane is intact (via a defect) or degraded.

While the geomembrane is intact, the flux through a single defect is around three orders of magnitude less than the flux from the waste directly. This is in part due to the attenuation provided by the profile fill above the waste, the additional path-length to travel from the release point laterally to the defect, and because it is assumed there is a defect every 400 m^2, so only radon within that area is modelled as migrating through the defect, with the rest migrating through other defects. However, the fluxes out of the top of cap (moisture retention layer) immediately below the foundations of the house are the same as through the defect. This is a consequence of the limited lateral spreading assumed through these layers and high advective velocity.

When the geomembrane is degraded, the flux out of the top of the cap is around 875 times less than the flux from the waste, accounting for the attenuation provided by all the cap layers and profile fill. Note that here the flux through the 'foundations' layer represents the whole flux across the cell, not accounting for the fact that the house foundations would only intercept a small proportion of this flux; the figure is simply intended to demonstrate the attenuation provided by each layer. The flux intercepted by the footprint of a house would therefore be another factor of ~30 lower (making it lower than the flux intercepted by a house over a defect in the geomembrane intact case).

The attenuation provided by the house foundation layers results in ~14% and ~15% reduction in flux into the house when the geomembrane is intact and degraded, respectively. This difference is again due to the bulk gas velocity being greater when the geomembrane is intact as it is not assumed to spread until it reaches the foundations.

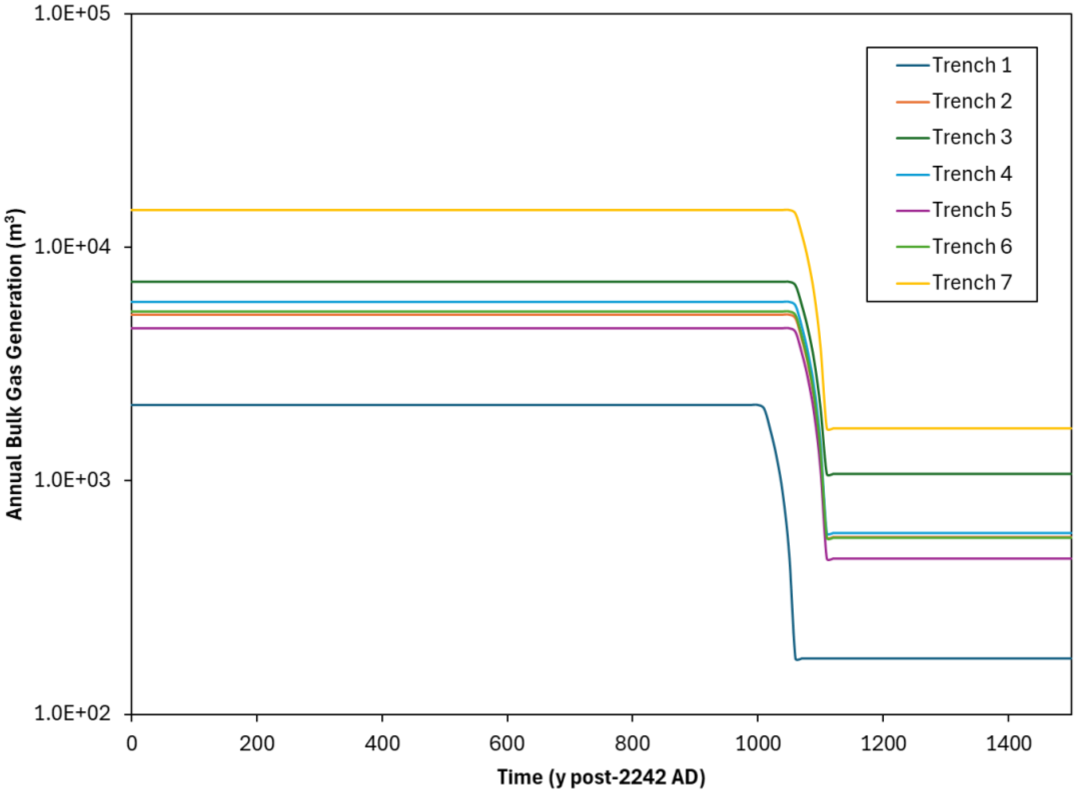

Figure 6.17: Annual bulk gas generation from the trenches

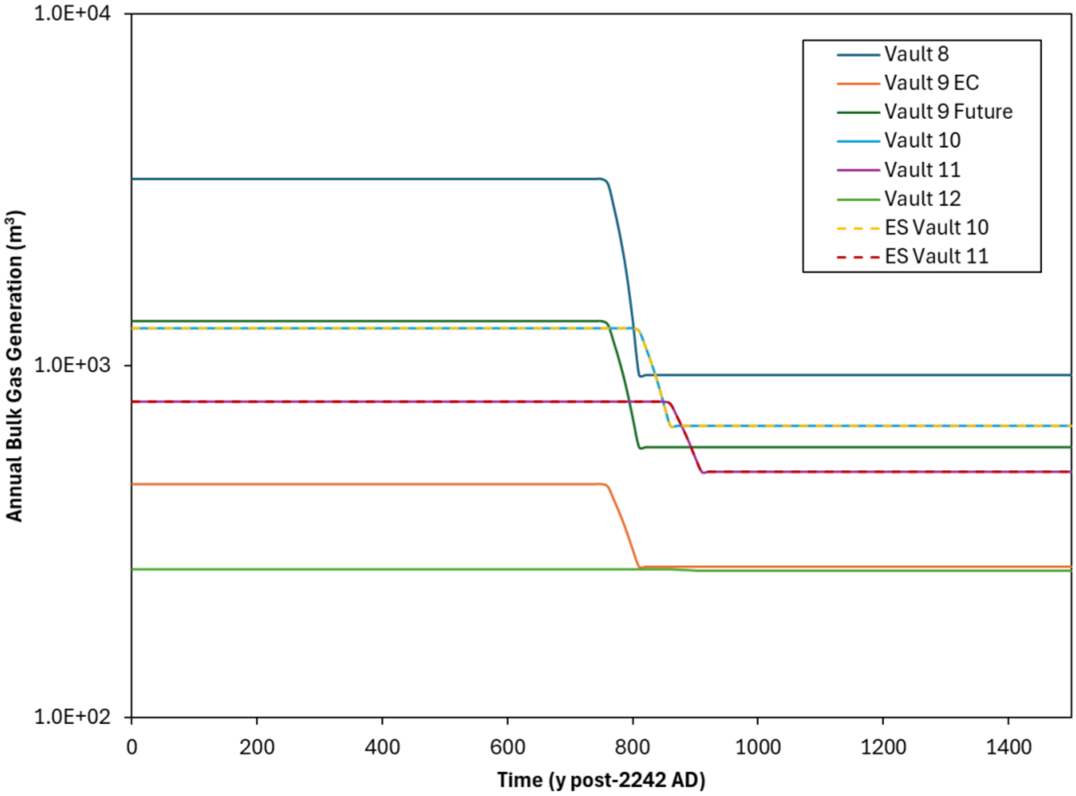

Figure 6.18: Annual bulk gas generation from the vaults

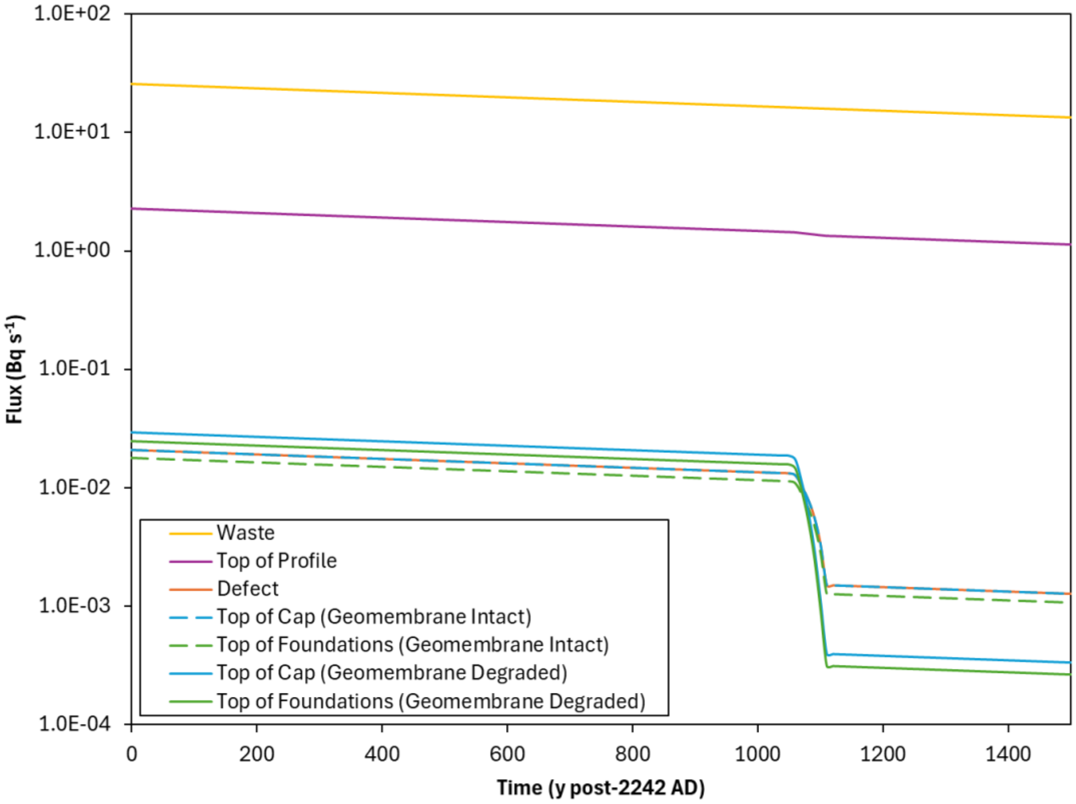

Figure 6.19: Flux of radon (Bq s^{-1}) through key parts of the pathway for trench cell 1T when the geomembrane is intact (through a single defect) or degraded (across the whole cell). Reference case.

Annual effective doses and risks

For the geomembrane intact, doses are calculated for a person in a house above a defect within each model grid cell. An area weighting is used to derive the average dose across the cap area above the vaults or trenches (accounting for houses being above or not above a defect), based on the area of the house, the area of the cap, and the number of defects per cell. For the geomembrane degraded, area-averaged doses are calculated for a person in a house which could be located anywhere on the cap and would intercept a proportion of the flux from that cell. Figure 6.20 and Figure 6.21 show the area-averaged dose across the trenches and vaults, respectively. Table 6.11 summarises the peak area-averaged effective doses as well as the peaks in the worst trench and vault cells.

For both the trenches and vaults, the peak doses are at the earliest times. Doses follow the trend of the bulk gas generation rates (Figure 6.17 and Figure 6.18). At the point the geomembrane fails, the dose would change from the geomembrane intact curves to the degraded geomembrane curves. Doses are higher with the intact geomembrane because bulk gas and radon are focused through defects, leading to higher bulk gas velocities.

The cells which give the highest doses (10V and 6T) are those with high Ra-226 inventories, thin profile thickness, and/or high bulk gas generation rates per unit area (leading to higher advective velocities).

For the situation where the cap geomembrane has degraded, the empirical model and analytical model give relatively similar doses, with the empirical model resulting in doses around a factor of 1.3 to 1.4 higher than the analytical model. There are uncertainties associated with both approaches, described in reference [26]. The range of uncertainty for each approach is greater than the factor of 1.3 to 1.4 difference in the best estimate results, so it is not possible to conclude whether there is greater confidence in the analytical model or the empirical model. Therefore, both are presented for all cases.

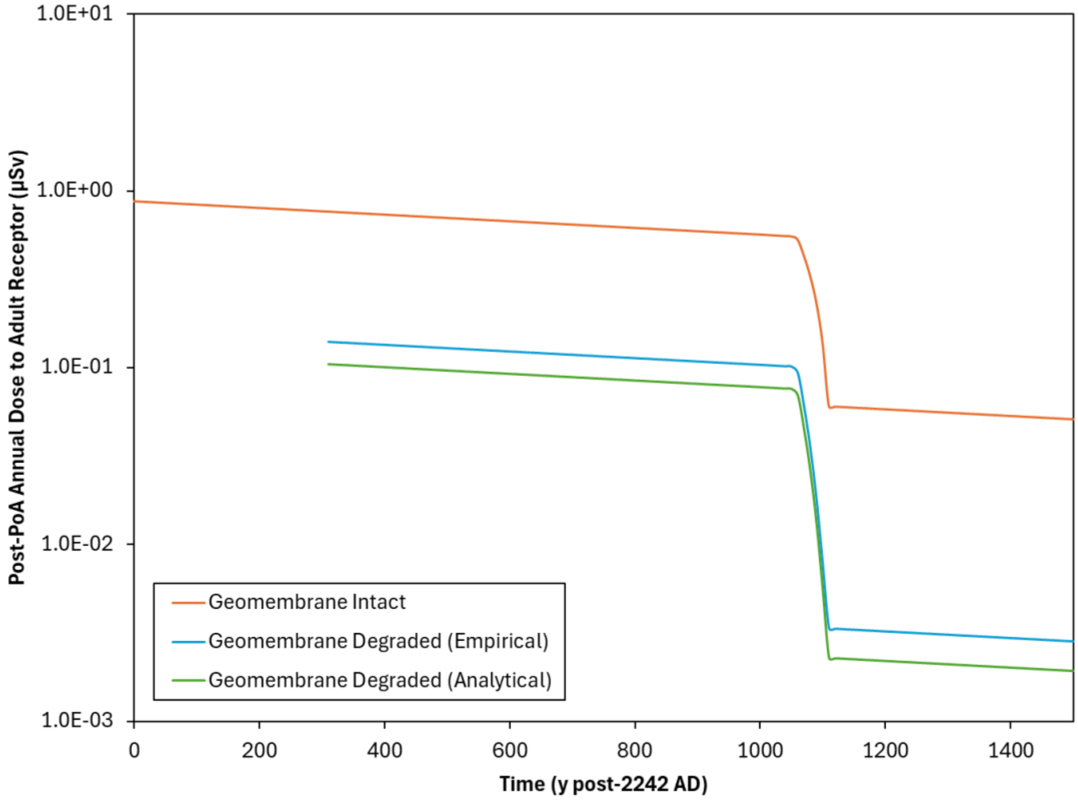

Figure 6.20: Calculated area-averaged annual effective doses from all trenches when the geomembrane is intact and degraded. Reference case.

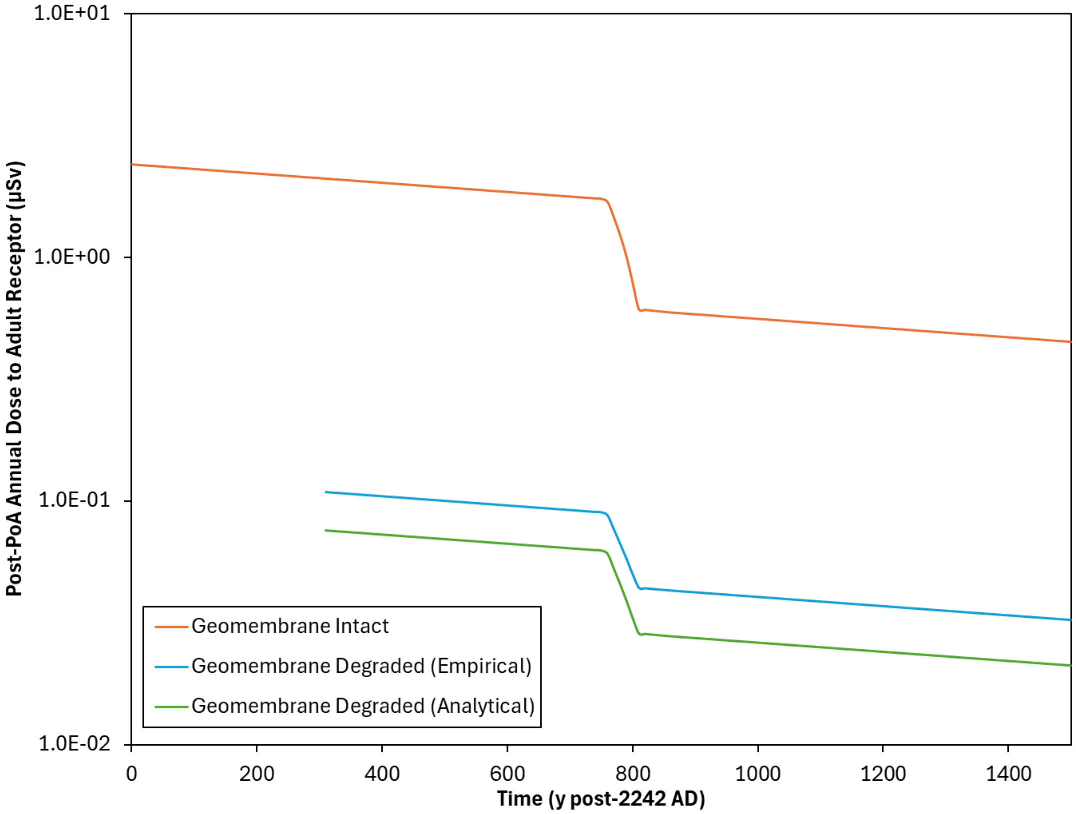

Figure 6.21: Calculated area-averaged annual effective doses from all vaults when the geomembrane is intact and degraded. Reference case.

Table 6.11: **Peak annual effective dose from trench and vault cells, area-averaged and for the worst cells. Reference case.**

	Geomembrane intact	Geomembrane degraded - empirical	Geomembrane degraded - analytical
Trenches			
Area-averaged peak over trenches (μSv)	0.9	0.1	0.1
Cell peak (μSv)	69 [Cell 6T]	1.9 [Cell 4T]	1.4 [Cell 4T]
Time of peak (y)	0	294	294
Vaults			
Area-averaged peak over vaults (μSv)	2.4	0.11	0.08
Cell peak (μSv)	60 [Cell 10V]	0.5 [Cell 10V]	0.4 [Cell 10V]
Time of peak (y)	0	294	294

Area weighted average doses for the whole repository are shown in Figure 3.12. Most of the potential dose is from existing disposals to the trenches. Thorite sands disposed in Trench 2 and Trench 3[15] are the main sources of Ra-226 in the trenches.

Risks from the repository can be calculated from these doses, taking account of the probability of a house being built on the cap, 0.83 (calculated from the spatial density data of all settlements on the West Cumbrian coast, Table 6.4). The peak risk (7.4 10^{-8} y^{-1}) occurs with the geomembrane intact (Table 6.12). This is around a factor of 14 below the risk guidance level.

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[15] Disposals from Thorium Ltd described as 'filter press cake' but interpreted to comprise thorite sand [63].

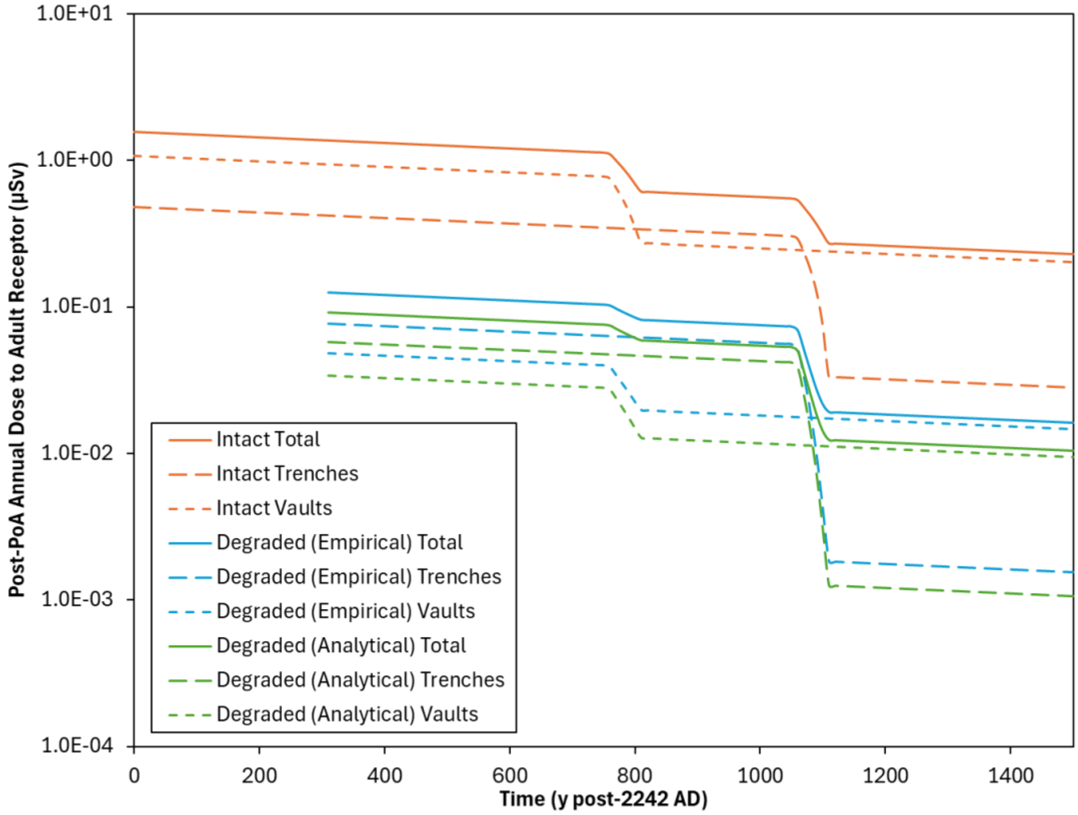

Figure 6.22: Calculated area-averaged annual effective doses from all trenches when the geomembrane is intact and degraded. Reference case.

Table 6.12: Peak calculated risk area-averaged over the whole cap. Reference case.

	Geomembrane intact	Geomembrane degraded - empirical	Geomembrane degraded - analytical
Area-averaged peak risk (y^{-1})	7.4 10^{-8}	6.0 10^{-9}	4.3 10^{-9}
Time of peak (y)	0	294	294

Doses are also calculated to an exposed person either spending time outside a house or walking across or near to the site. Peak doses are three to four orders of magnitude lower than indoors, over the worst cells. Doses at offsite locations are lower than the peak doses outside on the repository cap.

6.6.3 Alternative Cases

Results for the alternative cases, as identified in Table 6.10, are more fully presented in reference [26]. Here we present results from the cap vent open case and summarise results for the other cases.

Cap vent not closed case

As described in Subsection 6.5.3 for the C-14 gas assessment, it is uncertain whether a cap vent (or vents) will need to be left open post-PoA to manage bulk gas pressures. In this variant case, a cap mushroom vent (Figure 6.16) remains open post-PoA. Bulk and radon gas migrate vertically through the profile fill via advection and diffusion into the gas collection layer. Bulk gas advection in the gas collection layer is then modelled radially towards the vent, and released directly to atmosphere, bypassing the overlying cap layers.

Doses are calculated to a person inside a house located at the crest of the cap above the vent. The house has a footprint of 75 m^2 and the vent has area of 443 m^2. Due to the shape of the vent, the maximum area of the house footprint which can sit directly above the vent release area is 38 m^2 (Figure 6.23). This intersection area determines the proportion of flux released from the vent which enters the house.

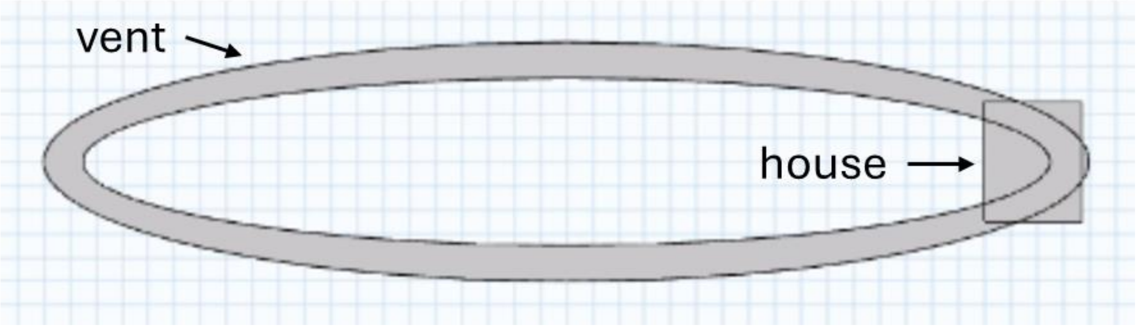

Figure 6.23: Vent and house geometry with maximum area of intersection [26]

The peak annual effective dose reaches 1.2 μSv, immediately after the end of the PoA (Figure 6.24). This is highly dependent on the assumptions that the house location maximises intersection with the vent, and that the entire flux of bulk and radon gas below the footprint of the house enters the house. Outdoor doses are more than four orders of magnitude lower than indoor doses.

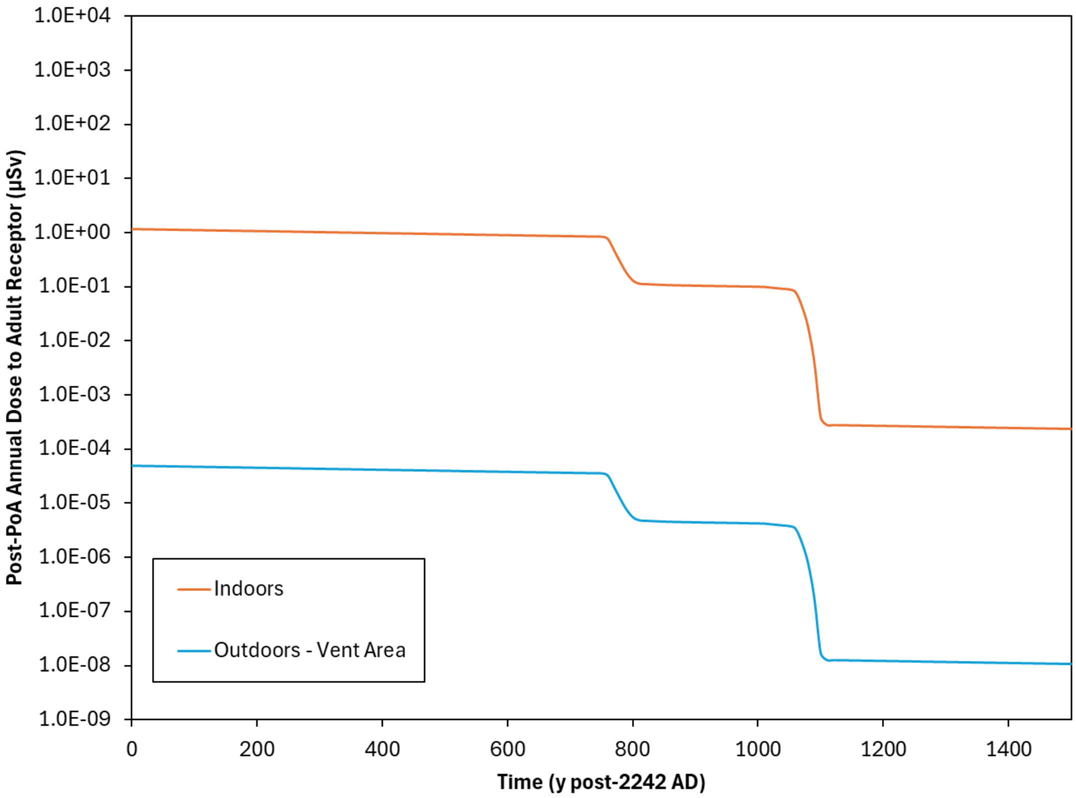

Figure 6.24: Calculated annual effective dose for the Cap Vent Not Closed case

The probability of a house on the vent is much lower than the probability of a house somewhere on the cap. Figure 6.25 shows the calculated risk associated with the vent open case, taking into account the likelihood of a house intersecting the area of the vent (0.0067). This probability assumes that a house is equally likely to be built on all areas of the cap.

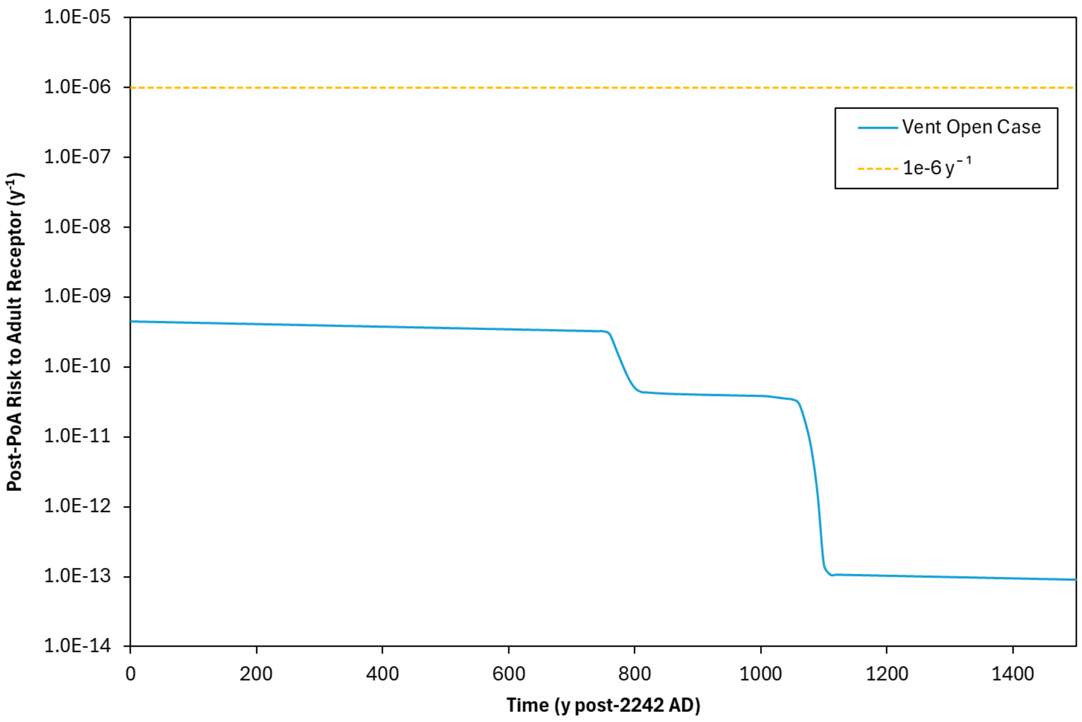

Figure 6.25: Calculated risks from the repository. Cap Vent Not Closed case.

The maximum capture zone (the radius from which radon contributes to the flux out of the vent) is approximately 60 m away from the vent in the direction of the vaults and 120 m away from the vent in the direction of the trenches. Any radon from further than this distance will have significantly decayed before it reaches the vent, providing negligible contribution to the flux out of the vent. The difference between the vaults and trenches is because bulk gas generation from the trenches is greater than from the vaults. The assessment model results show that Ra-226 disposed in the shielded modules in Vault 10 is the main source of radon discharging from the vent due to its proximity to the vent.

Other alternative cases

Peak annual doses for the reference and all alternative cases are shown in Table 6.13 for the intact geomembrane and Table 6.14 for the degraded geomembrane.

The results show that the most significant uncertainties are associated with the radial flow assumptions in the profile fill over the trenches. The peak doses for the trenches are much more sensitive to the assumed flow path than for the vaults, because the thickness of profile material is typically much greater over the trenches than over the vaults. Radial flow reduces the pathlength to a defect, and therefore the travel time and time for radon decay, compared with the reference case. It is unlikely that there would be radial flow in the trench profile fill as most of the trench profile material will have cohesive properties and is therefore expected to have low permeability compared with the gas collection layer. Therefore, the most efficient flow path (greatest pressure drop per unit length, i.e. greatest hydraulic gradient) will be directly across the profile, as assumed in the reference case.

A qualitative variant case not shown in Table 6.14 considered the impact of Ra-226 'hotspots' in the existing disposals to the trenches and Vault 8. The assessment model grid has been discretised to capture the Ra-226 hotspot in the trenches (thorite sands in Trenches 2 and 3), so the assessment calculations give a good estimate of potential radon doses from these. Ra-226 hotspots in Vault 8 are smaller and are likely to include Ra-226 sealed sources. The emanation fraction from sealed sources is likely to be lower than from other containerised wastes in the vault, but no account has been taken of this in the assessment. It is unlikely that small hotspots in the vaults would lead to higher than calculated doses, because of the spreading of radon from the hotspot as it migrates through the cap.

The presence of small hotspots should not significantly change the calculated risks from the facility, as the risk depends on the area weighted average dose across the facility. A calculation of radon risks from hotspots of Ra-226 in the facility would need to include the low probability of a house, or houses, intersecting gas from hotspots. This probability cannot be determined but is anticipated to be small compared with the probability of at least one house somewhere on the cap. Therefore, the risks from Ra-226 hotspots in the facility are expected to be similar to or lower than risks from all the Ra-226 in the facility.

Table 6.13: Peak annual effective doses for reference and alternative cases (μSv). Intact geomembrane.

Case	Area-averaged peak over trenches (μSv)	Area-averaged peak over vaults (μSv)	Peaks for highest trench and vault cells (μSv)
Reference	0.9	2.4	69 - Cell 6T 60 - Cell 10V
Radial flow in profile fill	52	6.2	4,800 - Cell 39T 130 - Cell 2V
Diffusion only through house foundations	0.7	1.9	53 - Cell 6T 47 - Cell 10V
No institutional control	0.9	2.5	72 - Cell 6T 63 - Cell 10V
Localised gas spreading	0.7	2.3	29 - Cell 4T 58 - Cell 2V
Gas pressure equilibration	0.4	6.0	32 - Cell 6T 132 - Cell 10V

Moisture retention layer replaced	0.7	1.9	55 - Cell 6T 48 - Cell 10V
No bulk gas	0.001	0.02	0.1 - Cell 6T 0.5 - Cell 10V
Gas spreading above defect	0.7	1.2	58 - Cell 6T 32 - Cell 2V
Emanation fraction - minimum	0.2	1.0	14 - Cell 6T 26 - Cell 10V
Emanation fraction - maximum	4.8	14	380 - Cell 6T 340 - Cell 10V
Radon dosimetry - minimum	0.7	1.9	55 - Cell 6T 48 - Cell 10V
Radon dosimetry - maximum	1.4	3.8	110 - Cell 6T 96 - Cell 10V

Table 6.14: Peak annual effective doses for reference and alternative cases (μSv). Degraded geomembrane. Empirical / analytical results.

Case	Area-averaged peak over trenches (μSv)	Area-averaged peak over vaults (μSv)	Cell peaks (μSv)
Reference	0.14 / 0.11	0.11 / 0.08	1.9 / 1.4 - Cell 4T 0.5 / 0.4 - Cell 10V
Diffusion only through house foundations*	0.08	0.06	1.1 - Cell 4T 0.3 - Cell 10V
Localised gas spreading	0.09 / 0.07	0.10 / 0.07	0.9 / 0.7 - Cell 5T 0.5 / 0.4 - Cell 2V
Degraded BES	1.0 / 0.7	11.2 / 7.8	15 / 11 - Cell 6T 47 / 33 - Cell 10V
Moisture retention layer replaced*	0.26	0.26	3.4 - Cell 4T 1.2 - Cell 10V

Case	Area-averaged peak over trenches (μSv)	Area-averaged peak over vaults (μSv)	Cell peaks (μSv)
No bulk gas	0.001 / 0.0008	0.02 / 0.014	0.02 / 0.01 - Cell 6T 0.10 / 0.06 - Cell 10V
Emanation fraction - minimum	0.03 / 0.02	0.05 / 0.03	0.4 / 0.3 - Cell 4T 0.2 / 0.2 - Cell 10V
Emanation fraction - maximum	0.78 / 0.58	0.62 / 0.44	11 / 7.9 - Cell 4T 2.9 / 2.0 - Cell 10V
Empirical ratio - minimum⁺	0.10	0.08	1.4 - Cell 4T 0.4 - Cell 10V
Empirical ratio - maximum⁺	0.15	0.12	2.0 - Cell 4T 0.5 - Cell 10V
Radon dosimetry - minimum	0.11 / 0.08	0.09 / 0.06	1.5 / 1.1 - Cell 4T 0.4 / 0.3 - Cell 10V
Radon dosimetry - maximum	0.22 / 0.17	0.17 / 0.12	3.0 / 2.3 - Cell 4T 0.8 / 0.6 - Cell 10V
'What-if' no attenuation	140 / 210	260 / 400	1,700 / 2,700 - Cell 39T 870 / 1,300 - Cell 2V
'What-if' cap erosion	0.62 / 0.37	0.69 / 0.38	8.2 / 4.9 - Cell 4T 3.1 / 1.8 - Cell 10V

*Only analytical results relevant to this case.

⁺Only empirical results relevant to this case.

6.7 Results - Total Gas Pathway Risks

Smallholders living on the cap could be exposed to radon and C-14 gas. The combined risks from radon and C-14 gas are calculated for the:

- Reference case, with the vent closed and the cap geomembrane intact.
- Early geomembrane failure case, which assumes the cap geomembrane fails 1,000 years after Vault 8 capping (radon) or 1,000 years after placement for each strip of the cap (C-14).
- Cap vent is not closed case.

6.7.1 Reference Case

The peak combined risk for the reference case is 7.9 10^{-8} y^{-1} and is dominated by radon (Figure 6.26). The key source of radon is Ra-226 associated with thorite sands in trenches 2 and 3.

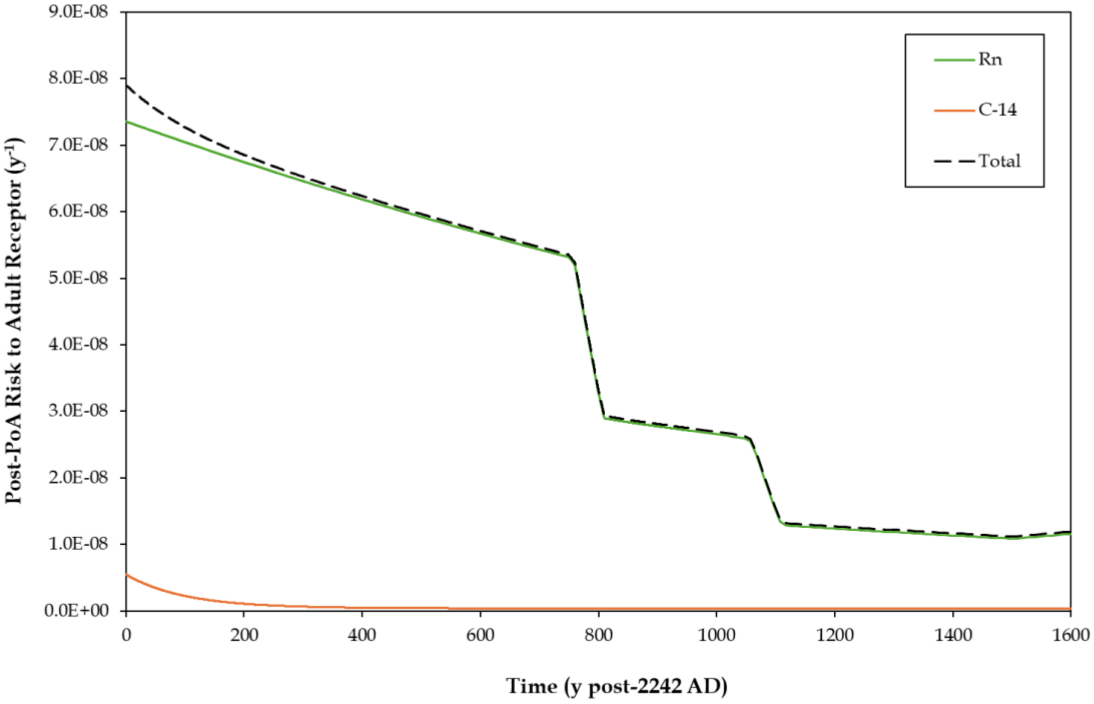

Figure 6.26: Total risk from the repository. Reference case.

6.7.2 Early Geomembrane Failure Case

The peak combined risk for the early geomembrane failure case is 1.3 10^{-7} y^{-1} with significant contributions from radon and C-14 (Figure 6.27). The peak risk is at the time of geomembrane failure and is higher than the peak risk at the end of the PoA with the geomembrane intact.

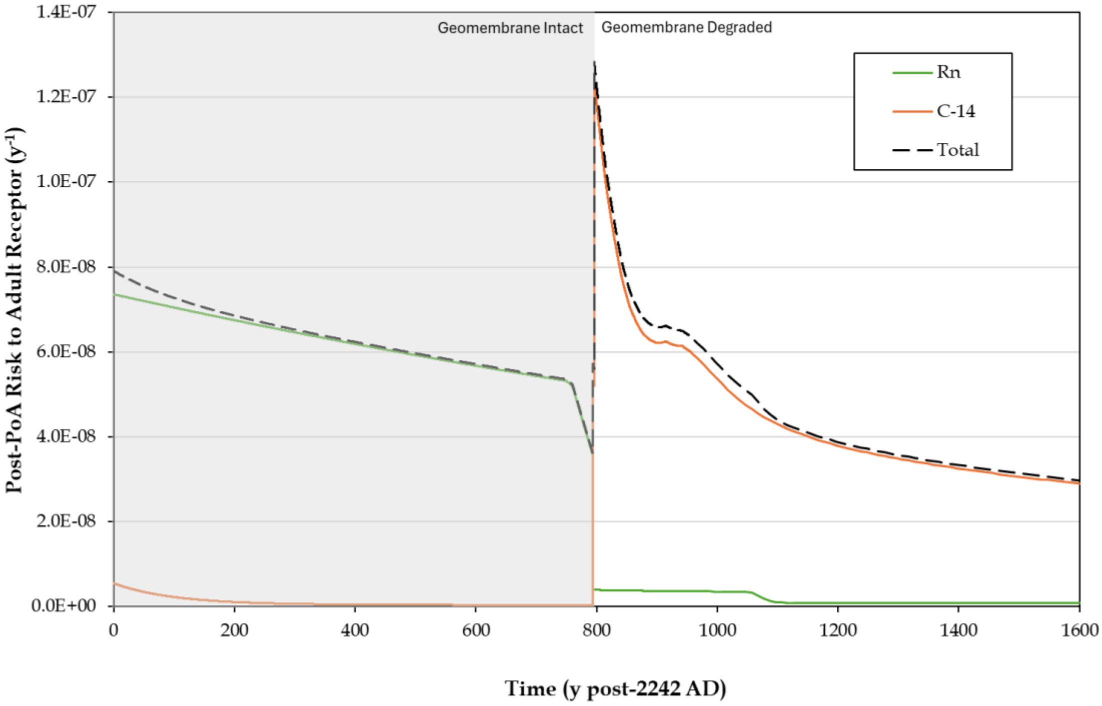

Figure 6.27: Total risk from the repository. Early geomembrane failure case.

6.7.3 Cap Vent Not Closed Case

The peak combined risk for the cap vent is not closed case is 1.4 10^{-9} y^{-1} with significant contributions from radon and C-14 (Figure 6.28). The key source of radon is Ra-226 associated with future disposals in the Vault 10 shielded modules, directly below the vent.

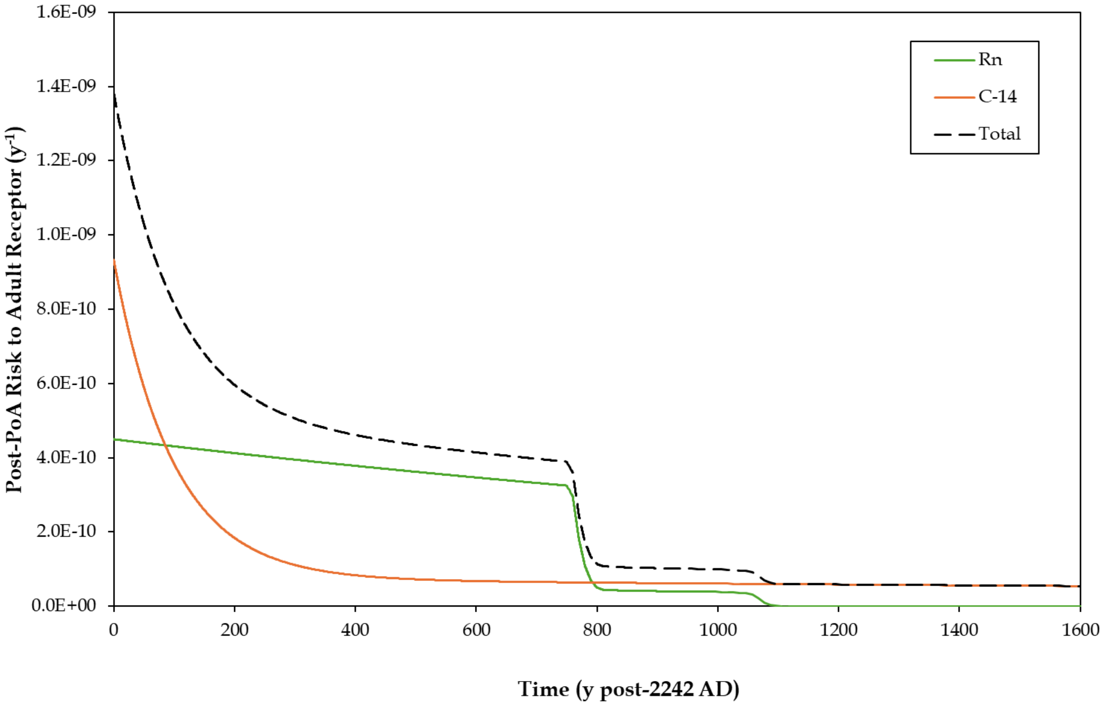

Figure 6.28: Total risk from the repository. Cap vent is not closed case.

6.8 Summary of Results and Assessment

The gases of interest are:

- C-14-bearing methane and carbon dioxide, generated from C-14-bearing wastes;
- Rn-222 (radon) generated from Ra-226 bearing wastes.

C-14

The reference case assumes the cap vent is closed at the end of the PoA. With the geomembrane intact, infiltration is too low for any significant release of C-14 in solution from containers. The pH in the containers is expected to be too high for significant microbial activity. Therefore, with the cap geomembrane intact, C-14-bearing small organic molecules released from the waste are retained in the containers and are not metabolised to C-14 gas. This limits the amount of C-14-bearing gas generated and released. The peak risk is 5.5 10^{-9} y^{-1}. This is more than two orders of magnitude below the risk guidance level.

The most significant uncertainties are associated with:

- The cap geomembrane failure time.
- The rate of transition in conditions as the cap geomembrane fails.

- The long-term performance of future stronger LLW containers and new containers for ILW.
- The impact of surcharging on the barrier performance of containers of existing designs.

The geomembrane is more likely to be intact and functioning well than failed when the repository starts to be disrupted by coastal erosion [13]. There is a strong probability the cap geomembrane would still be intact when the repository starts to be disrupted by coastal erosion (more than 50% based on the relevant timescales assumed for assessment calculations). However, there is also a significant probability that the cap geomembrane would fail before the repository starts to be disrupted by coastal erosion. With earlier than expected degradation of the cap geomembrane, at 1,000 years, the calculated peak risk increases to $1.2\ 10^{-7}\ y^{-1}$.

With early failure of the cap geomembrane, there is sufficient infiltration for C-14-bearing small organic molecules to be transported out of the containers by diffusion (future stronger containers and new containers for ILW) and leaching (existing container designs), into the gaps between container stacks and the bases of the vaults where the pH is lower. There, microbes are expected to be active and metabolise the small organic molecules forming C-14-bearing gas. Small organic molecules are expected to 'build up' in the container porewater while the cap geomembrane is intact. Once the geomembrane fails, there is a peak in releases of small organic molecules from the containers, leading to a peak in generation and releases of C-14 gas. The size and shape of the peak depends on the rate of release of C-14 in solution from containers, which is controlled by the rate of transition in conditions as the cap geomembrane fails, and the performance of the different container types in providing a barrier to release.

Irradiated graphite wastes are the main post-PoA sources of C-14 gas from LLW and ILW. In the 2013 assessment, irradiated graphite and irradiated steel wastes were the main sources of C-14 gas. The C-14 gas fluxes per unit inventory have decreased significantly from these waste types compared with the 2013 assessment. This is due to slower steel corrosion rates, improved understanding of the form of release from steels, and improved understanding of the availability and rate of release from graphite. The slower corrosion rates reflect updates to the near-field conceptual model (pH in the containers) and significantly updated information on steel corrosion rates under different conditions. Improved understanding of the form of C-14 release from irradiated steels comes from the EC CAST project. Improved understanding of the availability, rate and form of C-14 release from graphite comes from experiments on reactor core graphite, and reviews undertaken in the UK and internationally over the last decade.

If the cap vent is left open at the end of the PoA, releases of C-14 gas are focussed through the vent. Potential doses to occupiers of smallholdings on the vent are higher than in the reference case, which considers smallholdings anywhere on the cap. However, because the probability of at least one smallholding on the vent is lower than the probability of at least one smallholding on the cap, the risk from C-14 is lower for the open vent case than the reference case. The risks for the open vent case depend on the layout of the smallholding over the vent, i.e. whether the house and kitchen garden are located directly above the vent or away from the vent. Peak risks vary between 9.3 10^{-10} y^{-1} and 3.4 10^{-9} y^{-1} depending on the layout.

If there is no disposal of ILW, peak doses from the vaults are reduced by a factor of 10 compared with the reference case. Peak doses and risks across the cap (including contributions from vaults and trenches) are reduced by a factor of 5.

Overall, the results provide confidence that with appropriate WAC, and control of use of the radiological capacity [17], the potential post-PoA risks from C-14 gas can be limited to the levels described in the GRA.

Radon

The cap geomembrane is expected to have a lifetime of hundreds of years to a few thousand years. While the cap geomembrane is intact it will be a barrier to release of bulk and radon gas. With the cap vent(s) closed, a small gas overpressure is expected to develop in the repository, with bulk gas flowing through defects (small holes) in the cap geomembrane to discharge at the surface of the cap. The proportions of bulk gas, and associated radon, that discharge at the cap surface compared with the perimeter of the cap are uncertain. The reference case cautiously assumes all discharge is to the surface of the cap. This gives the shortest travel time and least decay of radon, for most waste, and therefore higher risks than discharge at the perimeter of the cap.

With the cap vent(s) closed and the geomembrane intact, the peak reference case radon risk to houses developed on the cap is 7.4 10^{-8} y^{-1}. This is around a factor of 14 below the risk guidance level.

The most significant uncertainties are associated with:

- the number of defects (small holes) in the cap geomembrane;
- the cap geomembrane failure time;
- requirements for a passive cap vent (or vents) to remain open in the post-PoA period;
- bulk gas generation rates;
- the gas flow and travel times; and
- radon emanation from the waste.

If the cap geomembrane fails before the repository starts to be disrupted by coastal erosion, then bulk gas velocities will decrease leading to slower transport of radon by bulk gas, greater decay of radon, and lower risks. Risks are around a factor of ten to twenty lower compared with when the geomembrane is intact. However, advection is still a significant process. If advection is removed from the assessment model, such that transport of radon is only by diffusion, then risks from radon are negligible, consistent with the assumptions and results of the 2011 ESC [89].

The case where the cap vent is left open and a house, or houses, are built on the vent results in doses comparable with the reference case. However, assuming there is no preference for constructing houses on certain areas of the cap, the risks from this case ($4.5\ 10^{-10}\ y^{-1}$) are two orders of magnitude lower than from the reference case. While there is high probability there would be at least one house on the cap, the probability of there being at least one house on the vent is considerably lower.

The risks from radon are primarily associated with historic disposals in the trenches and Vaults 8 and 9. The expected inventory of Ra-226 associated with future disposals of ILW is very low, so disposal of ILW has no significant impact on the assessment results. The exception is the case where the cap vent is left open, due to the location of the Vault 10 shielded modules directly under the vent.

The results provide confidence that with appropriate WAC, and control of use of the radiological capacity [17], the potential post-PoA risks from radon gas can be limited to the levels described in the GRA.

Combined impacts from radon and C-14 gas

People living in smallholdings on the cap may be exposed to radon gas in the house and C-14 from consumption of produce from the smallholding. The peak risks from radon and C-14 gas do not occur at the same time in all cases. Therefore, the risks are added for each future time and the combined peak found. The peak risks for the Stage 2 Reference Inventory are given in Table 6.15. They are all below the risk guidance level.

Table 6.15: Combined risks from radon and C-14 gas

	Vent closed, geomembrane intact	Vent closed, geomembrane degraded	Vent open
Risk y^{-1}	$7.9\ 10^{-8}$	$1.3\ 10^{-7}$	$1.4\ 10^{-9}$
Radon contribution	93%	3%	33%
C-14 contribution	7%	97%	67%

7 Assessment of Coastal Erosion

This section presents the background, approach to assessment, models and data, case selection, and assessment calculation results for coastal erosion. A complete description of these elements is presented in reference [25]. The final subsection provides an assessment of the results against the regulatory guidance.

This section does not describe the doses from scavenging waste exposed by coastal erosion. Doses from scavenging are described in Section 8, together with doses from inadvertent human intrusion into the repository. Although scavenging is not an intrusion event, human intrusion and scavenging are grouped together as they are both assessed against the dose guidance level (Subsection 2.13.4.2).

7.1 Assessment Scope

The LLWR is expected to be completely disrupted by coastal erosion (except potentially under high rates and amounts of sea-level rise, when the repository may be inundated before erosion is complete). Disruption of the repository is expected to begin well beyond the end of the PoA, on a timescale of several hundred to a few thousand years after present, with erosion of the repository being complete within one to a few thousand years after present [8]. It is possible that people may choose to build and maintain coastal defences along the West Cumbrian coast, but there is no reliance in this assessment on maintenance of existing defences or construction of new defences.

The scope is to assess the potential risks from disruption of the repository by coastal erosion, consistent with Requirement R6 of the GRA [19]. The period of interest extends from first breach of the repository by coastal erosion, to complete erosion of the repository, and then the long-term environmental impacts from the residual activity in the coastal and marine environments.

People are expected to occupy the whole of the eroding repository frontage and wider coastal environment. They may be exposed to radionuclides in the coastal and marine environments while undertaking recreational and occupational activities, and from consuming marine foodstuffs. The probability of people undertaking these activities is assumed to be one.

7.2 Previous Assessments and Environment Agency Comments

The 2011 ESC [100] included an assessment of the potential impacts of coastal erosion of the LLWR. The approach was guided by the outcomes of the Environment Agency's review of the 2008 performance assessment update [101].

The main assessment model provided a spatial representation of the repository, coastal and marine environments. The model calculated radionuclide fluxes and environmental radionuclide concentrations over time, as the repository erodes, and radionuclides are transported through the coastal system to the sediment sink in the marine environment.

Potential doses from occupancy of the eroding site frontage, the wider coastline, and consumption of marine foodstuffs were calculated. The potential impacts from recreational use of the coastline in front of the eroding repository, and high-rate consumption of marine foodstuffs, were used to set radionuclide capacities for the vaults.

The potential impacts from scavenging, seeking out, interacting with and collecting materials exposed by coastal erosion were assessed in the 2011 ESC as an inadvertent Human Intrusion event [102]. Reference [90] notes that the LLW consignment activity limits of 4 GBq t^{-1} alpha and 12 GBq t^{-1} beta-gamma would provide an effective constraint on levels of exposure from informal beach scavenging for most consignments.

The 2011 ESC assessed the potential doses from casual encounters with sealed sources exposed by coastal erosion [103]. Additional work was undertaken in 2013 to assess the potential impacts from casual encounters with discrete items [60], and any active particles released during erosion of the repository [104].

The Environment Agency's review of the 2011 ESC identified FIs that needed to be addressed and made recommendations for potential improvements [29]. We have taken these recommendations into account when developing the coastal erosion assessment. A full description of how they have been considered is provided in the *'Addressing Regulatory Requirements and Guidance'* report [18].

7.3 Approach in this Assessment

The model for the 2026 ESC builds on the model developed for the 2011 ESC. Since the 2011 ESC, there have been significant updates to understanding of climate change and sea-level rise as described in reference [8]. Updated sea-level rise projections have been used to develop updated projections of the timing and mode of coastal erosion of the repository. Understanding of the degradation, break-up and erosion of the wastes in the coastal environment has been substantially improved as described in reference [8]. This includes significant use of a range of analogues to illustrate and improve the conceptual model.

Potential environmental radionuclide concentrations are assessed considering spatial variability in the repository inventory, coastal and marine environments, and temporal variability in the material being eroded and its transport and fate in the coastal and marine environments.

People are expected to be exposed to radionuclides in the coastal and marine environments while undertaking recreational and occupational activities, and from consuming marine foodstuffs. For these ‘expected’ activities, the probability of exposure is one. As people are expected to be exposed, calculated doses are compared with an annual dose of 20 µSv, which is equivalent to the risk guidance level assuming exposure occurs (Subsection 2.13.3). Over an annual period, it is reasonable to assume people occupying the coast would be exposed to radionuclides present in the cliffs, storm beach and foreshore over the whole area in front of the eroding repository, and radionuclides transported further along the coast.

The type of activities leading to exposure to active particles, sealed sources, discrete items are summarised in Table 2.1. People using the coast for recreational and occupational activities are expected to have casual encounters with discrete items and potentially also active particles and sealed sources, so these encounters are considered within the coastal erosion assessment. However, they are modelled separately to the main coastal erosion assessment model. This section focuses on the main coastal erosion assessment; assessment of particles, sealed sources and discrete items is summarised separately under Subsection 10.4. Scavenging discrete items and taking items home is also discussed in Subsection 10.4, but these activities are assessed against the dose guidance level rather than the risk guidance level, see Table 2.1.

7.4 Assessment Model and Data

7.4.1 Conceptual Model

Material erosion and dispersion

As described in [48], it is anticipated that the coast will maintain its current form and swash alignment as it erodes. The hinterland and then the repository will erode due to the erosion of the platform and cliffs by waves, offshore transport of sand, silt and clay, and the slow attrition of gravels and cobbles.

The dispersion of eroded material is considered in terms of conservation of volume since the Quaternary geology, the wastes, storm beach gravels and foreshore sand all have similar density. This provides a simple and robust physical basis for the assessment model.

The conceptual model resulting from these considerations is illustrated in Figure 7.1.

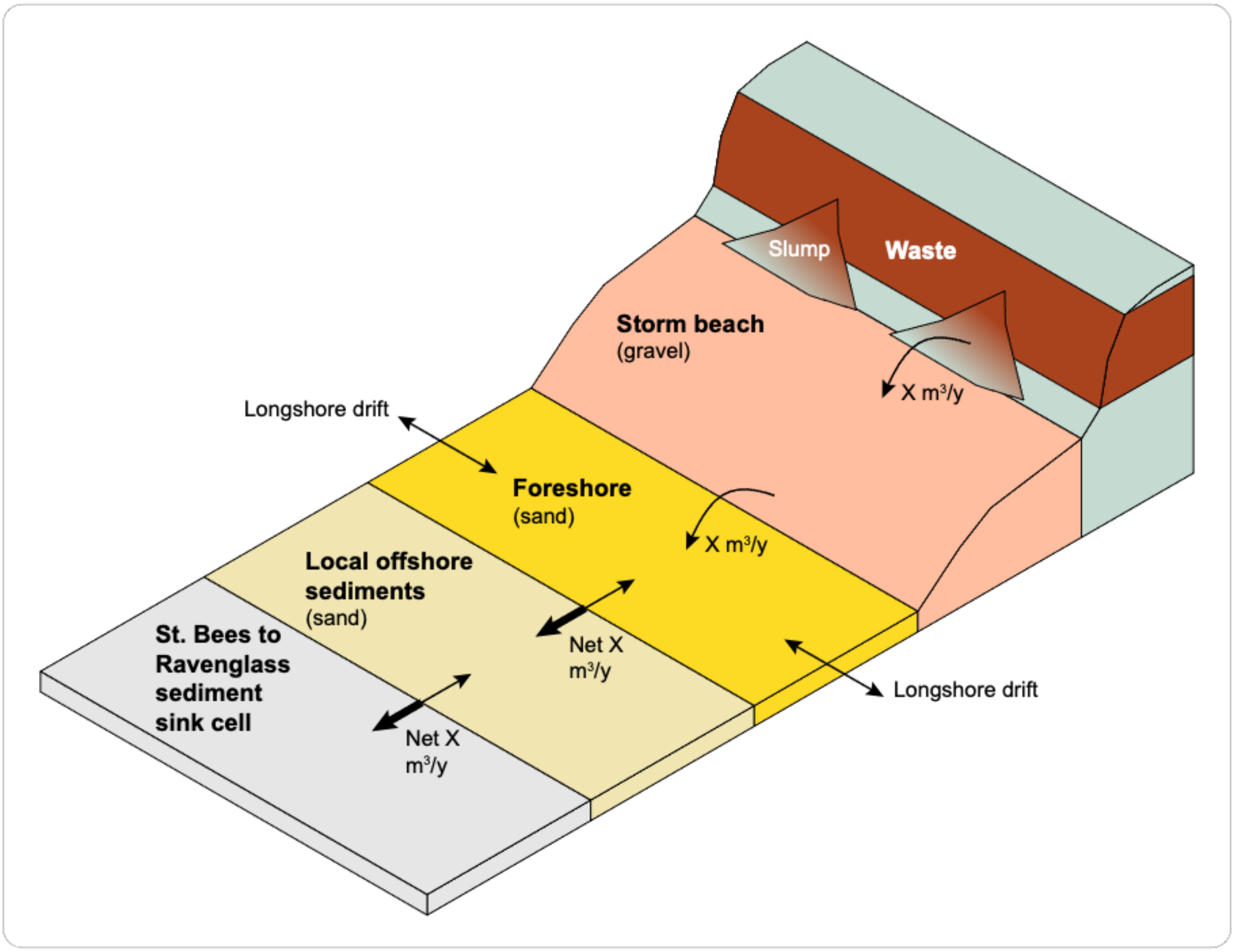

Figure 7.1: Stylised conceptual model of the movement of eroded materials during coastal erosion [25]

Table 7.1 summarises the modes of erosion of the repository relevant to each of the different emissions scenarios, as described in reference [8] [48]. There is always expected to be a small talus at the toe of the cliffs when the trenches are eroding When the vaults are eroding, higher rates of erosion are anticipated to result in a larger talus at the base of the cliff and vice versa (Figure 7.2). In the reference case, a large talus is assumed to form - a variant model assuming a small talus is used to bound the uncertainty.

Table 7.1: Projections of the mode of erosion [48]

Reference Emissions Scenario	High Emissions Scenario	Low Emissions Scenario
Undercutting. Wastes exposed above natural materials in the cliffs are undercut by wave erosion and periodically collapse onto the beach. Wastes will fall from the cliff to form a temporary talus slope.	Undercutting initially, with direct wave erosion for most of the repository. Waste will form a temporary talus slope. As sea-level rise continues, the eastern (i.e. more inland) part of the repository will experience direct erosion with wastes that are not eroded cropping out on the shore platform below mobile sand bars.	Undercutting. Wastes exposed above natural materials in the cliffs are undercut by wave erosion and periodically collapse onto the beach. Wastes will fall from the cliff to form a temporary talus slope.

The distribution of radionuclides across grain sizes in the disposed waste is complex and uncertain. Uncertainty is increased when the effects of degradation and break-up of the waste are considered. Therefore, the reference assumption is made that radionuclides are evenly distributed across grain sizes in the waste, and their distribution in the environment reflects the behaviour of bulk materials. Bulk materials include all the waste and engineering materials being eroded from the repository, and any underlying sediments which are also eroded.

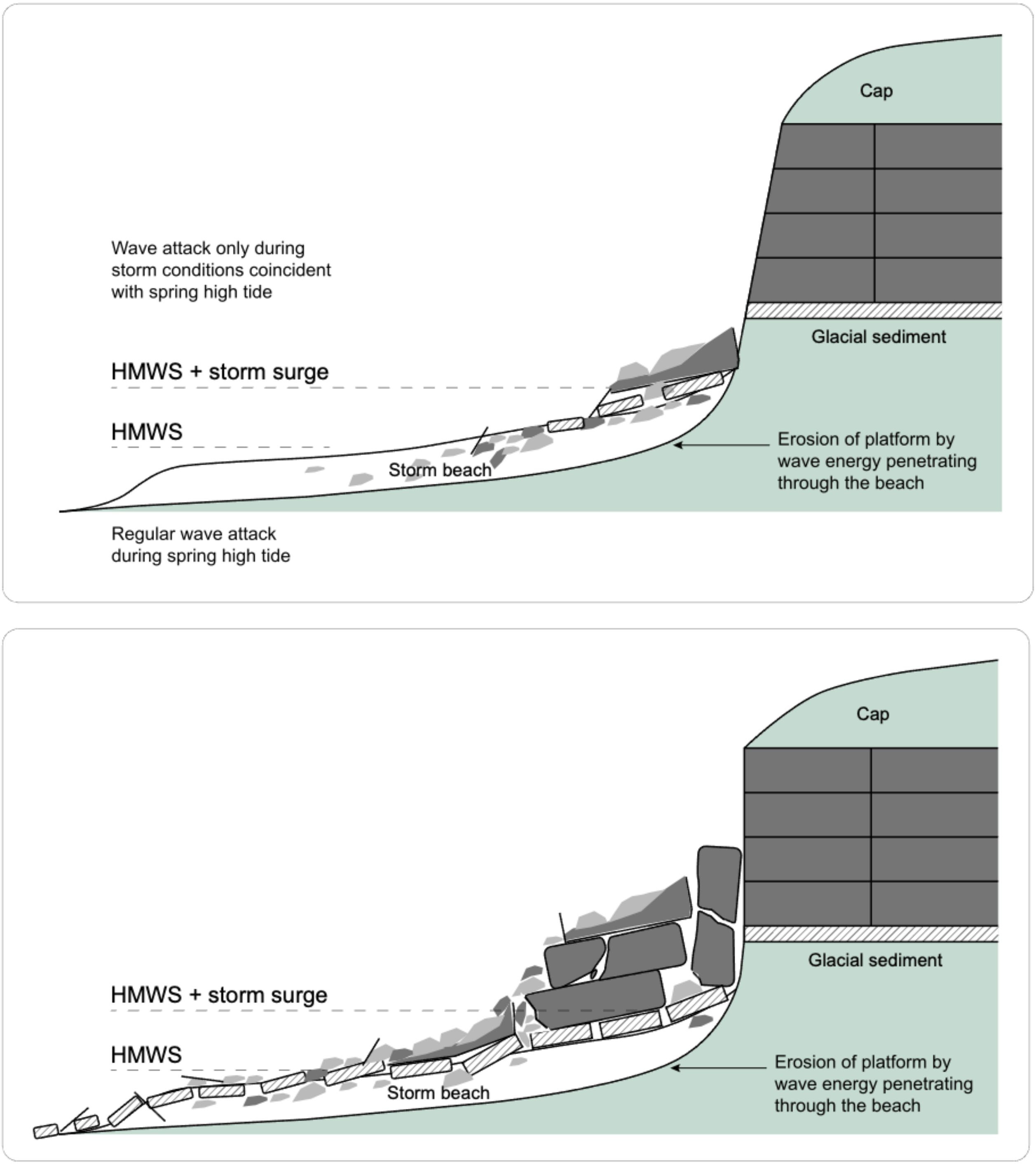

Figure 7.2: Conceptualisation of the smaller (top figure) and larger (bottom figure) talus associated with slower or faster rates of coastal erosion.

Radionuclide leaching and desorption

There will be some leaching of radionuclides from the waste during the operational phase, and post-closure prior to disruption by coastal erosion. Leaching is likely to be more significant for the trench waste than the vault waste, because the latter is containerised. The radionuclides leached from the waste prior to disruption of the repository will mainly be mobile radionuclides, including H-3, Cl-36, Tc-99, I-129 that are of relatively low importance for potential doses from disruption of the repository by coastal erosion. Therefore, leaching of the radionuclides from the waste has been neglected in the coastal erosion pathway assessment. This is a slightly cautious approach as the radionuclide inventory remaining in the repository when it is disrupted by coastal erosion will be overestimated.

Once the repository has been breached by coastal erosion, some of the water that infiltrates through the cap into the repository could discharge at the toe of the cliff. This seepage rate is expected to be so low that it would not be practicable to collect water for drinking. Doses from exposure to the water would be negligible compared with exposure to the waste in the cliffs.

A portion of the radionuclides in the foreshore, and in any in situ waste exposed on the foreshore, will be leached by tidal water flows. However, the extent to which the water would penetrate foreshore sands and any in situ waste and leach the radionuclides is uncertain. Radionuclides present in the matrix of in situ waste (e.g. activation products in irradiation metals) will not be readily released. Therefore, two bounding cases are considered to model leaching from the foreshore and in situ wastes: all radionuclides are retained with the solid phase (marine model turned off) or a sorption-controlled release is included allowing the release of radionuclides to seawater (marine model turned on).

Exposure pathways

The highest exposures can be expected to be to people that frequent the coastline where the wastes are being eroded, such as dog walkers. The main potential exposure modes are external irradiation, inadvertent ingestion and inhalation of contaminated wind-borne dust, and ingestion of contaminated marine foodstuffs. Two different exposure geometries need to be considered for external irradiation:

- exposure on the storm beach and foreshore;
- exposure from the cliffs.

With increasing distance from the cliffs, the wastes will become progressively well mixed with storm beach and foreshore materials. Furthermore, over the course of a year, any person frequenting the coast is unlikely to spend a significant portion of time in a particular location. Therefore, it is assumed that the occupancy over the storm beach and foreshore is uniform per unit accessible area, i.e. the area accessible accounting for twice daily tidal inundation of the foreshore, which tends to weight occupancy towards the beach (relatively high radionuclide concentrations) and away from the foreshore (lower radionuclide concentrations) that is only periodically accessible.

The cliffs present a more complex exposure situation, with significant uncertainties relating to the behaviour of people and the exposure geometry. The wastes will form a band in the cliffs. This may be elevated above the beach level or directly exposed at beach level, depending on the elevation of the erosion front. There will be slumps of waste material down the face of the cliffs and at the base of the cliffs, forming a temporary talus slope. The talus may have generally lower concentrations of radionuclides than the wastes exposed in the cliffs due to preferential removal of fine-grained material and mixing with clean profile and cap material that slumps from higher in the cliffs. Exposures can arise when standing several metres from the cliffs, from walking on and clambering over the slumped wastes and cliff face.

For the assessment, it is assumed that people clamber over the cliffs and talus. Some of the time will be spent clambering over the waste material directly, while the rest of the time will be spent clambering over relatively uncontaminated material if the natural sediments underlying the wastes are exposed in the cliffs. Exposures therefore depend on the average radionuclide concentration in the cliff face. While the trenches are eroding, the talus at the toe of the cliffs is expected to be small and therefore the average concentration in the cliff face includes any clean material exposed below the wastes. While the vaults are eroding, it is cautiously assumed that a large talus will form, with radionuclide concentrations in the talus the same as the concentration of the wastes exposed in the cliffs (Figure 7.2).

Radionuclides are transported into the marine environment by erosion. This process is included in the assessment model. If the elevation of the erosion front is sufficiently high, wastes below the elevation of the erosion front are not eroded and, as erosion continues, they may outcrop on the foreshore. Radionuclides can be leached from the foreshore, and any in situ waste exposed on the foreshore, and transported into the local coastal waters. Leaching is included in the assessment calculations when the marine model is turned on. A portion of the radioactivity in the marine environment will be distributed along the regional coastline (from St Bees to Ravenglass).

Occupancy of the regional coastline can lead to exposures by external irradiation, inadvertent ingestion and inhalation. Exposures can also occur from consuming contaminated marine foodstuffs from the local coastal waters (fish, molluscs, crustaceans, marine plants and algae).

7.4.2 Assessment Model and Data

Overview of the assessment model

We have developed an assessment model which represents the repository, the coast and the marine system. The model is implemented in GoldSim software. The model represents the whole lifetime of the repository, from first disposals in 1959 until the repository has been completely disrupted by coastal erosion, and residual radioactivity in the sediment sink has been buried by ongoing erosion of clean materials from the coast.

Model compartments are used to represent features of the repository, coast and marine system. Radionuclides are transferred between the compartments at rates reflecting transport processes, such as erosion. The calculated radionuclide concentrations in the compartments which represent the coast and marine system are used to calculate potential doses and risks to people occupying the coast and consuming marine foodstuffs. Key processes included in the model are:

- continuous erosion of the repository over time, and transport of eroded material onto the storm beach;

- transport and dispersal of eroded material through the coastal system into the offshore sediment sink;
- dissolution of radionuclides into seawater from suspended sediment and by tidal leaching of the foreshore sediments and any in situ waste exposed on the foreshore;
- uptake of radionuclides from seawater into marine foodstuffs;
- doses to people from occupying the contaminated coast (external irradiation, inhalation of particulates and inadvertent ingestion) and consumption of marine foodstuffs.

Model compartments

The LLWR site has been discretised using a grid that reflects the distribution of key radionuclides in the existing and potential future wastes. The discretisation is adjusted to reasonably capture the spatial variations of radionuclide concentrations in the trenches, represent each vault separately, and represent different waste types and stacking heights in the vaults.

The model grid is shown in Figure 7.3. Cells shaded yellow contain no waste. The inventory is assumed to be homogeneously distributed by volume within each vault. Vault 9 is separated into 'existing and committed' (covering existing LLW containers already in the vault, and LLW that is committed to disposal in the same container types) and 'future' disposals (future LLW in strengthened containers and future ILW) with the inventory assigned accordingly.

Shielded modules are assumed to contain ILW that cannot be managed in the same way as LLW. For the assessment model, the shielded modules are simplified into a single larger unit in each of Vault 10 and Vault 11 (the Vault 11 shielded module also contains the small amount of ILW that cannot be managed as LLW from Vault 12). The layout and number of the shielded modules is yet to be decided and can be easily changed as inventory projections are updated. In the reference case, the shielded modules are assumed to be distributed east to west across the vaults to maximise containment, by minimising the amount of activity potentially exposed at any given time during disruption of the repository by coastal erosion. Alternative shielded module layouts are considered in variant cases.

Each grid cell is represented in the GoldSim model using compartments (GoldSim 'container' elements). The existing and potential future radionuclide inventories are progressively added to the relevant compartments assuming a linear rate of disposals between the opening and final disposal dates for each trench or vault.

If the elevation of the erosion front is sufficiently high, wastes below the erosion front are not eroded. As erosion continues, these wastes may outcrop on the foreshore. Therefore, for each grid cell, the waste is proportioned into two cells; one representing waste above the erosion front which will be eroded onto the storm beach, and one representing waste below the erosion front which will remain in situ and outcrop on the foreshore).

The storm beach is represented by three compartments (north, middle and south); these represent three sections of the beach below the eroding repository. The foreshore and the offshore are each represented as a single compartment. This simulates the progressive mixing from the cliff, through the storm beach, to foreshore and offshore sediments, see Figure 7.3. Forwards and reverse transfers between the foreshore and local offshore sediment compartments are used to represent mixing of these sediments, but with net offshore transport (Figure 7.1).

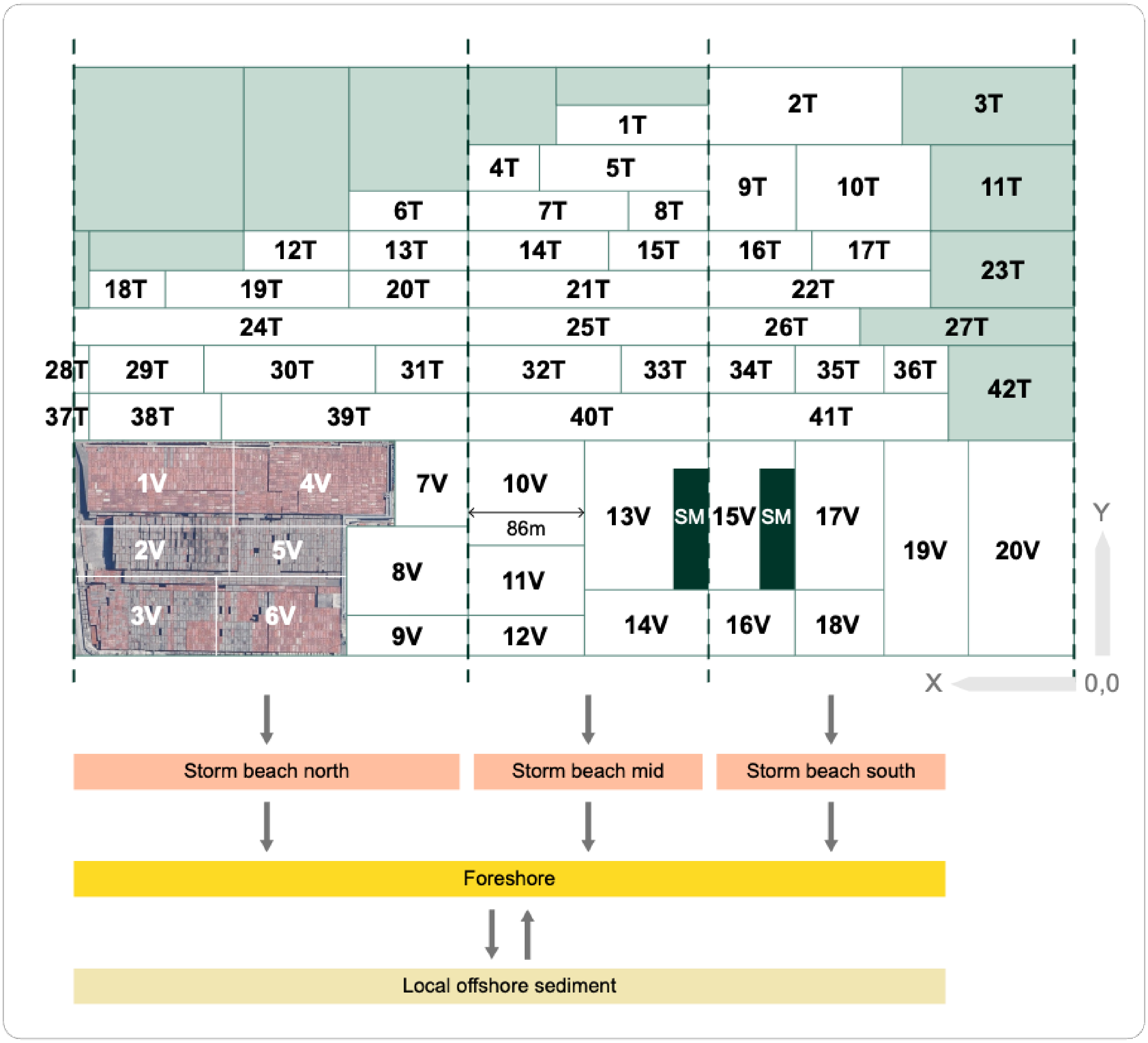

Figure 7.3: Compartments and transfers in the repository and coast model

The marine model represents transfers between the foreshore, local offshore water and sediment, and sink cells, as shown in Figure 7.1. There is exchange of sediment between the local offshore sediment compartment and sediment sink with net transfer to the sink (Figure 7.1). Material eroded from the repository mixes with clean material eroded from the wider coastline between St Bees and Ravenglass in the sink cell. As erosion continues, material deposited in the sink cell is progressively buried. Once the repository is completely eroded, ongoing erosion of the coast results in all the material eroded from the repository being covered with clean material.

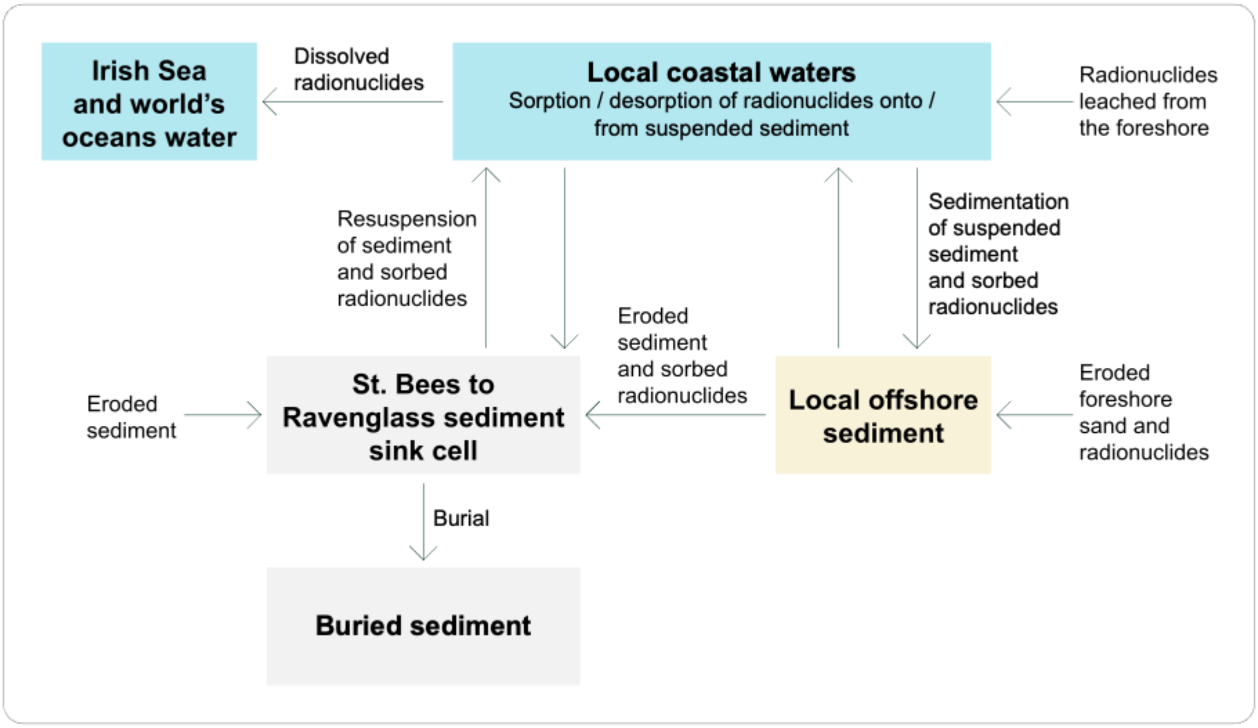

Figure 7.4: Compartments and transfers in the marine model

Table 7.2 shows the derivation of compartment dimensions for the beach, foreshore and further compartments on the coastal model. The transport processes between these compartments are further described below.

Table 7.2: Compartment dimensions for coastal and marine areas

Compartment	Value
Storm Beach	Length of LLWR disposal area, parallel to coast; sub-divided into North (312.7 m), mid (197 m) and South (140 m). Values change as site erodes and site length changes. Present-day width. North storm beach 50m, mid and South storm beach 40 m (based on the range given in reference [105]). Depth derived from volume per unit length, which is assumed to be as the present day [105]. This gives depths consistent with trial pits dug on the storm beach and foreshore [105]. North storm beach 2.0 m, mid and South storm beach 2.1 m.
Foreshore	Length of the eroding disposal area. Present day width, 400 m [105]. 0.5 m depth based on trial pits described in reference [105]. The thickness is described as being generally <0.5 m, increasing up to 2 m where there are sandbars. 0.5 m is taken as a cautious mean.
Local offshore sediments	Length of the eroding disposal area. Width 1500 m. Particle mobility assessments indicate that sand can be mobilised at distances up to 1500 m offshore. Assume depth as foreshore, i.e. 0.5 m. Consistent with observed foreshore sediment mixing depths, e.g. level changes and movement of sandbars.
St Bees to Ravenglass upper	Length, 25 km (approximate distance from St Bees to Ravenglass).

Compartment	Value
St Bees to Ravenglass lower	Width 1500 m. Particle mobility assessments indicate that sand can be mobilised at distances up to 1500 m offshore. The upper compartment also includes the foreshore between St Bees and Ravenglass (+ 400 m). Assume upper compartment has the same depth as the foreshore, i.e. 0.5 m. Depth of the lower compartment increases with time to represent the amount of material eroded from the entire coastline.
Marine water	Length, 25 km - correlates with St Bees to Ravenglass sub-cell. Width, 1500 m - correlates with St Bees to Ravenglass sub-cell. Average depth of 5 m (0 m at mean water, 10 m at 1.5 km offshore). Note that the compartment volume is smaller than assumed in the groundwater pathway and PoA biosphere models [23] [14]; this reflects the assumed behaviour of eroded materials, and hence distribution of contaminated sediment.

Erosion of the wastes and radionuclide concentrations in the cliffs

As the erosion front passes through the compartments representing the repository, radionuclides are transferred from the waste compartments to the storm beach compartments. The timing of the erosion of each compartment is determined by its initial distance from the coast and the erosion rate.

The model can represent a uniform regression of the coastline or preferential erosion of particular areas. For the latter purpose the erosion front is sub-divided into three zones (north, middle and south). Different erosion rates and distances from the coast can be set for each part of the front. This controls the erosion rate and the timing of onset of erosion. Erosion of the north, mid and south parts of the repository results in transfer of the wastes onto the north, mid and south parts of the storm beach.

The compartments that are being actively eroded form the cliff line. Trench and vault wastes and contaminated underlying sediments may be exposed in the cliffs. Cliff line radionuclide concentrations are calculated for each section of the cliff. The length weighted average concentration in the whole cliff line is then calculated.

The model represents differing sea-level rises by calculating exposure either from wastes exposed at the elevation of the beach, or wastes suspended in the cliffs with a specified thickness of underlying sediments exposed beneath them. The elevation of the beach at the base of the cliffs may gradually increase during erosion of the repository. In the assessment model this is represented by specifying different elevations when the vaults and trenches are eroding. The elevation when the vaults are eroding is taken to be the elevation when disruption of the repository starts (10.5 m OD in the reference case). The elevation when the trenches are eroding is taken to be the elevation when disruption of the repository finishes (12.5 m OD in the reference case).

During erosion of the trenches, there is expected to be a small talus at the toe of the cliffs so the average cliff concentration includes any underlying clean sediments. The size of the talus when the vaults are eroding depends on the timing of exposure and the extent of degradation of the waste, containers and engineering components. There is assumed to be a large talus in the High and Reference Emissions Scenarios, which completely covers any underlying clean sediments exposed in the cliffs, and a small talus in the Low Emissions Scenario.

As noted above, uncontaminated cap materials are not considered when calculating concentrations in the cliff that form the source for radiation exposure. This is a cautious approach. However, these materials are taken into account when calculating transport and dispersal of the eroded material and radionuclide concentrations on the storm beach and foreshore, as part of the volume balance.

Wastes which are below the elevation of the erosion front remain 'in situ' and are exposed on the foreshore as erosion progresses. The fraction of the in situ wastes which are exposed is calculated from the progression of the erosion front, the proportion of the waste that is not covered by foreshore sands (20%, estimated from the exposure of gravel lags as an analogue), and the proportion of time the wastes are covered by tides (a sinusoidal function).

Transport and dispersal of the eroded material

As discussed in Subsection 7.4.1, conservation of volume forms the main physical basis for the dispersion of eroded radionuclides and is used to define the transfer rates used in the model. Radionuclides are assumed to be evenly distributed across grain sizes in the waste, and their distribution in the environment reflects the behaviour of bulk materials.

Radionuclides in the storm beach are transported onto the foreshore, and then to the local offshore sediments, and finally to the deeper offshore sediments in the St Bees to Ravenglass coastal 'sub-cell'. Each of these regions is represented by a model compartment.

The transfer rates between compartments are automatically calculated within the model for each compartment using the ratio of volume of material eroded from the cliff per unit time and the volume of the compartment, i.e. storm beach, foreshore, etc. That is, to conserve volume, the same net material flux must move progressively through the model, see Figure 7.1.

There is longshore drift to the north and south, with no net drift, consistent with the swash alignment of the coast. The best approach to representing longshore drift in the assessment model is uncertain. In the 2011 ESC model, material which moves alongshore was assumed to be retained in the local offshore compartment, reflecting movement away from the foreshore in front of the eroding repository. Noting that recreational users of the coast would also use adjacent areas of the foreshore, which would contain radionuclides transported by longshore drift, a different approach has been adopted for the 2026 ESC. In the reference case, longshore drift is excluded from the model, and all eroded radioactivity is retained on the coast in front of the repository, maximising radionuclide concentrations.

In addition to the movement of eroded material from the cliff, through the storm beach, to the marine environment, there is also continuous mixing of foreshore sand and offshore sediments. Sand on the foreshore is assumed to mix with sand in the local offshore sediments once a year.

The model also represents the mixing between the sediments in the local offshore area and in the wider St Bees to Ravenglass sediment body. Uncertainty in the rate of mixing is large, but consistent with the foreshore mixing process it is assumed that one foreshore volume of sediment exchanges between the local offshore and the St Bees to Ravenglass compartment per year. The recreational PRP is not exposed to these offshore sediments. The St Bees to Ravenglass sediments are expected to mix with foreshore sediments along the regional coastline, to which the occupational PRP is exposed. Radionuclide concentrations in the foreshore sediments along the regional coastline are assumed to be the same as in the St Bees to Ravenglass sediment compartment.

Two model compartments are used to represent the St Bees to Ravenglass sediment sink (Figure 7.4). The depth of the upper compartment is constant. As eroded material is deposited in the upper compartment, material is transferred to the lower compartment at the same volumetric rate. This represents burial of material eroded from the repository and wider coastline between St Bees and Ravenglass. The thickness, and therefore the volume, of the lower compartment increases with time (Table 7.2).

In the real coastal system, only sand is retained in the St Bees to Ravenglass sub-cell. Silts and clays are transported further offshore and deposited in the Eastern Irish Sea mudbelt. As the distribution of radionuclides across grain sizes is uncertain, all sediment and associated radioactivity is assumed to be retained in the St Bees to Ravenglass sub-cell in the model. This model bias results in doses to occupational users of coast and marine food stuff consumers being overestimated, because the radioactivity is retained within a smaller area.

The preferential association of radionuclides with particular sized materials is represented in variant calculation cases by modifying the model transfer rates by a factor. This has the effect of increasing or decreasing the residence times of radionuclides on the beach and foreshore compared with bulk materials. Although modifying factors are applied to the transfer rates to represent radionuclide grain size associations, all the eroded sediment and associated radioactivity is still retained in the St Bees to Ravenglass sediment sink cell.

Marine model

The marine waters associated with the foreshore, local offshore and St Bees to Ravenglass sub-cell sediments are well-mixed and are represented as a single cell in GoldSim. Two processes result in transfer of radionuclides from the eroded sediment to the marine water: leaching from the foreshore and any in situ waste exposed on the foreshore, and desorption from suspended sediment. Sorption and desorption are modelled using solid-liquid distribution coefficients, K_d, and assuming linear, instantaneous, reversible sorption.

Deposition and re-suspension of local offshore and St Bees to Ravenglass sediments is modelled explicitly. Radionuclides dissolved in marine water that flows out of the St Bees to Ravenglass sub-cell are represented in the assessment model as a transfer from the marine water compartment to a sink compartment (Irish sea and world's oceans).

Exposure model

Annual effective doses are calculated from environmental radionuclide concentrations, standard dose coefficients [57], occupancy and intake rates (Subsection 7.4.3). The following exposure pathways are considered:

- external exposure - from the cliff, storm beach and foreshore in front of the eroding site and from contaminated foreshore sediments along the coast;
- inhalation of suspended sediment - as above;
- inadvertent ingestion of sediment - as above;

- ingestion of marine foodstuffs - taking account of radionuclide uptake in marine foodstuffs and annual intakes of foodstuffs.

7.4.3 Potentially Representative Persons (PRPs)

It is envisaged that the cliff face, beach, foreshore and the marine environment are freely accessible to members of the public for both recreational and occupational purposes. Based on local, e.g. [106] [107] [108], and national habit survey data [109], a generic occupancy value of 2,000 h y^{-1} has been selected for occupational users of the regional coast at greatest risk, e.g. an inshore fisherman, of which 1,500 h y^{-1} is spent onshore and 500 h y^{-1} in boats.

Local habit survey data indicate a 95th percentile occupancy of 293 h y^{-1} on the LLWR coast for people who participate in leisure activities such as dog walking, playing, horse riding [43]. This covers a 2.82 km length of coastline from Carl Cragg to Kokoarrah Scar. A recreational PRP is assumed to spend a length-weighted proportional amount of time directly in front of the eroding repository. They are assumed to spend around 5 h y^{-1} on the cliffs and talus, with the remaining time divided between the storm beach and foreshore in proportion to their relative useable areas.

According to present-day observations, the coastline adjacent to the LLWR is most commonly used for leisure walking, including dog walking, facilitated by the presence of the Drigg Shore Road. Occasional leisure fishing or bait collecting may take place, but full-time fishing activities are more likely to be based at Ravenglass, Seascale, St Bees or other coastal villages; at the present day, commercial fishing occurs from the port at Whitehaven.

On this basis, we choose to assess risks to the following PRPs, on the basis that the PRP exists at all times:

- **Recreational users of the coast adjacent to the LLWR**, conceived as dog walkers. Individuals spend a fraction of 293 h y^{-1} in front of the repository, calculated as a proportion of the time-varying repository length out of the full 2.82 km stretch covered by the habits survey. 5 h y^{-1} (around 7% of their time in front of the eroding repository) is spent on the cliffs. The rest of their time in front of the eroding repository is divided between the storm beach and foreshore, accounting for the foreshore being covered by the tide, and therefore inaccessible, for 50% of the time.
- **Occupational users of the St Bees to Ravenglass coast**, conceived as inshore fishermen. Individuals spend 1,500 h y^{-1} on the beach and foreshore areas at a location somewhere along the St Bees to Ravenglass coastline, plus 500 h y^{-1} in a boat in coastal waters. The PRP is regarded as being representative of exposures that might occur anywhere along this 25 km stretch of coastline.

High-rate (local) marine foodstuff consumers. This PRP could overlap with recreational or occupational users of the coast, but it is informative to calculate the annual dose separately. Individuals are assumed to consume marine foodstuffs harvested within the

St Bees to Ravenglass sub-cell at rates derived in reference [43]: 57.1 kg y^{-1} of marine fish, 40.7 kg y^{-1} of crustaceans, 12.6 kg y^{-1} of molluscs and 0.5 kg y^{-1} of marine plants and algae.

- In addition, 'what-if' calculations are made for the more pessimistic case that the beach in front of the eroding repository could be used, possibly seasonally, by an inshore fisherman, or other individual having some occupational connection with the local beach or residing on the beach. The individual spends 1,500 h y^{-1} on the beach and foreshore areas adjacent to the repository and is assumed to have a cabin or caravan on the beach, 10 m from the cliff, increasing their exposure to the cliff waste. We consider this case to be unlikely, i.e. the annual probability of occurrence is less than one, as structures built on the storm beach would be damaged by storms on timescale of around a decade [25].

As noted previously, the parameters governing leaching of radionuclides from the foreshore are uncertain, so the assessment model has the option to turn this process, and the wider marine component of the model, on and off. When calculating doses to recreational users of the coast, the marine model is turned off, maximising retention of radionuclides on the foreshore in front of the repository. Leaching from the foreshore (and the wider marine model) is turned on, maximising radionuclide fluxes to the marine environment, and wider coastline, when calculating doses to (regional) occupational users of the coast and high-rate consumers of marine foodstuffs. Calculating doses to occupational users of the coast with the marine model on is the preferred approach, as the marine model includes processes that are expected to occur, even if there is uncertainty in parameterisation of the processes.

7.5 Assessment Calculation Results

Subsection 7.5.1 defines the reference case and alternative assessment cases. Subsection 7.5.2 describes the results of the reference case at a level intended to illustrate the functioning of the model and provide reference case results. Subsection 7.5.3 describes the results of the alternative assessment cases more briefly and compares the results to the reference case results. Fuller presentation of the results is given in reference [25].

7.5.1 Assessment Cases

A reference calculation case has been developed which provides a central projection of disruption of the repository by coastal erosion and calculated impacts. This case uses best estimate parameter values but takes a cautious approach where the most realistic representation cannot be robustly quantified (Subsection 2.5.1).

The reference case is evaluated deterministically. Deterministic variant calculation cases are then used to explore the key uncertainties. Table 7.3 lists the variant cases and describes the changes compared with the reference case. The uncertainties can be grouped under the headings of sea-level rise projections, conceptual model uncertainty, modelling approach and parameter uncertainty, and 'what-if' situations.

Table 7.3: Calculation cases for coastal erosion assessment

Case	Description
Reference case	
Reference	Reference emissions, sea-level rise and erosion projection. Swash-aligned erosion of the repository. Undercutting and direct erosion with larger talus when the vaults are eroding and smaller talus when the trenches are eroding. Longshore drift excluded, retaining radionuclides on the eroding site frontage. With marine model on and off.
Variant sea-level rise projections	
Low projection	Low emissions, sea-level rise and erosion projection. Small talus.
High projection	High emissions, sea-level rise and erosion projection.
Conceptual model uncertainty cases	
Reduced amenity	Qualitative discussion of how changes to the amenity value of the storm beach might affect occupancy and dose.
Direction of erosion	Variant 1: Earlier and faster erosion of the northern part of the repository than the southern part. Variant 2: Earlier and faster erosion of the southern part of the repository than the northern part.
No SM ILW	The inventory of ILW in shielded modules is excluded from the repository. The remaining waste is distributed throughout the entire vault.
No ILW	The inventory of ILW is excluded from the repository. The remaining waste is distributed throughout the entire vault.
Shielded Module variants	Variant 1: The SMs are in the southeast corners of Vaults 10 and 11. Variant 2: The SM in Vault 10 is in the southeast corner. The SM in Vault 11 is in the southwest corner. Variant 3: The SMs extend over the full east-west width of the vaults.
Small talus	A small talus when the vaults are eroding.
Radionuclide grain size associations	Qualitative assessment of grain size associations for key radionuclides and associated key waste streams. Scaling factors are used to describe the relative transport rates of radionuclides and bulk materials.

Modelling approach and parameter uncertainty cases	
No sediment burial	The sediment sink cell is represented using a single compartment, so materials and radionuclides deposited in the sink cell are always assumed to be well-mixed by processes such as resuspension.
Longshore drift	Longshore drifted represented explicitly using additional compartments.
Spatial heterogeneity of the vault inventory	The impacts of spatial variations in the inventories of radionuclides in the vaults at smaller scales than described by the assessment inventory are assessed semi-quantitatively using the results of relevant conceptual model uncertainty cases.
Sea-level rise	Cases to explore uncertainty in the relationship between the amount of sea-level rise and the rate of coastal erosion. Higher and lower sea-level rise variants are considered.
Marine sorption K_ds	Maximum and minimum K_d values are assumed for the top 4 radionuclides contributing to doses to the marine foodstuff consumer, with non-zero K_d.
'What-if' cases	
'What-if' local residence PRP	A residential PRP is present on the coast in front of the eroding repository.
Credible maximum sea-level rise	Sea-level rise associated with the 95th percentile of the International Panel on Climate Change's (IPCC) Representative Concentration Pathway (RCP) highest (8.5) emissions scenario.
Ice sheet collapse	Explores the large uncertainties associated with projecting sea-level rise for extreme greenhouse gas emission cases.

7.5.2 Analysis of the Reference Case

A high-level description of the reference case is provided in Table 7.3. More details concerning our assumptions are provided below.

- Erosion of the disposal area (vault waste) commences at 1,250 y after present.
- There is a 5 m increase in sea level over the next 1,250 y, resulting in erosion by undercutting of wastes.
- Erosion continues through the repository at the same rate as during the approach, so that the site is completely eroded by 3,440 y after present.

- The erosion front is aligned with the current, swash aligned orientation of the coastline and hence parallel to the long axes of the vaults and the trenches.
- Radionuclides are assumed to be uniformly distributed across grain sizes, not taking account of any specific waste associations.
- The model runs from 1959, the year of first disposals to the LLWR, and represents gradual emplacement of wastes using recorded and anticipated future inventories, assuming a constant rate of disposal to each trench or vault between the dates of opening and closure of each trench or vault. Radioactive decay and in-growth are modelled from the time of disposal.

Erosion and dispersion of radionuclides

Figure 7.5 shows the calculated radionuclide concentrations (summed over all radionuclides) in the different sections of the cliff line with time. Since the reference case considers uniform erosion of the wastes, the peaks and troughs in concentration can be associated with the successive erosion of the vaults and shielded modules, Trenches 3, 2, 1, 4, 5, 6 and 7.

The average concentration in the cliffs along the site frontage is also shown. This average concentration is used to calculate external irradiation doses from the cliffs. The average concentration is directly proportional to the inventory, with the highest concentrations in the shielded modules which contain ILW. These are present in the Mid (Vault 10) and South (Vault 11) sections but not the North section, explaining the lower concentrations there.

There is a low concentration of long-lived radionuclides in Trench 3. The update of the grid discretisation since the 2011 ESC means that Trench 3 and Trench 2 are discretised separately, so the higher concentrations in Trench 2 are better resolved. This is potentially a cautious approach as it assumes that all wastes in Trench 2 and Trench 3 are exposed at separate times. Given the small widths of the trenches (about 25 to 30 m) this requires the erosion front to be almost perfectly aligned with the long axis of the trenches (maximum of 1 or 2 degrees variation), with little variability along its length.

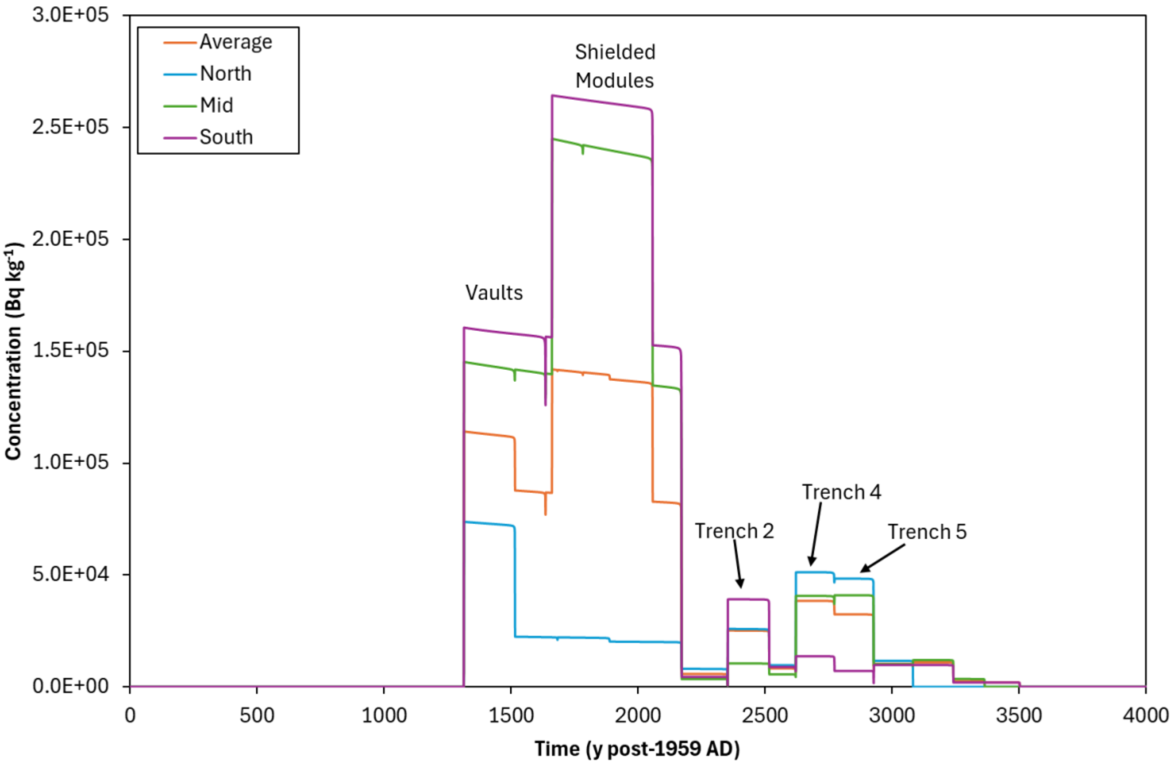

Figure 7.5: **Radionuclide concentrations in sections of the cliff for the Reference case**

Figure 7.6 shows calculated radionuclide concentrations in different parts of the coastal model - cliff, beach, foreshore and offshore. Radionuclide concentrations progressively decrease from the cliff to the beach, to the foreshore, and offshore, as radionuclides are progressively dispersed and diluted.

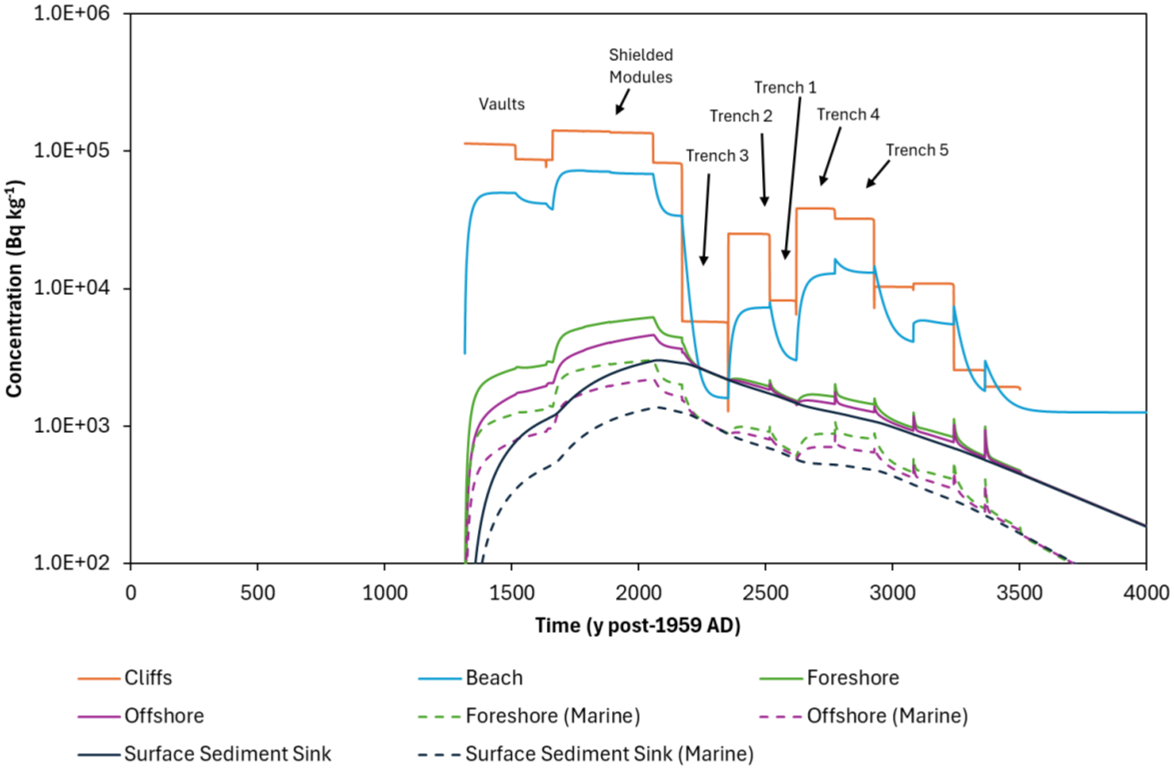

Figure 7.6: **Environmental concentrations for the reference case with the marine model switched off (solid lines) and on (dashed lines)**

Th-232 has a very long half-life with negligible radioactive decay over the assessment timeframe. Concentrations of Th-232 have been used to calculate factors for the dilution of radionuclides as they are transported away from the cliffs, see Figure 7.7. These are time-dependent due to changes in the waste elevation and profile thickness for each grid cell. The dilution factors are calculated at times immediately prior to a change in the cliff concentrations, when the concentrations in the beach, foreshore and local offshore are not changing significantly and are approaching equilibrium with the cliffs. The results show the expected pattern of increasing dilution with increasing distance away from the erosion front.

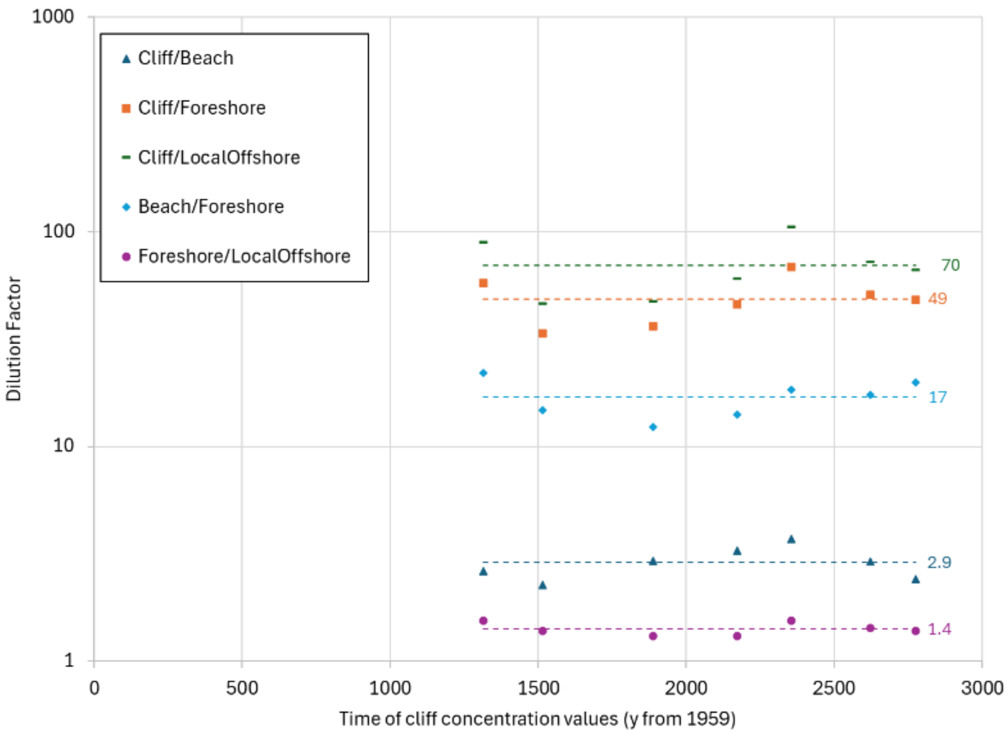

Figure 7.7: **Dilution and dispersion of Th-232 away from the eroding repository (marine model turned off)**

Distribution of activity in the model and fate of radionuclides

Figure 7.8 shows the distribution of total activity with time with the marine equilibrium desorption and sorption model turned 'off' (solid lines) and 'on' (dashed lines). This shows that activating the marine model does not significantly reduce the total activity on the beach and foreshore but does distribute the activity held in the regional offshore sediment. The total activity in the repository varies initially due to the succession of disposals to the trenches and vaults, i.e. activity being added to the model, and the decay of short-lived radionuclides. As the repository begins to be eroded, radionuclides are transported onto the storm beach, foreshore and offshore sediments in the St Bees to Ravenglass sub-cell. Only a few percent of the activity is present on the storm beach and foreshore at any time. Once the repository has been completely eroded, residual activity on the storm beach and foreshore is rapidly transferred into the offshore sediments.

The effect of the marine model is that a significant proportion of the activity of more soluble radionuclides is lost to the world's oceans. Sorbing radionuclides, however, remain mainly in the offshore marine sediments.

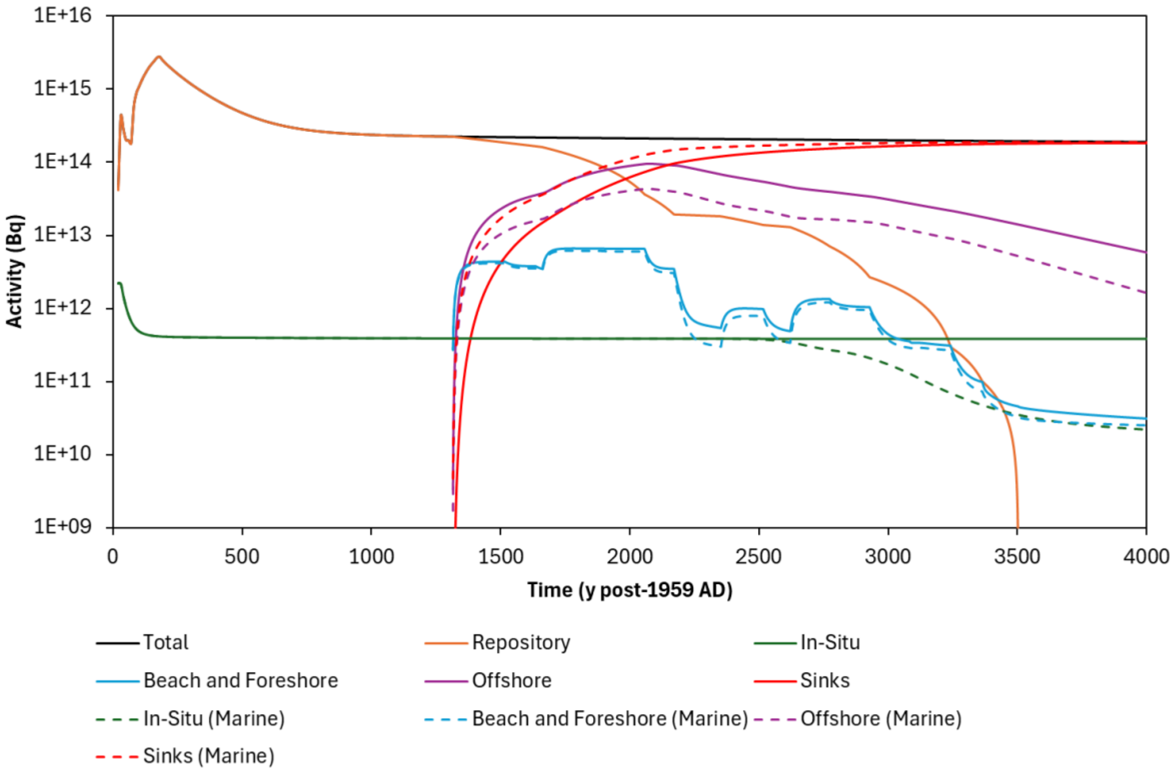

Figure 7.8: **Activity distribution for the reference case with marine model switched off (solid lines) and on (dashed lines)**

Annual effective doses

Recreational user of the coast adjacent to the eroding facility

Figure 7.9 shows the calculated annual doses as a function of time for each exposure pathway (external irradiation, ingestion and inhalation) and the total annual dose. The annual doses are dominated by external irradiation from the cliffs and the storm beach (which includes external irradiation from cliffs while occupying the beach), due to Nb-94 disposed in the vaults and particularly in the Vault 10 shielded modules, and due to Th-228 and Ra-228 ingrown from Th-232 in the trenches. The peak annual dose from the vaults is 11 μSv and from the trenches is 22 μSv.

Table 7.4 provides a summary of the peak dose and breakdown of the contributions by key radionuclides.

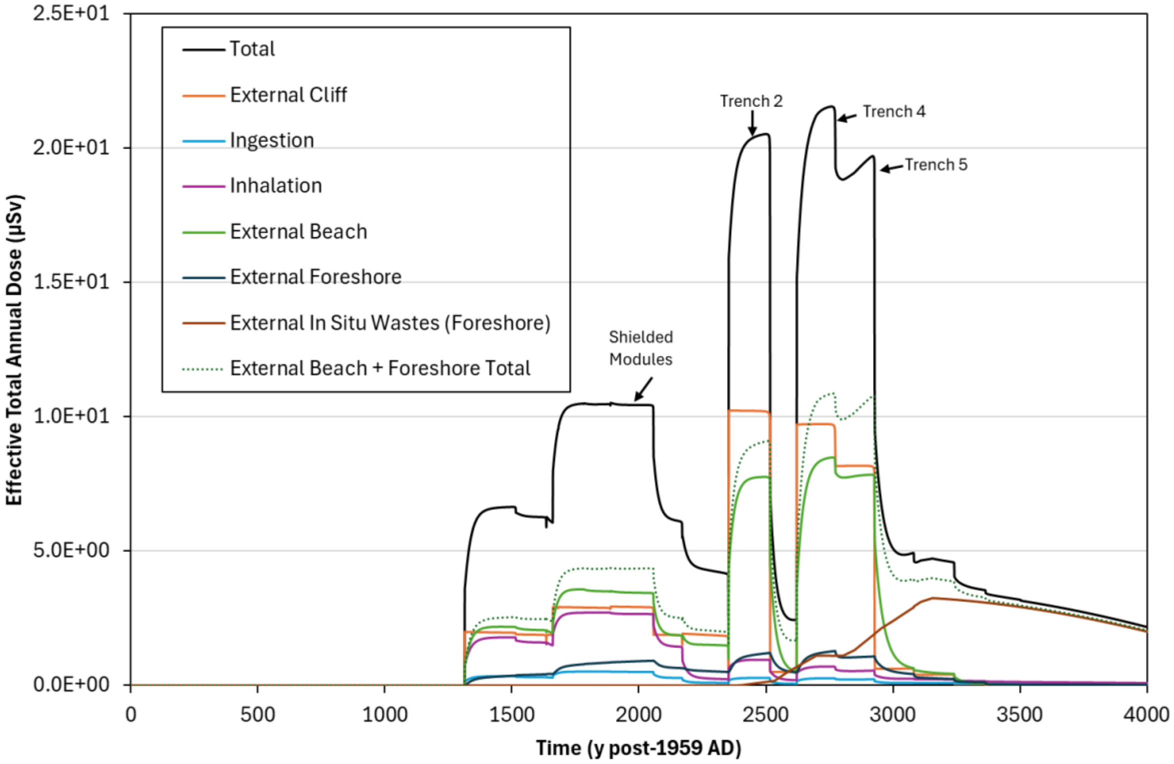

Figure 7.9: Total annual effective dose per pathway to the recreational PRP for the Reference case with the marine model switched off

Table 7.4: Peak annual effective doses to the PRP in the Reference Case

Source	Peak dose (µSv)	Key contributing radionuclides
Vaults (LLW + ILW)	11	Nb-94, Pu-239
Trenches	22	Th-228, Ra-228, U-238

Doses to children were also calculated, using child-specific inhalation and ingestion rates and dose coefficients. Habits surveys suggest that maximum occupancies for children and infants are considerably lower than for adults, and their occupancy is weighted towards the foreshore where radionuclide concentrations are lower [25]. A calculation was undertaken cautiously assuming that children have the same occupancies as adults. Given these cautious occupancy assumptions, the resulting peak dose to a child recreational beach user (25 µSv from the trenches, or 10 µSv from vaults), is deemed similar enough to the adult doses that is not necessary to consider doses to children for variant cases.

Calculated dose rates can be compared with recent measurements of dose rates from the waste and on the coast (Figure 7.10). The dose rate when standing on the LLW containers in

Vault 8 is around 2 μSv hr^{-1}. This is very similar to the peak dose rate from the cliffs when the trenches are eroding, but around a factor of three higher than the peak dose rate from the cliffs when the vaults and shielded modules are eroding. The locations of coastal dose rate measurements from the RIFE and Sellafield Environmental Monitoring reports are not available. The dose rates have contributions from background activity and liquid effluent discharges to the Irish sea. The contributions from both sources are expected to be reasonably uniform across the coast. The measured dose rates are broadly consistent with expected natural background (0.1 μSv h^{-1} = 0.9 mSv y^{-1}) indicating effluent discharges are a minor contributor to the total dose rate.

Calculated dose rates on the storm beach are a factor of three higher than the measured dose rates when the vaults and shielded modules are eroding, increasing up to a factor of eight higher when the trenches are eroding. However, peak dose rates on the foreshore (excluding in situ waste) are a factor of three lower than the measured dose rates. This shows that calculated dose rates on the main area of the coastline, i.e. the foreshore, would be slightly elevated slightly above background.

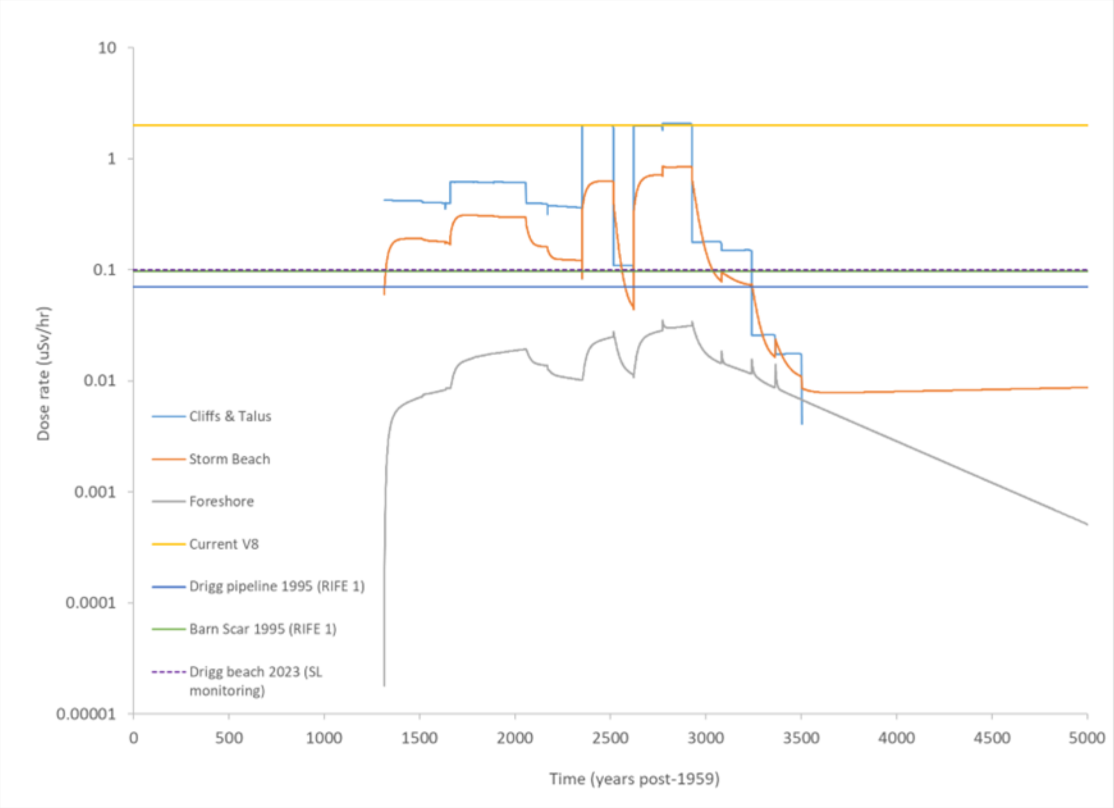

Figure 7.10: Comparison of contemporary and potential future dose rates

Occupational user of the St Bees to Ravenglass coast

Figure 7.11 shows the average radionuclide concentrations in marine sediment in the St Bees to Ravenglass compartment, with the marine model turned on. Foreshore sand between St Bees and Ravenglass is assumed to have the same concentration as the offshore marine sediment, except for the coastal section immediately in front of the LLWR.

The radioactivity is dominated by similar radionuclides to those giving rise to peak doses to the recreational PRP, notably Nb-94, Th-228, Pu-239 and Ra-228.

Figure 7.12 shows annual dose to occupational users of the St Bees to Ravenglass coast. The pattern differs from that seen for the recreational user, who is exposed to waste as it is eroded. The occupational user of the coast is exposed to the accumulation of radionuclides that have been dispersed into the offshore sediments along the coast and then returned to the regional foreshore. As a result, the calculated doses reflect the cumulative amounts of radionuclides that have been eroded. Therefore, annual doses to the occupational user of the coast show a general trend of increasing dose with time until all the key radionuclides have been eroded from the site.

The peak annual dose is 19 µSv at about 3,000 y post-1959, or 18 µSv before the trenches start eroding (due to contributions from the vaults only).

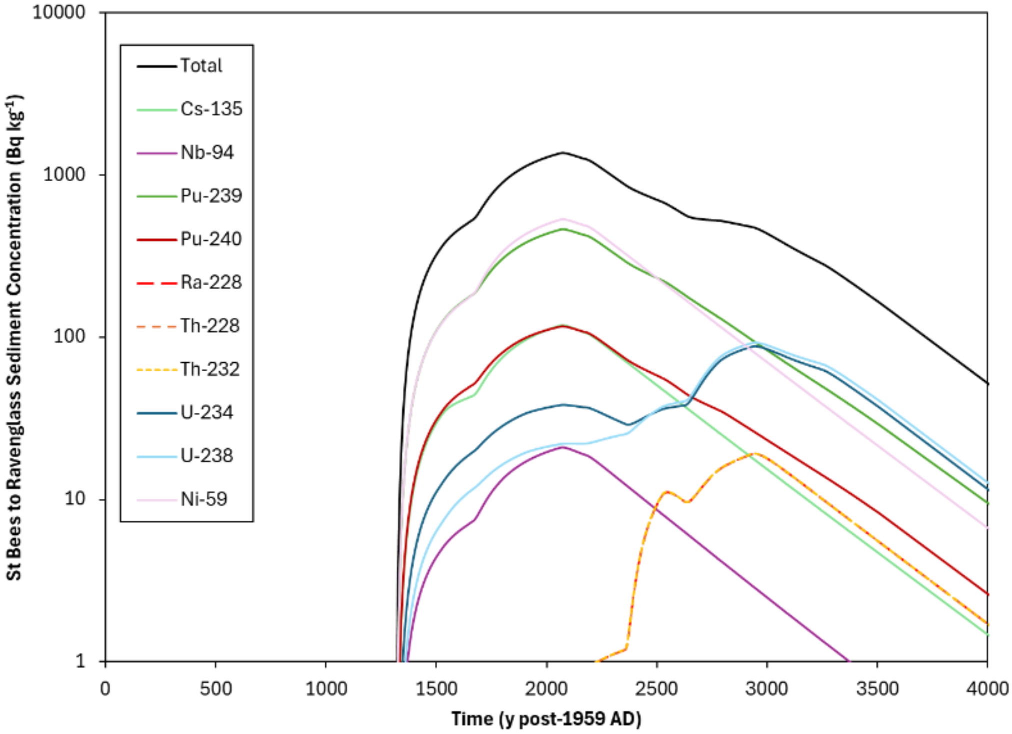

Figure 7.11: Radionuclide concentrations in the St Bees to Ravenglass surface sediment compartment for the reference case with the marine model switched on

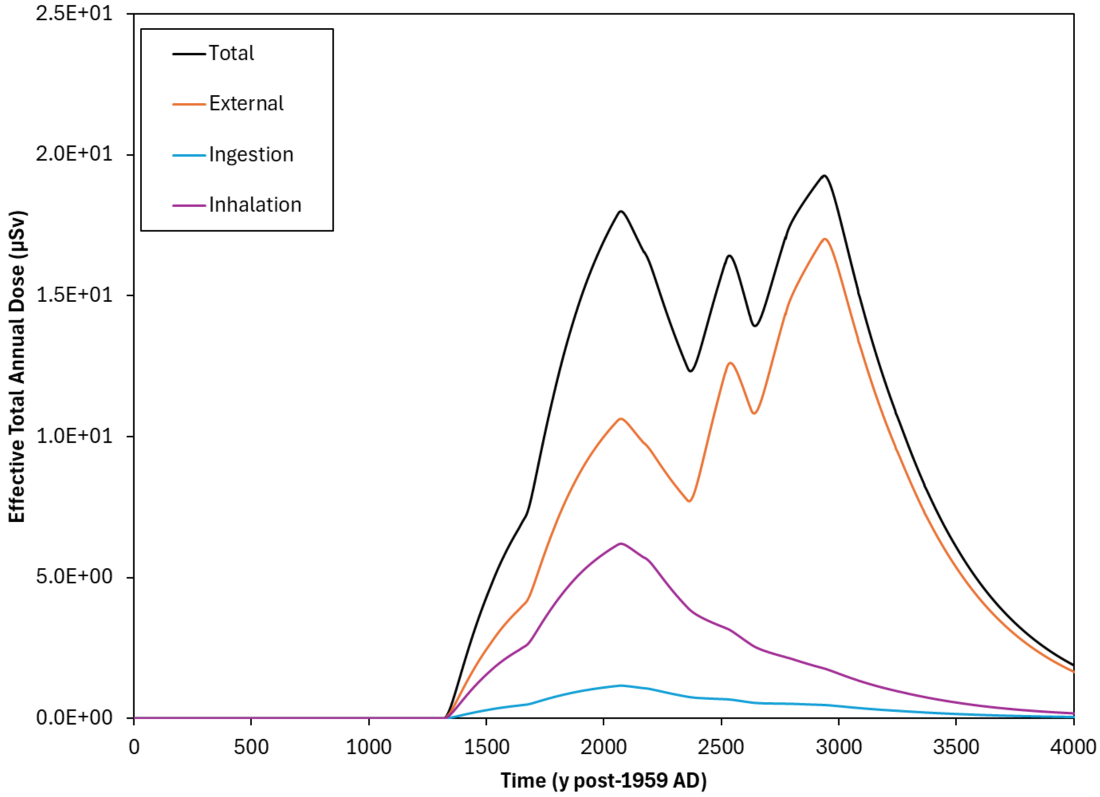

Figure 7.12: Total annual effective dose per pathway to the occupational PRP for the reference case with the marine model switched on

High-rate marine foodstuff consumer

Figure 7.13 shows annual doses from high-rate consumption of local marine foodstuffs. These are dominated by doses from Pu-239. The peak total dose is 15 μSv and occurs immediately following complete erosion of the shielded modules. The dose falls off as sediments in the regional St Bees to Ravenglass cell are gradually buried.

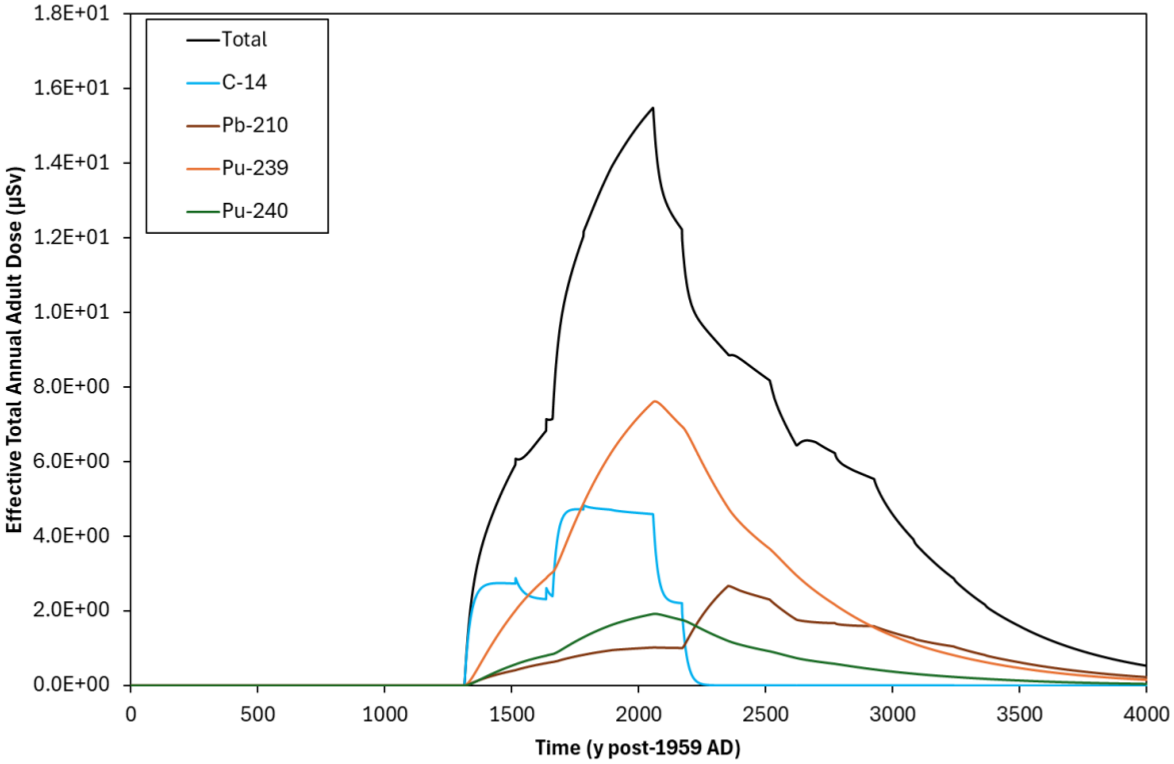

Figure 7.13: Total annual effective dose per radionuclide to the high-rate marine foodstuff consumer PRP for the reference case with the marine model switched on

The potential long-term impacts of residual radioactivity in the environment following complete erosion of the repository are assessed by comparison with environmental radionuclide concentrations from operational discharges to the Irish sea and from other UK nuclear sites. The results show that modelled sediment activity concentrations are comparable to those that have been routinely measured in the regional coastal environment for Pu-239+240, but orders of magnitude lower than measured concentrations for other radionuclides. The reference case shows that uranium concentrations should gradually decrease to background levels. This builds further confidence that the residual activity would not lead to significant impacts.

7.5.3 Alternative Cases

Results for the alternative cases are presented in full in reference [25]. Here we present results from the High and Low Emissions Scenarios and summarise results for the other cases.

High and Low Emissions Scenarios

In the Low Emissions Scenario, sea-level rise is lower than in the reference case and erosion is slower, with erosion of the repository commencing 1,750 y after present. There is a small talus at the base of the cliff when the vaults are eroding. All waste is eroded by undercutting; there is no in situ waste remaining to be exposed on the foreshore.

In the High Emissions Scenario, sea-level rise is higher than in the reference case and erosion is faster, with erosion of the repository commencing 750 y after present. There is a large talus at the base of the cliff when the vaults are eroding, as in the reference case. There is no undercutting of the trenches. All the trench wastes are below beach level when erosion of the trenches occurs, so the only exposure to radionuclides in the trenches is when in situ waste is exposed on the foreshore.

Figure 7.14 and Figure 7.15 show the dose to the recreational PRP as a function of time for each exposure pathway, for the Low and High Emissions Scenarios respectively.

In the Low Emissions Scenario, the peak dose is reduced compared with the reference case by a factor of 1.3. This is due to the lower beach level, the small talus assumption for the vaults, and no exposure to in situ wastes.

In the High Emissions Scenario, the peak dose from the vaults is increased by a factor of 1.3 compared to the reference case due to less dilution of waste in the storm beach and additional exposure to in situ wastes on the foreshore. By contrast, the peak dose from the trenches is decreased by a factor of 4 compared to the reference case due to the different mode of erosion. This result shows that when there is high and rapid sea-level rise during erosion of the trenches, the dose from the trenches is sensitive to the single value of sea-level chosen for the calculation. In practice, the sea-level would be expected to rise gradually throughout erosion of the repository, but a fixed value is used for the vaults and a higher fixed value for the trenches.

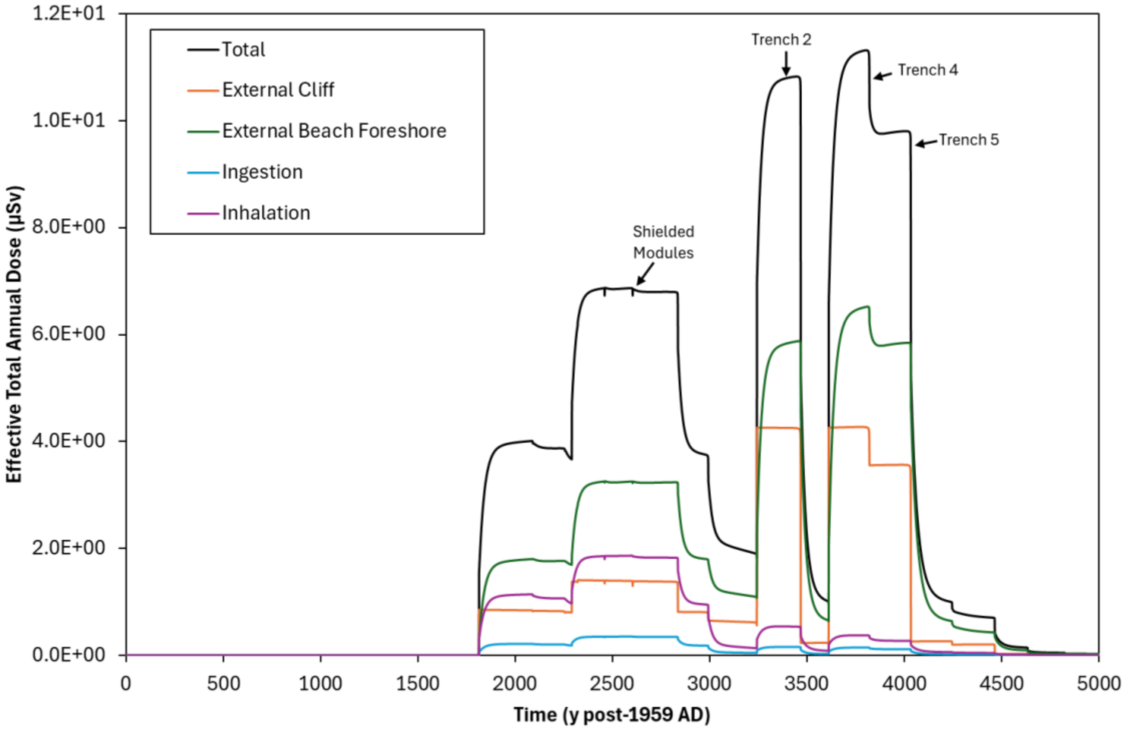

Figure 7.14: Total annual effective dose per pathway to the recreational PRP for the Low Emissions Scenario with the marine model switched off

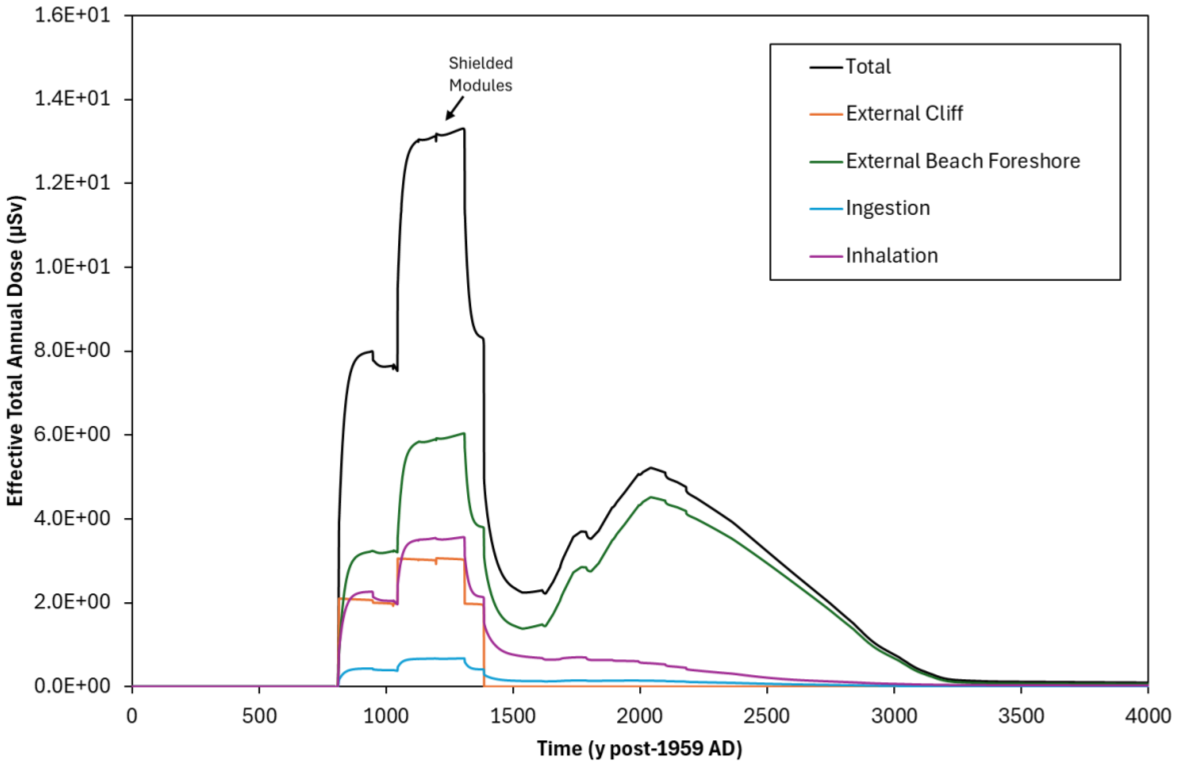

Figure 7.15: Total annual effective dose per pathway to the recreational PRP for the High Emissions Scenario with the marine model switched off

Other alternative cases

Peak annual doses for the reference and all alternative cases are shown in Table 7.5.

The peak dose from vault wastes reduces by a factor of two if the ILW in shielded modules is excluded from the assessment. If only LLW is considered in the assessment, the peak annual dose from vault wastes is reduced to 2.1 µSv. The key contributing radionuclides during erosion of the vaults from LLW are Ra-226, Th-228 and U-238.

The three shielded module layout sensitivity cases demonstrate the impact of large-scale heterogeneity. When the length of the shielded modules is decreased in the East-West direction (perpendicular to the erosion front), the ILW in the shielded modules is eroded over a shorter period. This increases the peak fluxes of radionuclides released to the coastal environment. By contrast, doses are not sensitive to heterogeneity of the waste in the North-South direction (parallel to the erosion front), as this is averaged out by receptors spending time equally across all areas in front of the repository.

The results show that the most significant biases and uncertainties are associated with the radionuclide particle size associations, which affect the radionuclide transport velocities compared with bulk materials, and the sorption distribution coefficients for radionuclides in the coastal and marine environment.

In the radionuclide grain size associations cases, calculations are undertaken assuming all radionuclides are transported a factor of five faster or a factor of five slower than bulk materials. These factors are towards the bounds of what is expected based on analysis of certain waste streams and radionuclides, so they are likely to be too extreme when applied to all waste streams and radionuclides. Consideration of key waste streams indicates that key radionuclides in the vaults are more likely to be transported at similar or slower rates than bulk materials, while key radionuclides in the trenches are more likely to be transported at similar or faster rates than bulk materials. Therefore, peak doses are unlikely to be as high as calculated in the slow case.

Similarly, in the marine sorption distribution coefficients cases, minimum and maximum K_d values are applied to the top four contributing elements with non-zero K_d (Pb, Pu, Ra and Th). Radionuclide concentration ratios (CRs) may be negatively correlated with sorption coefficients, but this corelation is not included in the variant case. If this correlation was included, doses calculated with maximum sorption coefficients would tend to be lower than calculated and doses calculated with minimum sorption coefficients would tend to be higher than calculated. However, these are close to bounding cases because it is unlikely all the key radionuclides for doses from marine foodstuffs would have sorption coefficients at their minimum or maximum values.

Table 7.5: Peak annual doses for reference and alternative cases (µSv). Values for vaults in parentheses.

Case	Local Recreational (µSv)	Occupational (µSv)	Marine Foodstuffs (µSv)
Reference	22 (11)	19 (18)	15 (15)
Low projection	11 (6.9)	14 (14)	12 (12)
High projection	13 (13)	25 (25)	23 (23)
Direction of erosion - faster north	16	-	-
Direction of erosion - faster south	21	-	-
No SM ILW	21 (6)	18 (11)	9 (9)
No ILW	21 (2.1)	17 (3)	5 (3)
SM variant 1 - East	22 (15)	-	-

Case	Local Recreational (μSv)	Occupational (μSv)	Marine Foodstuffs (μSv)
SM variant 2 - staggered	22 (13)	-	-
SM variant 3 - spread	22 (9)	-	-
Small talus	22 (9)	-	-
Radionuclide grain size associations - finer (faster radionuclide transport)	16 (6)	21 (19)	-
Radionuclide grain size associations - coarser (slower radionuclide transport)	44 (32)	14 (14)	-
No sediment burial	14 (13)	-	12 (12)
Longshore drift	25 (14)	-	-
Lower sea-level rise	14 (10)	-	-
Higher sea-level rise	11 (11)	-	-
Maximum marine sorption K_d	-	-	7(7)
Minimum marine sorption K_d	-	-	36(27)

A variant 'what-if' case considers doses to a local residence PRP that is semi-resident, e.g. in a beach hut or caravan, in front of the eroding repository. This 'what-if' case does not reflect behaviours observed on the LLWR coast today or deemed credible for the future, as the hut or caravan would be damaged by storms that overtop the beach. The PRP is assumed to spend 1,500 h y^{-1} on the beach and foreshore directly in front of the eroding repository. The total dose to this receptor (peak of 511 μSv) is much greater than for the recreational user, reflecting the extended occupancy of the storm beach. The probability of this case is much less than one but cannot be determined. Therefore, a risk cannot be calculated [19].

7.6 Summary of Results and Assessment

Coastal erosion is identified as the mode of natural disruption for the LLWR. We project that the repository will start to be disrupted by coastal erosion several hundred to a few thousand years after the present, with erosion of the repository being complete within one to a few thousand years after the present. Other possible modes of natural disruption, e.g. cap erosion or glacial scouring, will be of little effect or not occur before coastal erosion disrupts the facility [13] [63]. We consider that onset of coastal erosion before 750 years is unlikely; by this time the radiological activity in the repository will have declined by an order of magnitude. After 300 years, the short-lived activity in the repository will have decayed, so the remaining hazard only decreases a little during the expected time window for erosion of the repository (Figure 3.18).

The coastal erosion conceptual model developed for the 2026 ESC provides integrated projections of emissions, sea-level rise, and the timing, rate and mode of erosion. Notable features of the model are its capability to represent the spatial heterogeneity of the wastes, and the explicit treatment of the erosion of wastes and their dispersion from the eroding cliff, through the storm beach and foreshore, and into the marine environment. The model is flexible enough to enable a range of uncertainties to be explored.

The calculated concentrations of radionuclides in different parts of the shore environment change with time as different parts of the LLWR, containing different wastes, are eroded. The calculated concentrations reproduce the dilution of waste with co-eroded materials and due to foreshore and offshore sediment tidal mixing and sediment burial. Thus, for example, the radionuclide concentration in the foreshore sands is a factor of about 50 less than at the cliffs. Assessed annual doses to PRPs take account of exposure to the different parts of the local coastal environment and more widely dispersed radionuclides. Locally, the dominant exposure pathway is external irradiation from waste present on the beach and foreshore; along the coast, external exposure to more generally dispersed contamination and doses via marine foodstuffs are important.

Three reference PRPs are considered:

- a locally based recreational user of the shore adjacent to the LLWR site, e.g. a dog walker;
- an occupational user of the coast basing their activities anywhere between St Bees and the Ravenglass Estuary, e.g. an inshore fisherman;
- high-rate consumers of marine foodstuffs that are harvested from the local coastal water between St Bees and the Ravenglass Estuary.

The reference case considers erosion of the vaults by undercutting, beginning at 1,250 years after present, and passing through the repository aligned parallel to the long axis of the vaults and then each trench in turn.

The reference calculation case yields a peak annual dose of 22 µSv to the recreational PRP, principally due to Th-228 and Ra-228 ingrown from Th-232 in the trenches. The peak annual dose from the vaults of 11 µSv occurs earlier and is mainly due to Nb-94 disposed in the Vault 10 shielded modules. The peak annual dose to the occupational PRP is 19 µSv, contributed to by radionuclides from both the trenches and the vaults. Calculations of potential dose from high-rate ingestion of marine foodstuffs lead to an annual dose of 15 µSv, dominated by Pu-239.

Annual doses to the recreational PRP show peaks as specific wastes are eroded. Doses to the occupational PRP and marine foodstuff PRP reflect concentrations in the wider marine sediment and show a general trend of increasing dose with time until all the key radionuclides have been eroded from the repository, followed by decreasing doses.

Variant cases have investigated the impact of biases and uncertainties related to the timing and rate of erosion, whether the vaults and trenches are eroded by undercutting or direct erosion, the alignment of the erosion front, alternative inventory and shielded module layouts, radionuclide grain size associations, sediment burial and longshore drift assumptions, sorption coefficients, and doses from discrete items.

The most significant biases and uncertainties are associated with:

- the amount of relative sea-level rise, and therefore the elevation of the back of the storm beach, at the base of the cliffs, when the trenches are eroded;
- radionuclide-particle size associations, which affect the radionuclide transport velocity compared with bulk materials; and
- sorption distribution coefficients for radionuclides in the coastal and marine environment.

Doses are directly proportional to the radionuclide inventory. If only LLW is disposed, the peak dose to the recreational PRP from the vaults is 2.1 µSv.

The spatial distribution of the inventory in the vaults can also affect the calculated peak doses. The peak doses quoted above assume the radionuclide inventory in LLW and ILW that can be managed as LLW, in each vault, or vault sub-area, is uniformly distributed in that area. The shielded modules are spread west (coastwards) to east (inland) across the vaults, minimising the amount of activity exposed by coastal erosion at any given time. Variant layout cases show that this is an optimal approach that minimises potential impacts from erosion of the repository.

Overall, the results provide confidence that with appropriate WAC, and tracking use of the radiological capacities, the potential impacts from coastal erosion of the vaults can be limited to the levels described in the GRA. A separate assessment of the dose from casual encounters with discrete items on the beach has been undertaken to confirm that they do not add disproportionately to doses from this assessment (covered in Subsection 10.4).

8 Assessment of Human Intrusion and Scavenging of Eroding Waste

8.1 Assessment Scope

This section describes calculation of doses from inadvertent human intrusion into the repository and from scavenging waste exposed by coastal erosion. Although scavenging is not an intrusion event, human intrusion and scavenging are grouped together as they are both assessed against the dose guidance level (Subsection 2.13.4.2).

At the end of the PoA active institutional control will cease. Physical barriers preventing access to the repository will be removed. Passive institutional control, such as planning covenants will likely remain. As the nature of the repository progressively fades from memory it is increasing likely people would inadvertently intrude into the repository.

The GRA (paragraph 6.3.41 in reference [19]) requires assessment of human intrusion into the facility assuming the intruder does not have prior knowledge of the disposal facility, or that the intruder has knowledge of the existence of underground workings but does not understand what they contain. It is not necessary to assess intrusions undertaken with full knowledge of the existence, location, nature and contents of the disposal facility; the environment agencies take the view that a society that preserves full knowledge of the disposal facility will be capable itself of exercising proper control over any intrusions into the disposal system. Therefore, the human actions that must be assessed are deliberate acts, for example, to excavate a void or recover materials, but where the intruder is uninformed or oblivious concerning the radiological hazard. Inadvertent exposures following an accident that uncovers waste are also relevant.

The scope is to assess the potential radiological impacts from inadvertent human intrusion into the repository after the end of the PoA, consistent with requirement R7 of the GRA [19]. A range of different events can be conceived. Many of these events would not penetrate through the cap and expose the waste, for example the foundation trenches for a typical house. However, some events involving deeper intrusion (around 5 m to 13 m deep depending on location on the cap) could expose some waste. For example, this might include drilling site investigation boreholes or the digging of deep trial pits. Construction of a building with deep foundations, or a house with a basement, could breach important barriers in the cap, increasing potential radon concentrations in a house or building.

8.2 Previous Assessments and Environment Agency Comments

We have developed our approach through previous iterations of assessment to give confidence in the range of potential events that are assessed and our approaches to calculating doses from these events. It is not possible to assess every conceivable type of intrusion event. However, we assess a broad range of representative events, with exposure pathways and durations underpinned by data for undertaking the same activities elsewhere. Some of these events have very low probability (e.g. aircraft crash into the repository) or are towards the upper end of behaviours (e.g. depth of intrusion assumed for constructing a septic tank or sewage system).

We expect people will use the coast for recreational and occupational purposes effectively uninfluenced by the waste (Section 7). These people will be exposed to radionuclides released to the coastal environment and they will have casual encounters with resilient waste items that have not fully degraded e.g. pausing to look at them, picking up and examining them. As such exposures are expected, they should be assessed against a dose guidance level of 20 μSv, which is equivalent to the risk guidance level assuming exposure occurs. Conversely, deliberate scavenging of materials, investigation of, or excavation into the exposed waste, i.e. investigating or disturbing the waste at its original disposal location, should be assessed against the dose guidance level. This is consistent with the GRR [28]. Informal scavenging and organised materials recovery are discussed in this section, as they are assessed against the dose guidance level.

Our 2011 ESC assessment approach is summarised below.

- Consider, based on current technology and motivations, the potential modes of intrusion taking account of site characteristics, facility design, site supervision and record keeping.
- Identify the intrusion events and modes that are most likely (in that they are known practice) and relevant (in that they have capacity for direct excavation of the waste or damage to the engineered or natural barriers).
- Identify the events to be carried to quantitative assessment. Record reasons for not carrying forward other events.
- Derive parameter values for the quantitative assessment cases based on realistic site investigation and building practices.
- Calculate doses both to intruders and to people subsequently exposed as a result of the intrusion, e.g. as result of excavated waste left on the ground surface, taking account of heterogeneity of disposed waste at an appropriate scale.
- Compare calculated doses with the GRA (paragraph 6.3.36 in reference [19]) Requirement R7 Dose Guidance Level range of 3 to 20 mSv y^{-1}: 20 mSv y^{-1} being generally appropriate for exposures of limited duration, for example associated with an event or occurring during one year but not subsequent years, and 3 mSv y^{-1} being appropriate for exposures taking place over several years.
- Use calculated doses as the basis for deriving allowable radionuclide concentrations for future waste disposals, to be considered in the development of WAC, see reference [110].

- Confirm that no intrusion events can be envisaged that would lead to deterministic effects, including exceeding dose thresholds for severe deterministic injury to individual body tissues (paragraph 6.3.40 in reference [19]).

As we have developed our approach, we have consulted the Environment Agency. Our 2011 assessment report [88] provides more information on the discussions leading up to the 2011 ESC including written submissions and responses.

We reviewed our approach to the human intrusion in the 2011 ESC in preparation for the 2026 ESC [111], taking into consideration:

- technical queries and recommendations resulting from the Environment Agency's review of the 2011 ESC;
- relevant regulatory developments in the UK that have taken place since 2011;
- recent developments in international and national guidance for human intrusion assessments;
- recent UK human intrusion assessments carried out for decommissioning nuclear sites and landfills.

We drew the following conclusions.

- The approach used in the 2011 ESC should be retained.
- The human intrusion assessment for the 2026 ESC should include exposure to discrete items of waste, particles and sealed sources.
- Aircraft impact should be added to the list of events that are assessed quantitatively.
- The consequences of intrusion events for the evolution and performance of the repository should be assessed.

The Environment Agency commented on the outcomes of our review [112]. Priorities for the Environment Agency were to understand more details on the scope and implementation of the assessment. Other comments indirectly supported the conclusions of our review and identified potential approaches to building confidence in the assessment.

8.3 Approach in this Assessment

We have broadly retained the approach and human intrusion events assessed in the 2011 ESC. However, we have improved some aspects of the approach.

We have reviewed the potential intrusion events to ensure they remain reasonable and robust to build confidence that activity limits derived from the assessment will continue to protect people who might intrude into the waste without knowledge of its radioactive nature.

We have updated the details of the conceptual models for the intrusion events, for example to take to account for changes in the repository design and closure engineering. These changes include plans to limit the stack heights for existing and committed containers, move to stronger containers that can be stacked higher with top of stack CPUs, potential to dispose of ILW in the vaults, including within shielded modules, and an increase in the minimum thickness of profile material from 1 m to 2 m.

We have updated our assessment calculations to provide more detailed resolution of the areas of the repository with the highest concentrations of the key radionuclides for doses from human intrusion. We have updated our assessment model data, for example to use the latest internationally recommended dose coefficients [57].

Our 2011 ESC assessed the potential doses from handling and taking home low-activity sealed sources exposed by coastal erosion of the repository [103]. We also undertook a simple assessment of potential doses from scavenging and taking home items of waste exposed by coastal erosion [102]. Subsequently, we developed radionuclide activity limits that would protect people from encounters with discrete items of waste at close proximity and included these in our WAC [60].

We have reassessed the potential doses from scavenging low activity sealed sources and discrete items exposed by coastal erosion (Table 2.1). We have also used disposal records for Vault 8 to examine the potential doses associated with events that might intrude into 'hotspots' of activity within a vault, e.g. drilling a borehole through an area (section) of the vault with relatively high concentrations of the key radionuclides for potential doses from human intrusion.

This section explores the potential doses from human intrusion into different areas of the repository and deliberately interacting with waste exposed by coastal erosion (Subsection 2.13.4.2). Potential doses from particles, sealed sources, discrete items and hotspots are discussed in Section 10.

Human intrusion into the repository will damage the final cap, which is a most important barrier to radionuclide release [13] [12]. We have considered the types of damage that would be associated with the intrusion events included in the 2026 ESC. We have then considered how this damage could affect long-term evolution and performance of the repository. This is discussed in Subsection 8.7.

8.4 Assessment Model and Data

8.4.1 Human Intrusion Events

8.4.1.1 Events Considered in the 2011 ESC

The human intrusion events identified by reference [90] for the 2011 ESC were classified as follows:

- Events that could occur at any time after disposal.
- Events that could occur after the PoA.
- Events possible during coastal erosion of the repository.

The events considered in each category identified for the 2011 ESC were as follows:

A. Events that could occur at any time after disposal:
    1) Aircraft crash.

B. Events that could occur at the site after the PoA:
    1) Geotechnical investigations
    2) Onsite water abstraction well
    3) Housing development
    4) Light industrial development
    5) Heavy or high-rise development
    6) Leisure development
    7) Farm/smallholding
    8) Building with cellar
    9) Cap removals
    10) Large infrastructure development.

C. Events possible after coastal erosion impacts on the site:
    1) Informal scavenging
    2) Material recovery by local small contractor
    3) Larger-scale, planned excavation
    4) Technical or archaeological investigation.

For the 2011 ESC, a discussion of each type of intrusion event was provided in reference [90]. In each case, the reasons for retaining the event for quantitative assessment or the arguments for excluding it were provided. Events were generally excluded where the likelihood is exceedingly low due to the site conditions, or the potential dose consequences were judged to be similar or less than for an event already selected for assessment.

The following events were subject to quantitative assessment as part of the 2011 ESC:

- B1: Geotechnical investigations
- B3: Housing development
- B7: Smallholding
- C1: Informal scavenging during coastal erosion.
- C2: Local organised material recovery during coastal erosion.

8.4.1.2 Events Considered in the 2026 ESC

A key initial stage of the 2026 ESC human intrusion assessment was to revisit the events assessed in the 2011 ESC, to ensure that these comprise a reasonable and robust set of events for assessment. This was accomplished by undertaking a completely new examination of the full range of events that could, in principle, take place in West Cumbria, using a systematic approach to the derivation of events.

Tree diagrams were used to identify specific aspects of human behaviour and characteristics, such that FEPs describing human intrusion events can be identified and recorded. The FEPs, and associated events, derived using this approach was then reviewed against the FEP lists provided by references [56], [54], [113], and [114].

Each FEP was then classified according to whether it requires explicit consideration in the 2026 ESC, or whether the FEP is screened from further consideration (e.g. because LLWR site is unsuitable for a particular activity, the potential consequences are low, or some other reason). Full details are provided in reference [24].

The outcome of the systematic review was a set of events for inclusion in the 2026 ESC that is very similar to the set included in the 2011 ESC. Given that the systematic review was independent of the work undertaken for the 2011 ESC this builds confidence the events included in the 2026 ESC are comprehensive and relevant to the site conditions. The events identified for inclusion in the 2026 ESC are summarised in Table 8.1.

The aircraft crash case was not included in the 2011 ESC. However, following review of the 2011 ESC by the Environment Agency [115] and subsequent work summarised in reference [116], it was decided to include aircraft crash in the events assessed. Although an aircraft could crash into the repository at any time, potential doses were only calculated for crashes that could occur after the end of active institutional control.

Potential exposure to radon in a house with a basement was also added to the list of events. In the 2011 ESC, this was included in the gas pathway assessment. Even though this event does not involve exposing or contacting waste, consistent with paragraph 6.3.42 of the GRA, this is treated as a human intrusion event because it involves breaching some of the key barriers to radon release.

The probability that an inadvertent aircraft crash occurs at the LLWR is low [116]. The probability people will undertake geotechnical investigations of the repository, intrude into or through the cap, or scavenge waste exposed by coastal erosion of the repository cannot be determined. However, these events are all plausible and considered to be more probable than an aircraft crash.

We acknowledge that it is always possible to imagine other intrusion events, to characterise events differently, or to take account of more specifically located intrusion events. However, we consider that a reasonable and robust set of events have been identified and included in the assessment for comparison with the dose guidance level and to derive controls on radionuclide concentrations [17].

Table 8.1: Events selected for quantitative assessment

Period	Identifier	Name	Description
A. Events that could occur at any time after disposal	A1(a-b)	Aircraft crash	A light aircraft crash would impart insufficient energy to penetrate the engineered cap and profile fill. A large aircraft crash could be of sufficient energy to penetrate to waste depth, resulting in exposure to some waste. Doses would be received by the emergency services that attend the crash (A1a), and subsequently the crews that attend to clean up the site and crash debris (A1b). Doses are calculated for an aircraft crash after the end of the PoA.
B. Events that could occur after the PoA	B1(b, d, e)	Geotechnical investigations	Borehole drilling and trial pit excavations could be undertaken as part of geotechnical investigations. The presence of the cap and profile materials above the wastes would significantly reduce the potential for intrusion into waste. The excavation of a trial pit could disturb waste where the profile fill underlying the cap is thinnest (and if the trial pit was of sufficient depth). Selected contaminated samples may be taken to a laboratory for inspection. Trial pit investigations (B1b) and borehole drilling (B1d) are included in the human intrusion assessment. Additionally, the laboratory analysis of borehole samples (B1e) (which bounds doses received by alternative trial pit samples) are included in the human intrusion assessment.
	B3(a-b)	Housing development	Construction activities for housing developments would include shallow excavations and cap disturbance to prepare the site and install roads and services. Foundations for domestic and light buildings would typically be unlikely to penetrate the 3-m-thick engineered cap and 2-m-thick profile fill. A type of construction that might intersect waste is the excavation of tanks for a sewage treatment plant on site. Excavated material could be used as backfill and in landscaping. Those involved in the excavation (B3a) would be exposed to the hazard and, in the long term, site occupants (B3b) could be exposed to contaminated materials that remain in the surface environment.

Period	Identifier	Name	Description
	B7	Smallholding	Construction activities might result in contaminated material being left at the surface, although it is more likely that such materials would be disposed of. Exposure pathways for occupancy of a smallholding on contaminated material include consumption of contaminated foodstuffs from the site. Occupancy of a smallholding (B7) is more cautious than a larger farm because it allows crops to be grown on a more concentrated activity source.
C. Events possible during coastal erosion of the repository	C1	Informal scavenging	Waste exposed in the cliffs by coastal erosion may become a target for occasional scavenging. Radiological doses would arise from being close to exposed waste and handling contaminated items, followed by further exposure after taking the object home (C1).
	C2(a)	Organised material recovery	Waste exposed in the cliffs by coastal erosion could become a target for organised recovery, involving small groups of local people or local workers (e.g. farmer with digging equipment). Organised material recovery is included in the human intrusion assessment (C2a).

8.4.2 Overview of the Assessment Model

The assessment model is implemented in GoldSim. The model is used to calculate the spatial distribution of activity remaining in the repository with time. The calculated radionuclide concentrations are used to calculate the doses from each intrusion event, into relevant areas, with time. The main steps in the calculations are:

1) Calculate radioactive decay and ingrowth of the radioactive inventory in different parts of the repository.
2) Estimate concentrations of radionuclides in relevant media after a human intrusion event taking account of dilution of wastes with other materials (e.g. profile fill or soil).
3) Convert concentrations into radiation doses using simple stylised models that reflect the habits of those exposed.

As discussed previously, the repository has been discretised into a 2D grid (plan view) that reflects the spatial distribution of key radionuclides for potential impacts from human intrusion, radon gas and inadvertent intrusion into the repository (Figure 3.12). The same approach was used for the 2011 ESC, but the grid has been revised for the 2026 ESC to reflect changes to the optimised repository design and development plan. Discretisation has also been increased to better define areas with higher concentrations of key radionuclides. Consistent with the approach used in the coastal erosion and radon assessments, shielded modules are represented as distinct features in the relevant grid cells.

In the assessment model, each grid cell and shielded module is represented by a single compartment (a GoldSim ‘cell’ element). GoldSim calculates radionuclide decay and ingrowth for the inventory in each compartment, and radionuclide concentrations in the waste represented by each compartment.

The radionuclide inventory in each compartment is assumed to be disposed at the closure date for the relevant trench, vault or shielded module. Decay and ingrowth calculations for the inventory in the grid cell start at the time of disposal. Therefore, radionuclide concentrations in the waste in each grid cell are calculated over time.

Doses are calculated from the end of the PoA to the start of disruption by coastal erosion, for each model grid cell and all relevant events for that cell. The relevant events are those which could penetrate through the thickness of cap and profile material in the grid cell and expose waste. The waste in all grid cells is assumed to be exposed by coastal erosion.

In some intrusion events, such as analysing borehole core samples, people are exposed to undiluted waste. In others they are exposed to a mixture of waste and clean cap and profile materials. Mixing of waste with clean material reduces the average concentrations. We assume people involved in informal scavenging (event C1) or organised material recovery (event C2(a)) are exposed to undiluted waste.

The radionuclide concentrations in each compartment are used in GoldSim to calculate the doses for the relevant intrusion events using simple analytical models.

8.4.3 Spatial Variations in Relevant Events and Conditions

The impact of future human intrusion at any location in the disposal facility will depend on the local properties of the waste and barrier system and their evolution, as well as the type of intrusion event. Radionuclide activities, waste volumes, and wasteform and barrier properties vary significantly across the vaults and trenches, and this variability requires consideration in the human intrusion assessment calculations.

The average thickness of the profile material, and the waste, has been calculated for each grid cell using a computer aided design (CAD) model. The average thickness of profile material in each grid cell is shown in Figure 6.7. The cap is 3 m thick everywhere. Boreholes could intrude through the full thickness of the repository in any model grid cell. However, deep trial pits, and excavations for septic tanks or sewer systems would only penetrate the top of the waste in a few grid cells. When calculating the depth of penetration into the waste we make some cautious assumptions.

- We assume the cap been locally thinned by 1 m through erosion, for example by runoff of rainwater. This degree of erosion is not expected [13].
- We take no credit for the top of stack CPUs as a barrier to intrusion, or in the calculation of the depth of penetration into the waste, i.e. the top of the waste is taken to be at the elevation of the top of the CPU. However, we do take credit for protection provided by the roofs of the shielded modules.

We also calculate doses from radon gas for a house with a basement for each model grid cell. The basement penetrates through the cap resistive layers (geomembrane and BES) and intersects the gas collection layer. The basement does not penetrate the underlying profile fill. Radon enters the house through the floor and walls of the basement. Radon concentrations in all floors of the house are assumed to be the same as in the basement.

The radon flux into the gas collection layer depends on the Ra-226 inventory, the bulk gas generation rate (per unit area) and the profile fill thickness (Figure 6.7). Therefore, the radon flux into the gas collection layer is grid cell specific and varies with time. Radon attenuation by the basement floor and walls also depends on the bulk gas generation rate (per unit area), so it varies with time and is grid cell specific, as is ratio of radon flux through the basement floor and the walls.

8.4.4 Exposure Model

The GoldSim implementation of the human intrusion model solves identical expressions for the calculation of dose for each human intrusion event discussed in Subsection 8.4.1 at each grid location across the disposal facility, as shown in Figure 3.12.

Where the exposure is short-term and directly related to the event (transitory exposures of no more than a few hours to weeks duration), doses are calculated as doses from the event. Where the exposure is prolonged, as could occur following an event in which radionuclides are distributed in the accessible environment, doses are calculated as annual doses. In both cases, the dose calculated is the sum of effective doses from external irradiation received, plus committed effective doses from inhalation and ingestion of contaminated material. The doses are calculated by conventional exposure models as detailed in reference [24].

The exposure pathways evaluated for each event are summarised in Table 8.2. Parameterisation of the events is described in the next Subsection.

If a single individual can be involved in several events, e.g. drilling several boreholes, excavating several trial pits, or analysing several samples, the reported dose is the sum of the dose from the appropriate number of events.

Table 8.2: Exposure pathways modelled for each human intrusion event

Event identifier	Event name	Exposure pathways
A1a	Aircraft crash (rescue workers)	Inhalation of contaminated dust: dust is generated by the penetration of the waste by crash debris. A worker inhales the dust during rescue and recovery activities. Ingestion of contaminated material: a worker ingests contaminated material because of, for example, hand-to-mouth contact and licking of the lips. External irradiation: contaminated material is exposed following the aircraft crash and a worker in proximity to this material is exposed to external irradiation.
A1b	Aircraft crash (clearing the site)	
B1b	Geotechnical investigations (trial pit excavation)	Inhalation of contaminated dust: dust is generated by the geotechnical investigation. A worker inhales the dust. Ingestion of contaminated material: a worker ingests contaminated material because of, for example, hand-to-mouth contact and licking of the lips. External irradiation: contaminated material is left on the ground by the geotechnical activities and a worker in proximity to this material is exposed to external irradiation.
B1d	Geotechnical investigations (worker involved in borehole drilling)	

Event identifier	Event name	Exposure pathways
B1e	Geotechnical Investigations (laboratory analysis of borehole core samples)	Inhalation of contaminated dust: inspection of contaminated soil and core materials generates dust. An analyst inhales the dust. Ingestion of contaminated material: an analyst ingests contaminated material because of, for example, hand-to-mouth contact and licking of the lips. External irradiation: the analyst is exposed to external irradiation while inspecting contaminated samples. Radon and thoron inhalation: the analyst is exposed to radon and thoron (indoors in laboratory) emanating from the sample.
B3a	Housing development (construction)	Inhalation of contaminated dust: dust is generated by the construction work. A worker inhales the dust. Ingestion of contaminated material: a worker ingests contaminated material because of, for example, hand-to-mouth contact and licking of the lips. External irradiation: contaminated material is left on the ground by the construction activities and a worker in proximity to this material is exposed to external irradiation.

Event identifier	Event name	Exposure pathways
B3b	Housing development (occupancy)	Inhalation of contaminated dust: dust is left on the ground at the site after the construction work. A site occupant inhales the dust. This pathway is only modelled during the time spent outside (i.e. not accounting for any ‘track back’ of contaminated dust to the indoor environment). Ingestion of contaminated material: a site occupant ingests contaminated material because of, for example, hand-to-mouth contact and licking of the lips (whilst e.g. gardening). External irradiation: the house is built on contaminated ground, and contaminated material is present in garden soil. A site occupant is exposed to external irradiation whilst indoors and outdoors. The concrete floor of the house provides some shielding from gamma radiation. Radon inhalation: the occupant is exposed to radon (indoors) emanating from contaminated material below the house.
B7	Smallholding	Inhalation of contaminated dust: dust is left on the ground at the site after the construction work. A smallholder inhales the dust whilst working outside. Ingestion of contaminated material: the smallholder ingests contaminated foodstuffs that have grown on site. A smallholder ingests contaminated material because of, for example, hand-to-mouth contact and licking of the lips (whilst e.g. gardening). External irradiation: the house is built on contaminated ground, and contaminated material is present in garden soil. A site occupant is exposed to external irradiation whilst indoors and outdoors. The concrete floor of the house provides some shielding from gamma radiation. Radon inhalation: the occupant is exposed to radon (indoors) emanating from contaminated material below the house.

Event identifier	Event name	Exposure pathways
C1	Informal scavenging	Inhalation of contaminated dust: whilst collecting materials from the eroded site, a scavenger inhales the dust. Ingestion of contaminated material: a scavenger ingests contaminated material because of, for example, hand-to-mouth contact and licking of the lips. External irradiation: a scavenger in proximity to the material revealed by erosion is exposed to external irradiation. The scavenger is also periodically exposed to external irradiation from items recovered from the site and kept at home.
C2a	Local organised material recovery (worker)	Inhalation of contaminated dust: whilst collecting materials from the eroded site, a worker inhales the dust. Ingestion of contaminated material: a worker ingests contaminated material because of, for example, hand-to-mouth contact and licking of the lips. External irradiation: a worker in proximity to the material revealed by corrosion is exposed to external irradiation.
-	House with a Basement Intruding the Gas Collection Layer	Radon inhalation: the occupant is exposed (whilst indoors and occupying the ground floor) to radon within the gas collection layer which can migrate into the basement below the house and then the ground floor.

8.4.5 Model Data

A full compilation of data used in the human intrusion model is presented in Section 9 of reference [24]; here only key parameters are discussed.

The activity of each radionuclide identified in Subsection 4.2 is allocated to each model grid cell. The radionuclide activity is based on disposal records and the Stage 2 projected future inventory, with radionuclide decay and ingrowth accounted for from the date of completion of each trench or vault.

The volumes and density of the trench waste take account of degradation and compaction, including the effects of surcharging, prior to final capping. There are assumed to be no volume decreases and density increases post surcharge and capping.

Degradation, volume and density changes in the vaults are more minor but are also represented. Again, there are assumed to be no volume decreases and density increases post surcharge (of existing and committed containers) and capping.

As noted previously, the events that could penetrate through the cap and profile material are grid cell specific. They depend on the thickness of the cap and profile material (Figure 6.7). In some intrusion events through the cap people are exposed to undiluted waste. In others they are exposed to a mix of waste and clean cap and profile materials. This is illustrated in Table 8.3.

Table 8.3: Example of the grid cell specific thicknesses of waste and clean material exposed by intrusion

Grid cell Trenches T1	Relevant events	Intrusion depth (m)	Thickness of clean material (m)	Thickness of waste (m)
Location - see Figure 3.12 Profile thickness 2.5 m - see Figure 6.7	Borehole	20 m	4.5 m* + 9.3 m = 13.8 m	6.2 m
	Site occupier or smallholder	5 m	4.5 m*	0.5 m

* Cautiously assumes 1 m thickness lost from the cap by erosion.

We have used realistic exposure pathway parameter values for the assessment calculations but have taken a cautious approach where the most realistic representation cannot be robustly quantified (Subsection 2.5.1).

- Characteristics of the category 'A' human intrusion event are based on Ministry of Defence guidance [117].

- Characteristics of the category ‘B’ human intrusion events, and the number of events that might be undertaken by a single geotechnical worker or laboratory investigator, draw on the information and elicitation studies in support of the 2002 PCRSA [118], the 2011 ESC [90], and are internet resources.
- Characteristics of the category ‘C’ human intrusion events cannot readily be compared with similar activities undertaken elsewhere, so parameter values are based on reasonable assumptions.

A summary of exposure pathway parameter values for each intrusion event is given in Table 8.4. For human intrusion events that include multiple occurrences of an action (such as event B1b that models the excavation of twenty trial pits on the site), the calculations assume all occurrences take place within the same grid cell and that the same workers are involved for all excavations. This is a cautious approach because it will give rise to larger maximum doses than if the occurrences take place at different locations on the cap, and more than one worker may be involved for the various excavations.

Table 8.4: Intrusion excavation event and exposure duration parameter values

(a) Short-duration intrusion events or event series

Event Parameter	A1a Rescue worker	A1b Clean-up crew	B1b Trial pit excavator	B1d Borehole worker	B1e Laboratory analyst	B3a Construction worker	C2 Material recovery worker
Intrusion depth, m	21.3 m	21.3 m	6 m	20 m	–	5 m	–
Area of intrusion, m^2	1425 m^2 at surface (conical crater)	1425 m^2 at surface (conical crater)	2.3 m^2	0.025 m^2	–	400 m^2	–
Volume of excavated material per event, m^3	10120 m^3	10120 m^3	13.8 m^3	0.5 m^3	0.03 m^3 (undiluted waste)	2,000 m^3	–
Number of events to which an individual is exposed	1	1	20	5	8 (samples)	1	1
Exposure duration per event (and per series), hr	30 hr over 3 days	8 hr d^{-1} for 30 days (240 hr total)	1 hr per pit (20 hr total) over 3 days	16 hr over two days per borehole (80 hr total)	2 hr per sample (16 hr total)	80 hr over 10 working days	8 hr d^{-1} for 5 days (40 hr total)
Outdoor / indoor	Outdoor	Outdoor	Outdoor	Outdoor	Indoor	Outdoor	Outdoor

(b) prolonged exposure conditions or events

Event Parameter	**B3b** **House occupant**	**B7** **Smallholder**	**C1** **Scavenger**
Intrusion depth, m	5 m	5 m	–
Area of intrusion, m^2	400 m^2	50 m^2	–
Volume of excavated material, m^3	2,000 m^3	225 m^3 (Subsequently mixed with 2,000 m^3 clean soil)	–
Number of events to which an individual is exposed	Continuous	Continuous	Periodic visits
Exposure duration	8766 hr y^{-1}	8766 hr y^{-1}	Scavenging 1 hr d^{-1} for 50 d^{-1} y^{-1} Scavenged item placed in home 1 hr d^{-1} Total 415 hr y^{-1}
Outdoor / indoor proportion	10% / 90%	17% / 83%	12% / 86% *

* For event C1, outdoor exposure while scavenging and indoor exposure to items taken home.

8.5 Assessment Calculation Results

Complete results for the human intrusion calculations are presented in reference [24]. This section presents selected results to provide understanding of the calculated dose impacts and their key features, plus a summary of calculated doses.

For most cases, committed effective doses or annual committed effective doses are calculated as a function of time of event occurrence between 2242 AD and 5775 AD, i.e. the end of active institutional control and the longest timescale for complete erosion of the northern part of the repository in the Low Emissions Scenario. Selected results illustrating the spatial variability of dose are presented at these times. These are calculations of the peak dose per event, or peak annual dose, occurring at the time of the event or sometime shortly afterwards assuming an intrusion occurred at the given location and given time. They are not doses or annual doses that would be sustained over indefinite periods.

8.5.1 Event A1: Aircraft Crash

Event A1 includes doses to the rescue crew who attend the crash, and doses to people who come to clean-up the site once rescue work is complete.

Rescue crew

The impact has sufficient energy to expose waste from anywhere under the cap. Doses are calculated for each grid cell. The calculations assume waste is only exposed from a single grid cell. The largest dose is 0.72 mSv from exposure of waste disposed to a shielded module in Vault 11 (Figure 8.1). Doses are dominated by external irradiation from Cs-137, with inhalation from Pu-239 also being important.

The peak dose from an impact with the shielded module decreases with time due to decay of Cs-137. At around 2300 AD, doses from Cs-137 and Pu-239 are the same, and doses from Pu-239 become increasingly dominant thereafter.

After about 2800 AD, the peak dose is from cell 28T, due to exposure of monazite sands in the trenches. The very long half-life of Th-232 means that the dose from exposure of monazite sands does not decrease with time.

Depending on the direction and angle of impact, waste may be exposed from an area that covers more than one assessment model grid cell. This means that the peak dose would be some average across the impacted grid cells, and the peak dose would be lower than calculated. For example, grid cell 28T covers a small area at the north end of Trench 2 where monazite sands were disposed (Figure 6.6) [62]. It is unlikely an aircraft crash would only impact such a localised area.

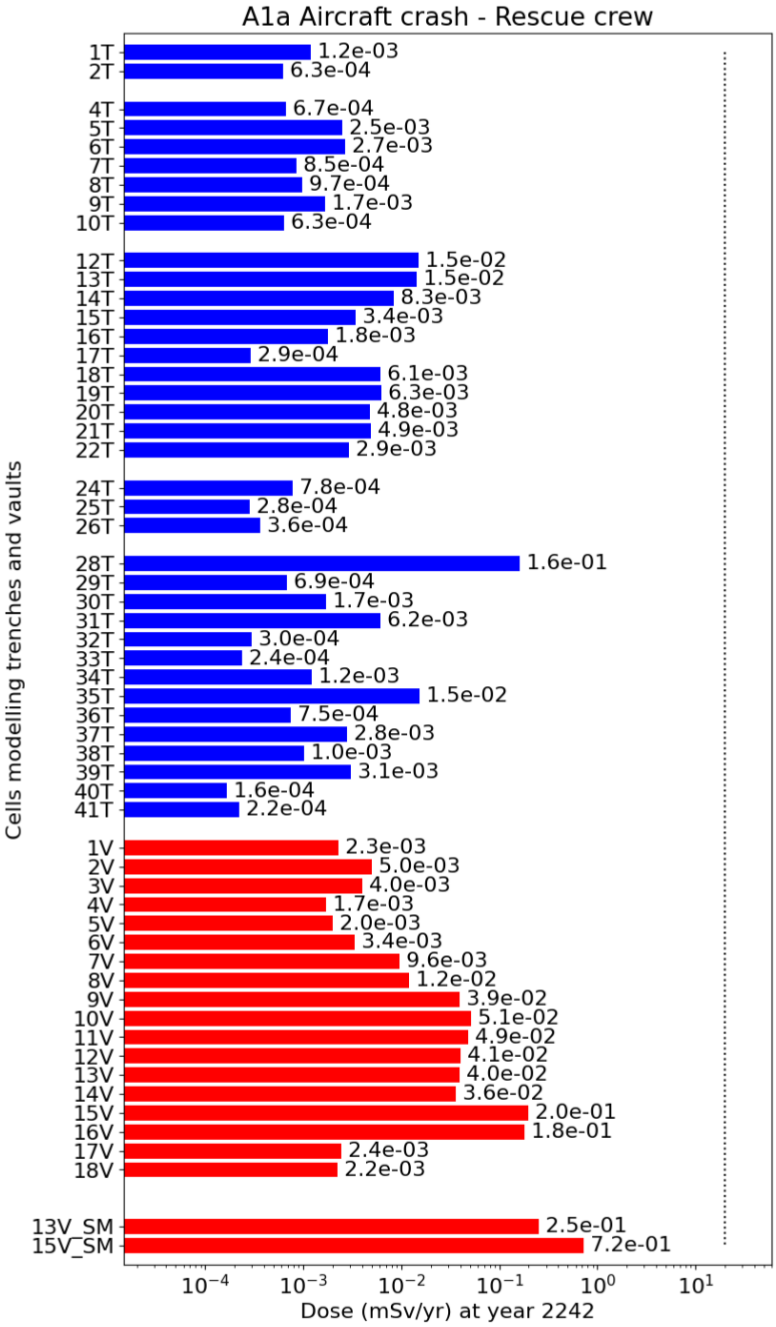

Figure 8.1: **The total dose (in mSv) to the rescue crew (A1a) from an aircraft crash intrusion event at each location in the LLWR in the year 2242 The dotted line indicates the upper end of the dose guidance level range (20 mSv).**

Clean-up crew

Calculated doses show a similar pattern to those calculated for the rescue crew. This is because the source material is the same and the exposure modes are very similar.

The largest dose is 5.8 mSv from exposure of waste disposed to a shielded module in Vault 11 (Figure 8.2) at 2242 AD. Doses are dominated by external irradiation from C-137, with inhalation from Pu-239 also being important. The patterns of peak doses and contributing radionuclides post-2242 are as previously described for the rescue crew.

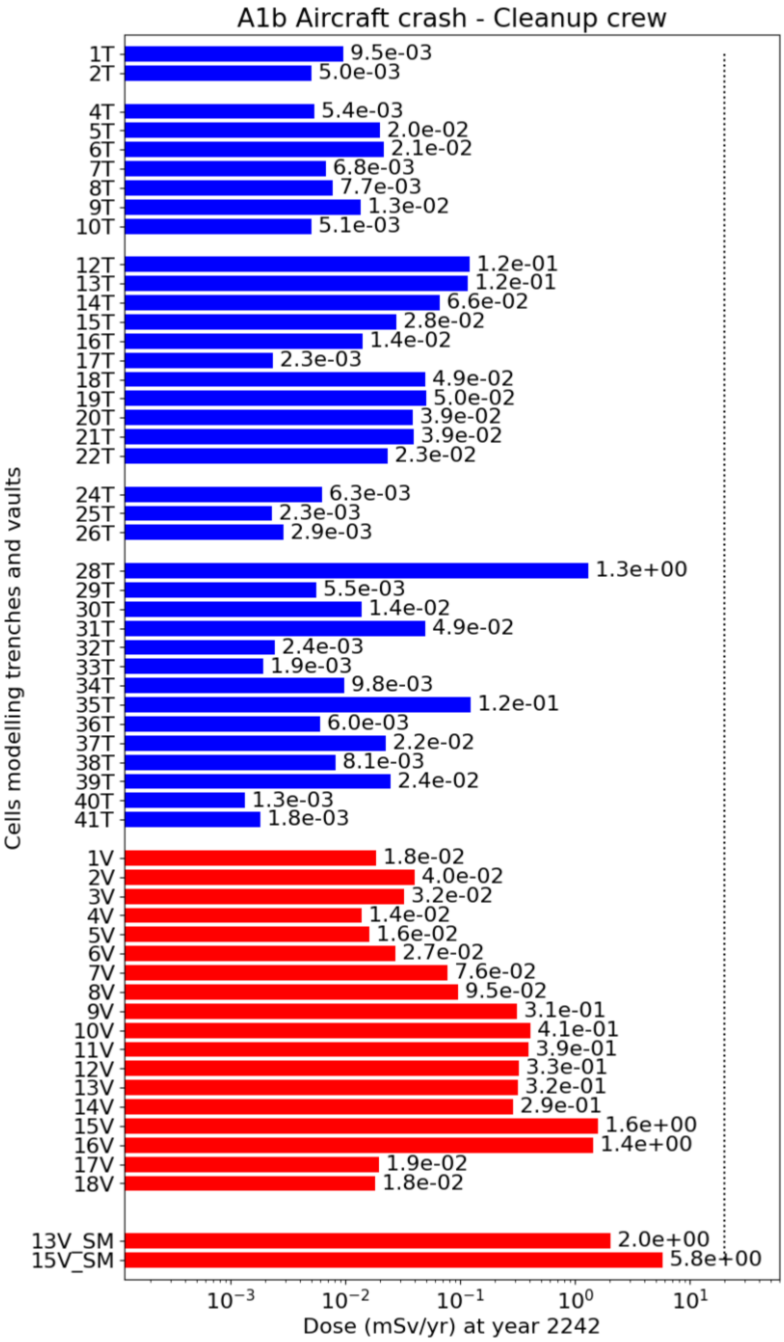

Figure 8.2: **The total dose (in mSv) to the clean-up crew (A1b) from an aircraft crash intrusion event at each location in the LLWR in the year 2242. The dotted line indicates the upper end of the dose guidance level range (20 mSv).**

8.5.2 Event B1: Geotechnical Investigations

Event B1 includes doses to those who take part in geotechnical investigations, trial pits (B1b), borehole drilling (B1d) and laboratory examination of borehole core samples (B1e). These are all doses directly related to the event, or to a series of events, taking place over a few hours per event within a period of a few weeks or less. The human intrusion model calculates the dose per event for a given location within the repository grid, taking account of the amounts of waste involved, the average concentration of radionuclides for that grid cell, and the event exposure parameters.

Trial pit digger

The maximum trial pit depth is 6 m, so wastes are only exposed in some model grid cells. The highest dose is 0.09 mSv from intruding into a shielded module in Vault 11 at 2242, i.e. immediately following the end of the PoA. This dose is dominated by Pu-239 and the inhalation pathway. The results cautiously assume that all twenty trial pit intrusions are carried out by the same person and into the shielded module. It is questionable whether a trial pit could expose waste in a shielded module given their robustness and expected durability [13] [66]. Also, given the expected dimensions of the individual shielded modules (the shielded modules in a vault are amalgamated into a single module for assessment calculations), it is very unlikely that 20 trial pits would be dug into the one module, or that all 20 pits would intersect modules.

The highest calculated dose for trial pits that do not excavate waste from a shielded module is 0.04 mSv from digging into Vault 11. If the trial pits are dug at different locations on the cap, then the expected combined dose is less than 0.09 mSv, and likely less than 0.04 mSv, especially as the trial pits would not intersect waste in many locations.

It should be noted that the doses are calculated from the inventory averaged over the grid cell volume, and in the case of shielded modules, the module volume. If a particular region of high activity is dug out in the trial pit sample, then the dose to the trial pit digger may be larger than calculated. This is particularly relevant for Vault 11 which could contain areas of ILW bearing higher concentrations of Pu-239, more similar to the shielded modules, and areas of LLW bearing lower concentrations of Pu-239. However, it is unlikely that the peak dose would be much greater than 0.09 mSv, and it is not expected that it would approach 20 mSv.

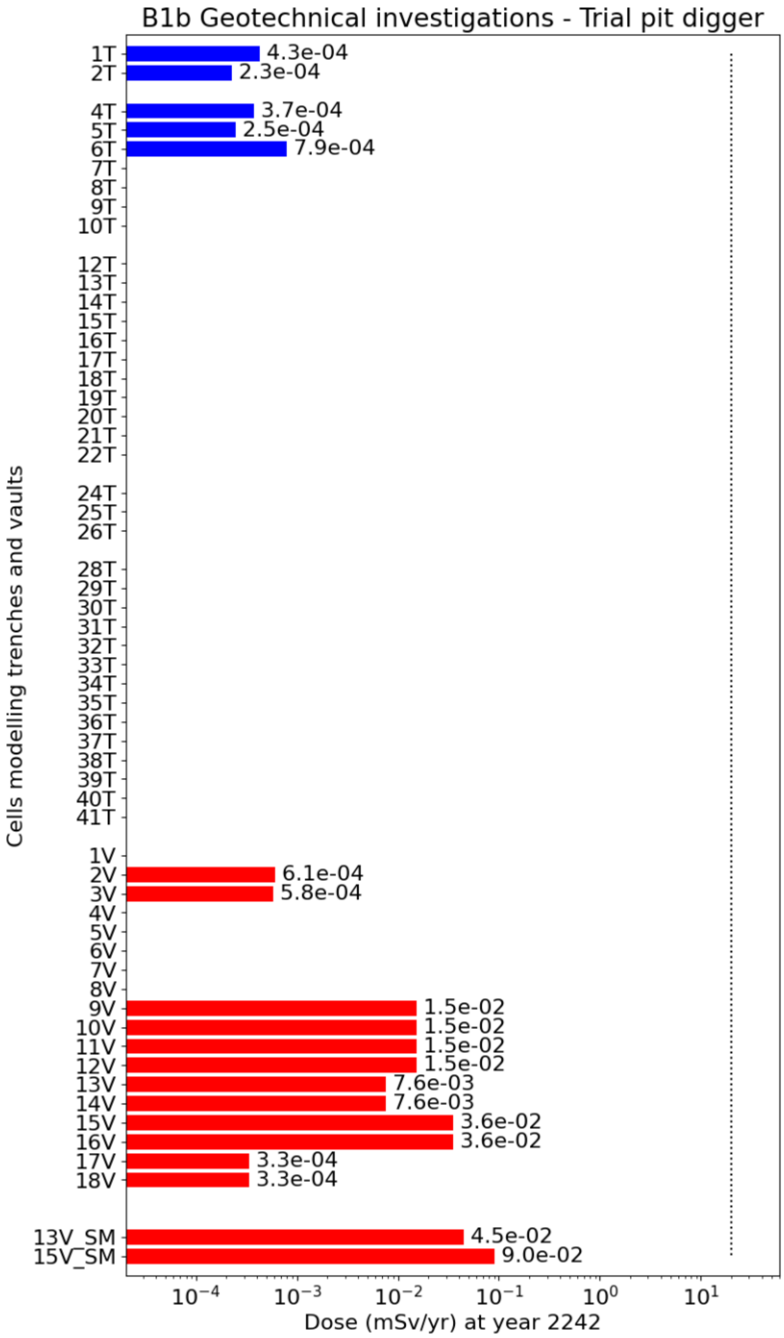

Figure 8.3: The total dose (mSv) to the digger (B1b) from a digging intrusion event at each location in the LLWR in the year 2242. The dotted line indicates the upper end of the dose guidance range (20 mSv).

Borehole worker

The peak dose to a borehole worker is also from drilling into shielded modules in Vault 11. The peak dose of 1.1 mSv occurs at 2242 AD, with similar contributions from external irradiation (dominated by Cs-137) and inhalation (dominated by Pu-239). At later times the total dose decreases and becomes dominated by inhalation of Pu-239 as Cs-137 decays.

The calculated peak dose assumes 5 boreholes are drilled into a shielded module. This is unlikely as the shielded modules are small compared with the size of the vaults. The peak dose from drilling five boreholes into the other wastes in Vault 11 (cells 15V and 16V) is around 0.3 mSv (Figure 8.4). The peak dose from drilling five boreholes spaced out across the repository would likely be less than 0.3 mSv.

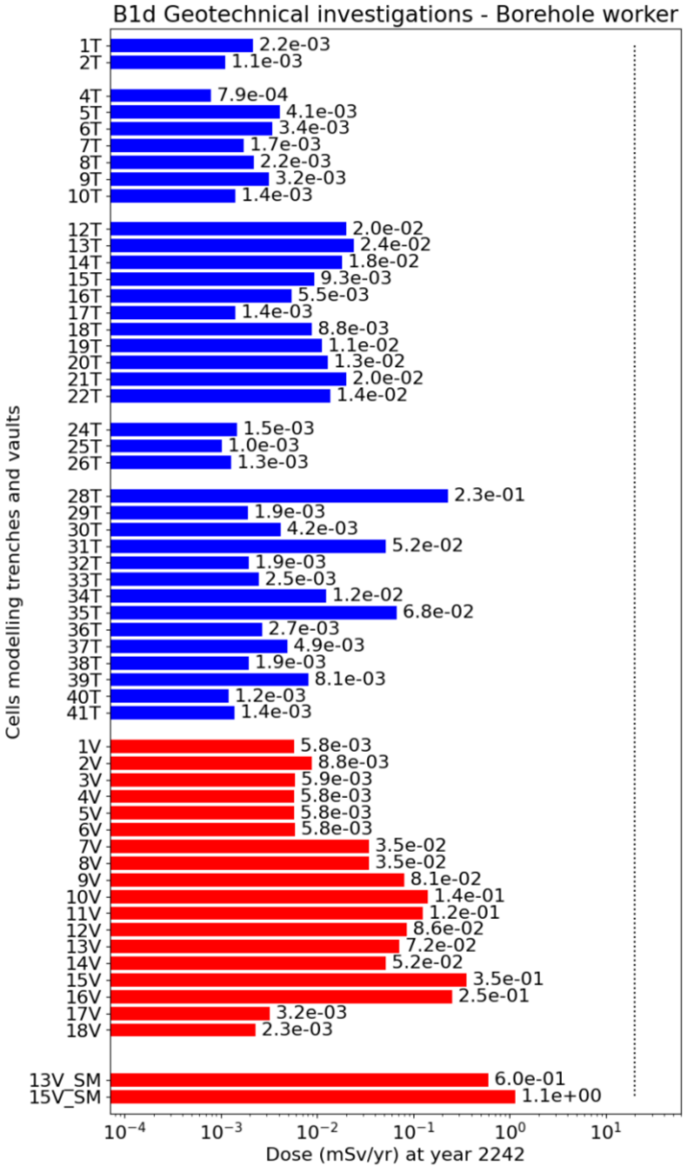

Figure 8.4: The total dose (mSv) to the borehole worker (B1d) from a drilling intrusion event at each location in the LLWR in the year 2242. The dotted line indicates the upper end of the dose guidance range (20 mSv).

Again, these doses are based on average radionuclide concentrations in each cell, so the dose from drilling five boreholes into Vault 11 could be higher than 0.3 mSv if some boreholes intersected shielded modules or more hazardous disposals of LLW or ILW that can be managed as LLW. It is unlikely that the peak dose would be much greater than 1.1 mSv, and it is not expected that it would approach 20 mSv.

Laboratory analyst

The peak dose to the laboratory analyst is 3.7 mSv from exposure to thoron gas generated by monazite sands in the trench waste (cell 28T, Figure 8.5). The peak dose is at 5775 AD but it does not change significantly during the assessment timeframe.

The results cautiously assume that all eight borehole samples originate from the same location (grid cell). It is unlikely all eight boreholes would be drilled into cell 28T, which has a small area. If the borehole samples originate from different locations on the site, then the expected combined dose will be less than 3.7 mSv. Again, it should be noted that the dose is calculated from the inventory averaged over the cell volume. If a particular region of high activity is present in the borehole sample, then the dose to the analyst may be large, but it is unlikely that the dose guidance range will be exceeded.

The peak doses for the vault waste are from analysing boreholes samples drilled from the shielded modules in Vault 11. The peak dose is 0.11 mSv. It is unlikely that all the borehole samples would originate from shielded modules in Vault 11.

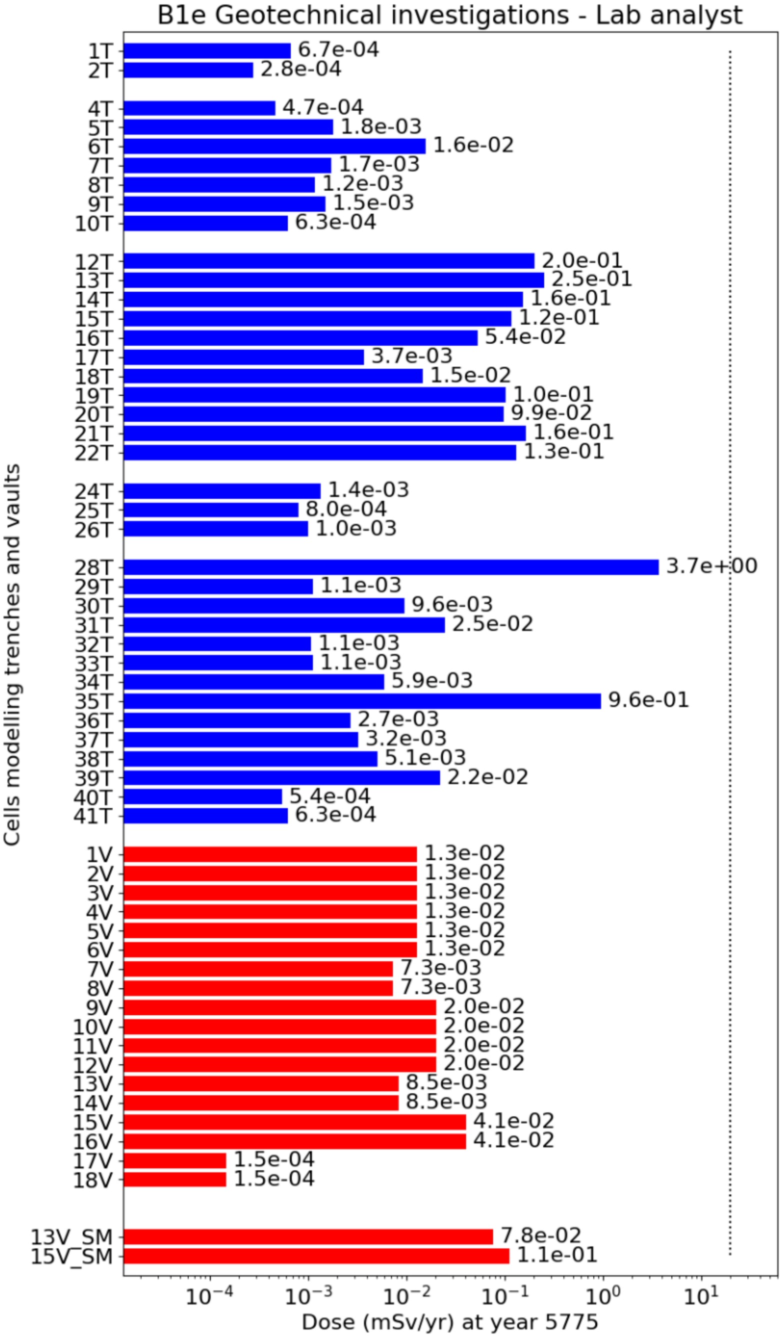

Figure 8.5: The total dose (mSv) to the analyst (B1e) from a borehole intrusion event at each location in the LLWR in the year 5775. The dotted line indicates the upper end of the dose guidance range (20 mSv).

8.5.3 Event B3: Housing Development

Event B3 includes doses to construction workers (builders) (B3a), which are doses for the event occurring over a matter of weeks, and doses to those who later occupy the houses on the site (B3b), in which case the exposure may be prolonged and annual doses are calculated. House foundations would not usually be deep enough to penetrate into the waste, but the housing development is assumed to include excavations to 5 m deep with the excavated material distributed about the housing development site, including building on the excavated material. 5 m deep excavation is sufficient to expose some waste around the periphery of the cap, and future vault LLW and ILW in new strong containers and boxes that can be stacked higher than current containers (assuming there is some erosion and thinning of the cap and ignoring the presence of the CPUs). ILW in shielded modules is not exposed.

Builder

The largest dose is 0.08 mSv at cells 15V and 16V (Vault 11) at the year 2242 (Figure 8.6). The contributing pathways are like event B1 with doses at early times mainly from external irradiation (Cs-137), and later from both the external irradiation (Nb-94) and inhalation (Pu-239) pathways.

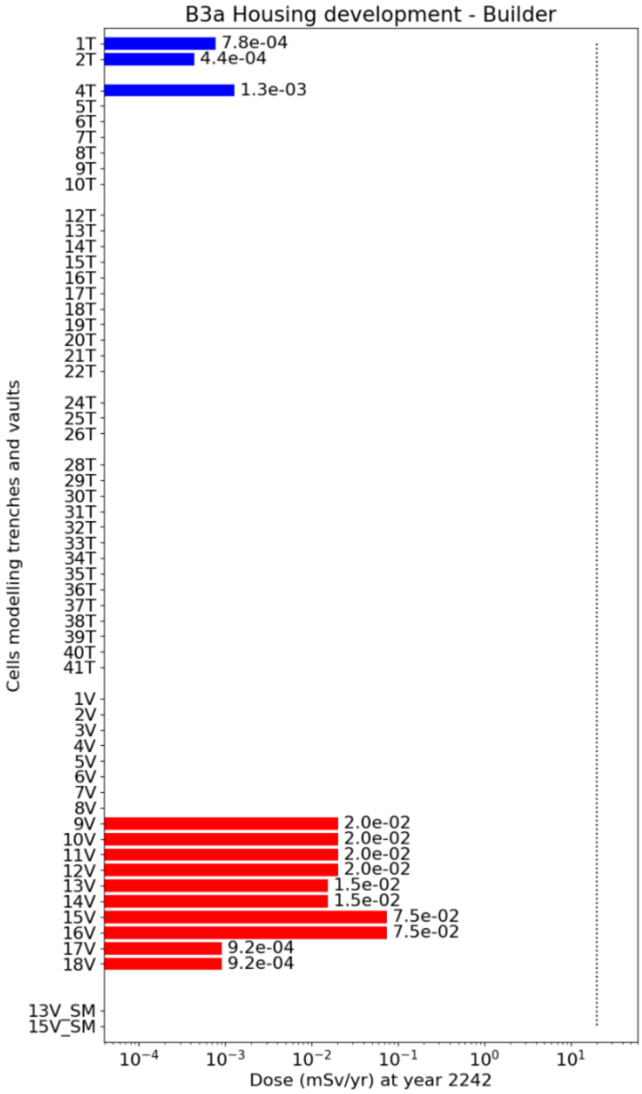

Figure 8.6: The total dose (in mSv) to the builder (B3a) from a construction intrusion event at each location in the LLWR in the year 2242. The dotted line indicates the upper end of the dose guidance range (20 mSv).

House occupier

The largest dose is 2.1 mSv from cells 15V and 16V (Vault 11) at the year 2242 (Figure 8.7). At early times, the dose is dominated by external irradiation (from Cs-137). By 2400 AD the total dose has decreased by over an order of magnitude due to decay of Cs-137. Most of the Cs-137 inventory is in ILW.

Post-2400 AD doses from cells 15V and 16V are mainly from inhalation of Pu-239, with a secondary contribution from external irradiation by Nb-94.

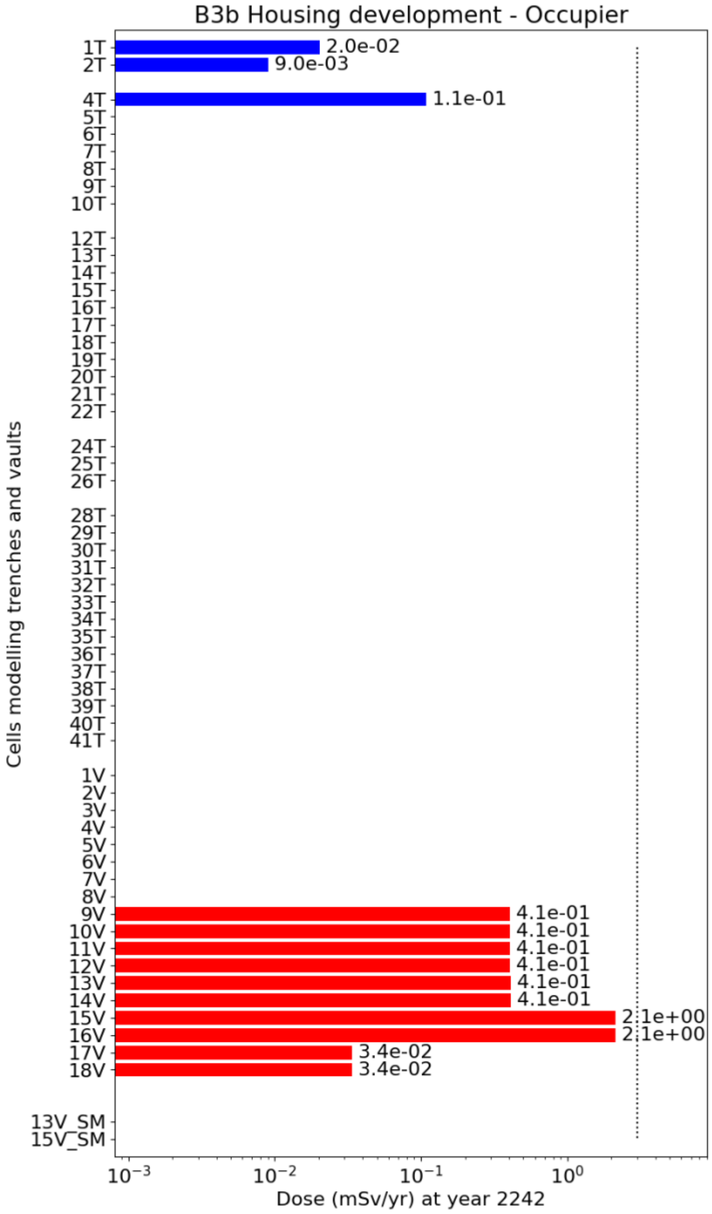

Figure 8.7: **The total dose (in mSv) to the house occupier (B3b) from a construction intrusion event at each location in the LLWR in the year 2242. The dotted line indicates the lower end of the dose guidance range (3 mSv).**

8.5.4 Event B7: Smallholding

The depth of intrusion is the same as the housing development event discussed above, i.e. 5 m. Therefore, the areas (model grid cells) where waste could be exposed are the same.

The largest dose is 1.8 mSv from cells 15V and 16V (Vault 11) at the year 2242 (Figure 8.8). At early times, the dose is dominated by external irradiation (Cs-137). By 2400 AD the total dose has decreased by around an order of magnitude due to decay of Cs-137. Most of the Cs-137 inventory is in ILW.

Post-2400 AD doses are mainly from ingestion of foodstuffs grown on the smallholding. Cl-36 is the key radionuclide for doses from foodstuffs. It is present in crops and goat milk. Although the peak dose to the house occupier is greater than to the smallholder, the smallholder dose is higher post-2400 AD.

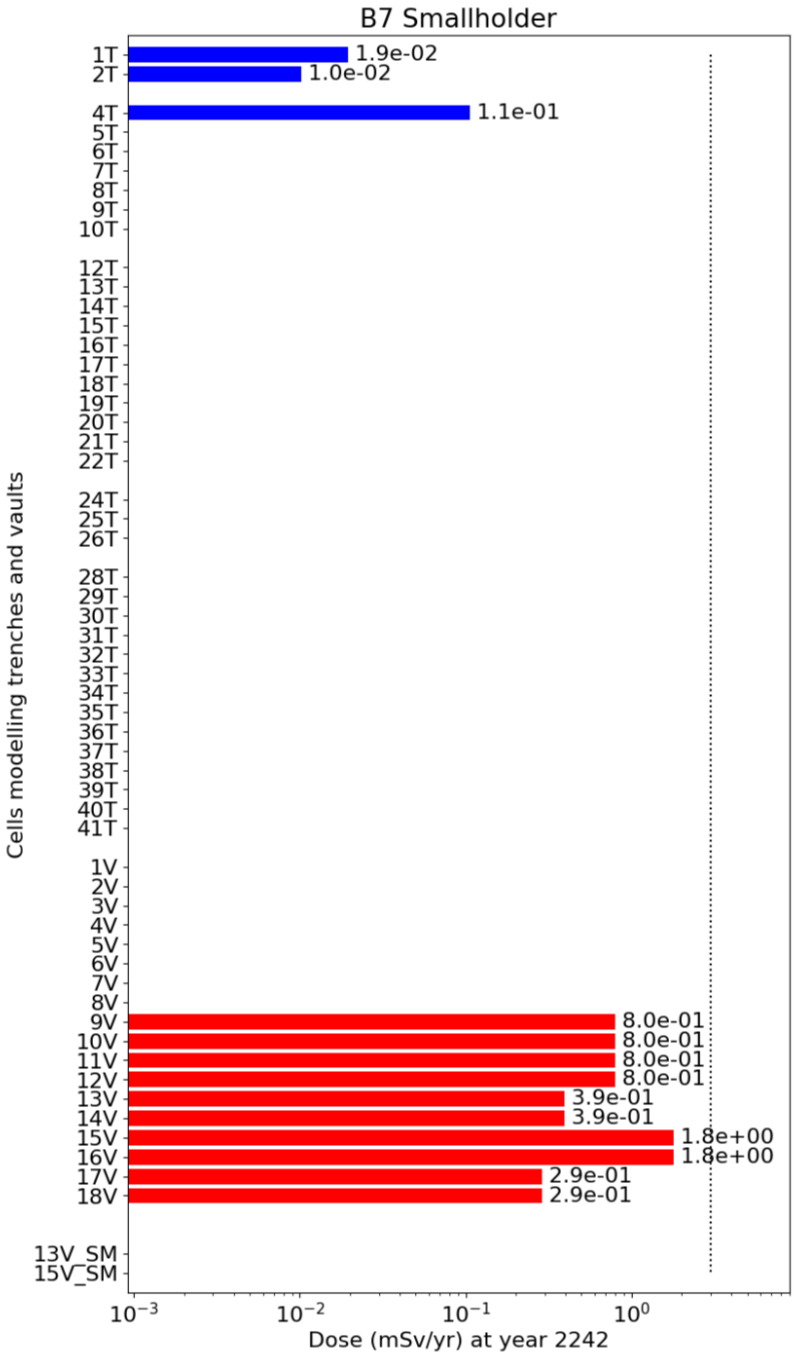

Figure 8.8: **The total dose (in mSv) to the smallholder (B7) from a construction intrusion event at each location in the LLWR in the year 2242. The dotted line indicates the lower end of the dose guidance range (3 mSv).**

We note that this case is cautious for the vaults, as the excavated material would include container metal, grout, waste metals and other materials that could not be incorporated into productive soil. Nevertheless, for the assessment calculations, the total inventory of the excavated waste is assumed to be incorporated into the soil of the smallholding.

8.5.5 Event C1: Informal Scavenging

All the waste in the repository is assumed to be exposed by coastal erosion. In the assessment model doses are calculated for all grid cells at all times between 2400 AD and 6000 AD, although the potential for a dose related to any individual grid cell exists only during the time period in which that cell is being eroded and the waste is accessible.

The times for start of the disruption of the repository are:

- around 2775 AD in the High Emissions Scenario;
- around 3275 AD in the Reference Emissions Scenario;
- around 3775 AD in the Low Emissions Scenario.

Doses in many grid cells, including the cells with the highest doses, are constant with time. The doses are dominated by very long-lived radionuclides.

The peak dose at all times is 1.8 mSv from grid cell 28T (Figure 8.9). Doses are dominated by external irradiation from monazite sands. It is unlikely this waste would be attractive for informal scavenging, so the probability of this event is very low (large quantities of better-quality sand could be taken direct from the beach and foreshore, and the monazite sands do not have any other qualities which make them attractive). Excluding informal scavenging of monazite sands, the next highest doses are from informal scavenging from the shielded modules, at around 0.5 mSv.

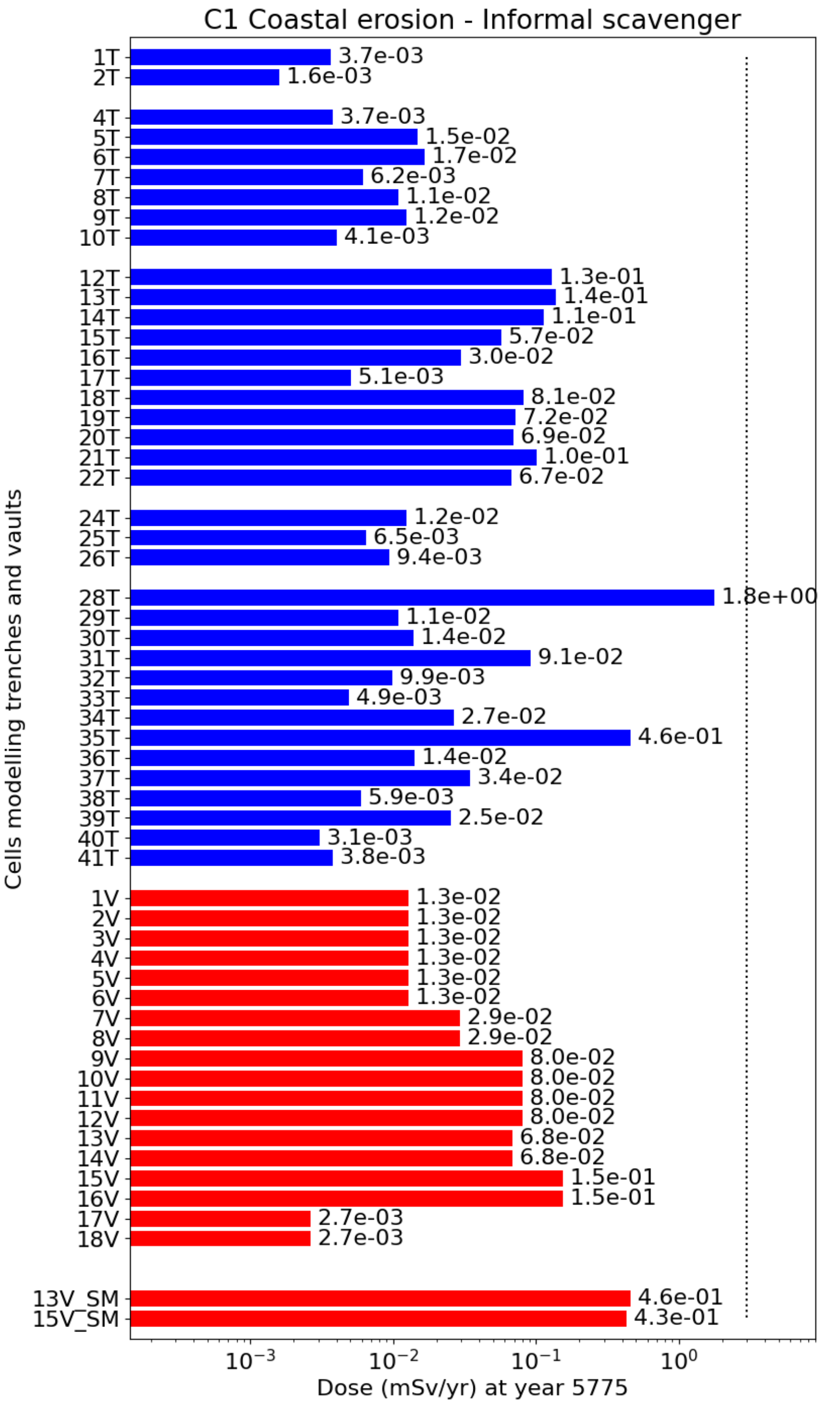

Figure 8.9: **The total dose (in mSv) to the informal scavenger (C1) from a coastal erosion intrusion event at each location in the LLWR in the year 5775. The dotted line indicates the lower end of the dose guidance range (3 mSv).**

8.5.6 Event C2a: Organised Scavenger

Calculated doses show a similar pattern to Event C1. This is because the source material is the same and the exposure modes are very similar.

The peak dose at all times is 1.4 mSv from grid cell 28T (Figure 8.10). Again, doses are dominated by external irradiation from monazite sands. It is unlikely these wastes would be selected for organised scavenging unless their nature was known, which would not be an inadvertent intrusion event. Again, the next highest doses are from organised scavenging from the shielded modules, at around 0.4 mSv.

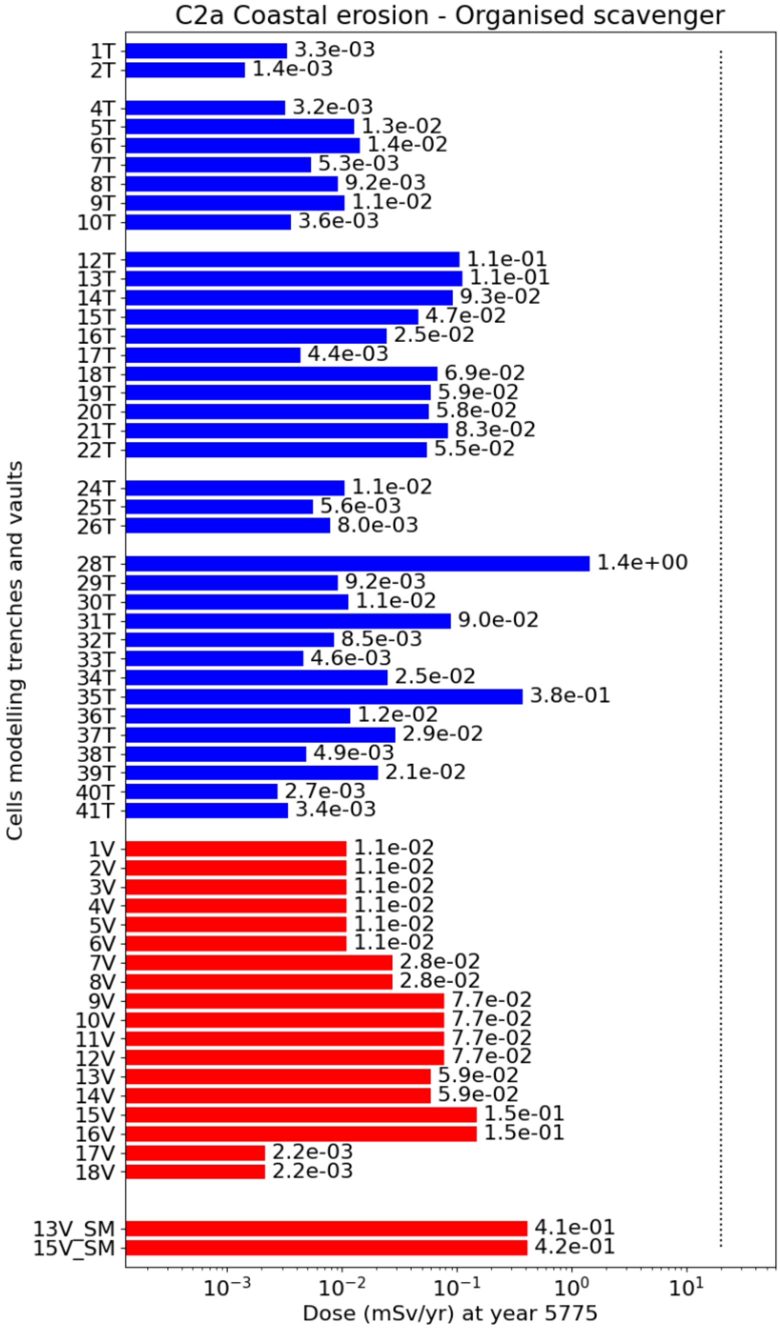

Figure 8.10: The total dose (in mSv) to the organised scavenger (C2a) from a coastal erosion intrusion event at each location in the LLWR in the year 5775. The dotted line indicates the lower end of the dose guidance range (20 mSv).

8.5.7 Doses from Radon in a House with a Basement

The peak dose to a house with a basement is 1.3 mSv above future disposals to Vault 9 (cell 10V) (Figure 8.11). The thickness of profile material over cell 10V is at the minimum value of 2 m, so calculated doses are higher than other locations which have higher concentrations of Ra-226 but are covered with thicker profile material.

Calculated doses at all locations slowly decrease with time due to decay of Ra-226. Consistent with the results of the radon gas pathway assessment (Section 6) ingrowth of Ra-226 is not significant over the assessment timeframe.

The calculated doses are likely to be overestimates for three reasons.

- Decay of radon as it migrates up through the trench wastes and the gaps between vault waste stacks is not included in the calculations. There is around an order of magnitude difference in the amount of decay for a bottom of stack container compared with a top of stack container [26].
- Radon is a dense gas and will tend to sink, especially in the vaults where the bulk gas generation rate is low, so advection of radon by bulk gas is small. The high density of radon gas and tendency to sink is not included in the calculations.
- The gas assessment [26] notes that most of the Ra-226 in the future Vault 9 inventory is in sealed sources. The fraction of radon that emanates from these sources may be lower than the fraction assumed in the assessment calculations.

The available information indicates some, maybe all, of the Ra-226 sealed sources in the forward inventory would not meet the current WAC. Therefore, they might not be accepted for disposal without a bespoke assessment, in which case the highest calculated annual dose would likely come from existing disposals.

The highest annual doses from existing disposals are in the range ~0.2 mSv to ~1.1 mSv: cell 4T 0.24 mSv, cell 6T 0.38 mSv, cell 2V 1.1mSv, cell 3V 0.70 mSv, cell 6V 0.29 mSv, cell 8V 0.22 mSv. However, these calculated doses also exclude decay of radon while it migrates up through the waste to the base of the profile fill, and density effects, so they are also expected to be too high.

If Ra-226 sources are accepted for disposal in Vault 9, calculated doses could be reduced by up to an order of magnitude by placing these wastes in bottom of stack positions [26].

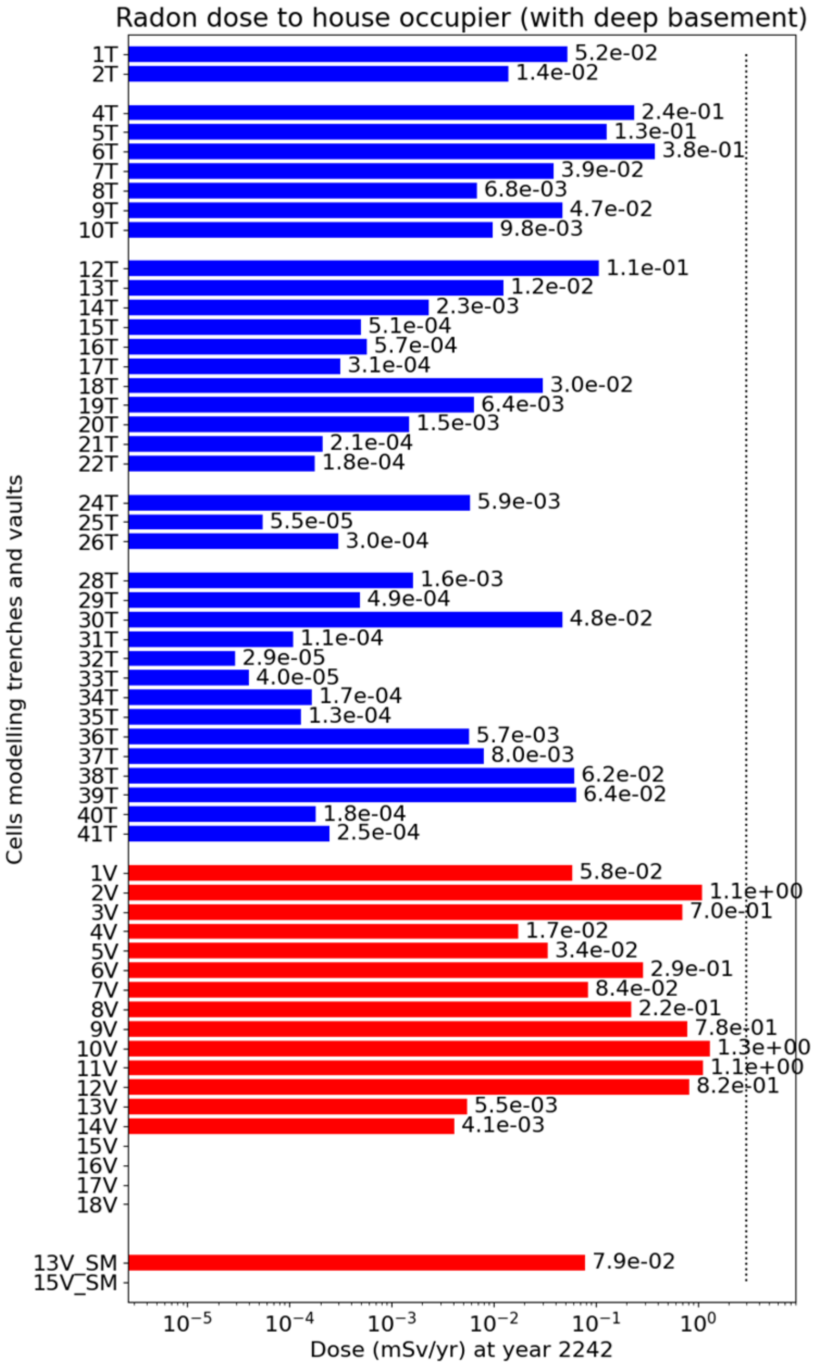

Figure 8.11: The total radon dose (in mSv) to the occupier at all grid cells where radon is produced at the year 2242 in the LLWR. The dotted line indicates the lower end of the dose guidance range (3 mSv).

8.5.8 Doses from LLW

We have undertaken variant calculations assuming only LLW is disposed in the future, and ILW is not disposed. Calculated doses from the trenches and existing vault disposals are as described above. Only the calculated doses from future disposals to the vaults change. The results are summarised in the next subsection where they are compared with the summarised doses from LLW and ILW.

8.6 Summary of Results and Assessment

The GRA [19] requires that human intrusion into a near-surface disposal facility after the end of management control of the site is to be assessed on the basis that it occurs, and against a dose guidance level in the range of around 3 mSv for exposures continuing over a period of years to around 20 mSv for exposures that are of limited duration.

We have identified a range of possible human intrusion events and more prolonged exposure situations, assuming the repository is no longer protected under planning procedures and, or the presence and nature of the disposal facility is forgotten. We have assessed these qualitatively and selected for quantitative assessment those that we consider are representative and have potential to cause radiological exposures. This includes events that occur while the repository is still intact and scavenging of materials when the facility is eroding and wastes are more directly accessible at the coast.

We have characterised the selected events and exposure situations based on geotechnical practice and cautious assumptions concerning the fate of excavated material and subsequent use of any contaminated environment created. We have also examined the impact of heterogeneity of radioactivity in past and projected waste disposals for sub areas of the trenches and vaults which container the highest concentrations of the key radionuclides for potential impacts from human intrusion.

Overall, we have quantitatively assessed a reasonable and robust set of events that are sufficient to:

- explore the key uncertainties around potential future impacts;
- guide our optimisation of protection against human intrusion;
- yield dose estimates that are appropriate for comparison to the dose guidance levels;
- develop radionuclide concentration limits for inclusion in our WAC.

The potential impacts of intrusion events intercepting 'hotspots' of activity or people being exposed to particles, sealed sources or discrete items that attract attention are assessed in Section 10.

None of the events lead to whole body effective doses above 100 mSv, where deterministic effects could occur [119]. We have also calculated doses to individual organs and confirmed doses are below the levels that could lead to deterministic effects. The calculations are presented in reference [24].

Summaries of calculated effective doses for limited-duration intrusion events, and annual effective doses for post-intrusion situations and prolonged intrusion activities, are shown in Figure 8.5 and Table 8.6, respectively.

All transitory exposure events are below the relevant regulatory dose guidance level (20 mSv) for each grid cell. The highest doses from existing and potential future disposals are to the clean-up crew following an aircraft crash into a shielded module in Vault 11 (5.8 mSv) and from analysing borehole core samples from monazite sand waste in the trenches (3.7 mSv). Although the probabilities of an aircraft crash and intruding into the specific area where monazite sands have been disposed cannot be quantified, they are both very low.

All prolonged exposure events (the occupiers and the informal scavenger) have a maximum dose below the relevant regulatory dose guidance level (3 mSv) for each grid cell. The highest calculated annual dose is 2.1 mSv from occupying a building constructed on contaminated soils containing waste from cells 15V and 16V, i.e. Vault 11. The thickness of the cap and profile fill over the most hazardous trench wastes, including the monazite sands, is too great for these wastes to be exposed by the intrusion events leading to a contaminated site.

The peak annual dose from informal scavenging is 1.8 mSv, again from the monazite sands. It is unlikely this waste would be attractive for informal scavenging.

The highest annual dose from radon in a house with a basement is 1.3 mSv above future disposals in Vault 9 (grid cell 10V). The gas assessment [26] notes that most of the Ra-226 in the future Vault 9 inventory is in Ra-226 sealed sources. The available information indicates some, maybe all these sealed sources would not meet the current WAC. Therefore, they might not be accepted for disposal, and the peak calculated annual dose is too high.

If these Ra-226 sources are accepted for disposal, calculated doses could be reduced by up to an order of magnitude by placing these wastes in bottom of stack positions [26]. Cautiously, the calculations take no credit for the decay of radon while it migrates up the gaps between vault waste stacks, or the propensity of radon gas to sink due to its high density.

Calculated peak doses assuming only LLW is disposed in the future are summarised in Table 8.7 and Table 8.8. For events where the peak dose is from future disposals, the peak dose is reduced by a factor of around four to nine by only disposing LLW.

Table 8.5: Summary of doses for limited-duration intrusion events

	A1a **Aircraft crash** **rescue crew**	**A1b** **Aircraft crash clean-up crew**	**B1b** **Trial pit excavator** **(20 trial pits)**	**B1d** **Borehole worker** **(5 boreholes)**	**B1e** **Borehole core analysis** **(8 samples)**	**B3a** **Building construction**	**C2a** **Organised scavenger**
Maximum dose (mSv)	0.72	5.8	0.09	1.1	3.7	0.08	1.4
Location (cells)	15V_SM (Vault 11 shielded module)	15V_SM (Vault 11 shielded module)	15V_SM (Vault 11 shielded module)	15V_SM (Vault 11 shielded module)	28T (North end of Trench 2)	15V, 16V (Vault 11)	28T (North end of Trench 2)
Inventory status	Future	Future	Future	Future	Existing	Future	Existing
Time of maximum (year AD)	2242	2242	2242	2242	5775	2242	5775

Table 8.6: Summary of annual doses for post-intrusion situations and prolonged intrusion activities

	B3b **Building occupation**	**B7** **Smallholding**	**C1** **Informal scavenger**	**House with a basement**
Maximum annual dose (mSv)	2.1	1.8	1.8	1.3
Location (cells)	15V, 16V (Vault 11)	15V, 16V (Vault 11)	28T (North end of Trench 2)	10V (Vault 9 Future)
Inventory status	Future	Future	Existing	Future
Time of maximum (year AD)	2242	2242	5775	2242

Table 8.7: Summary of doses for limited-duration intrusion events - LLW only, changes compared with Table 8.5 highlighted

	A1a Aircraft crash rescue crew	**A1b** Aircraft crash clean-up crew	**B1b** Trial pit excavator (20 trial pits)	**B1d** Borehole worker (5 boreholes)	**B1e** Borehole core analysis (8 samples)	**B3a** Building construction	**C2a** Organised scavenger
Maximum dose (mSv)	**0.16**	**1.3**	**0.01**	**0.23**	3.7	**0.01**	1.4
Location (cells)	**28T**	**28T**	**9V, 12V**	**28T**	28T (North end of Trench 2)	**9V, 12V**	28T (North end of Trench 2)
Inventory status	**Existing**	**Existing**	Future	**Existing**	Existing	Future	Existing
Time of maximum (year AD)	**5775**	**5775**	2242	**5775**	5775	2242	5775

Table 8.8: Summary of annual doses for post-intrusion situations and prolonged intrusion activities - LLW only, changed compared with Table 8.6 highlighted

	B3b **Building occupation**	**B7** **Smallholding**	**C1** **Informal scavenger**
Maximum annual dose (mSv)	**0.4**	**0.54**	1.8
Location (cells)	**9V, 12V**	**9V, 12V**	28T (North end of Trench 2)
Inventory status	Future	Future	Existing
Time of maximum (year AD)	2242	2242	5775

8.7 Consequences of Human Intrusion for Repository Evolution and Performance

Human intrusion into the repository will damage the final cap, which is a most important barrier to radionuclide release [12] [13]. We have considered how this damage could affect long-term evolution and performance of the repository [120]. Our approach is to:

- identify the consequences of intrusion events for repository evolution and performance;
- use the results of our pathway assessments to understand the potential implications of changes to repository evolution and performance for doses to people.

Consistent with the requirements of the GRA [19], potential doses to people occupying the site following intrusion induced damage to the cap are compared against the dose guidance level.

Only deeper intrusion events that would penetrate the low permeability layers of the cap have the potential to significantly change the long-term evolution and performance of the repository. These events are relatively unlikely compared with shallower intrusion events that would little affect long-term evolution and performance. The probability of these deeper intrusion events is expected to be low but cannot be quantified. The potential for multiple deeper intrusion events over the assessment timeframe is even more uncertain. Therefore, our approach is to show that one or a small number of deeper intrusion events are unlikely to have effects that result in doses which exceed the dose guidance level.

8.7.1 Consequences for Repository Evolution and Performance

Only deeper intrusion events that would penetrate the low permeability layers of the cap have the potential to significantly change the long-term evolution and performance of the repository. The implications of damage to the cap would depend on:

- the location(s) of the intrusion event(s);
- the depth(s) of the intrusion event(s);
- the size(s) of the intrusion event(s);
- the timing(s) of the intrusion event(s).

The location affects:

- the depth of the waste below the cap surface, and whether the cap and profile are fully breached;
- the type of waste below the damaged area;
- the upslope area and the volume of rainwater runoff that might flow into the damaged area.

The potential consequences of intrusion events for repository evolution and performance are summarised in Table 8.9. All the events would lead to localised deviations in repository evolution and performance. However, none of the events would damage a substantial proportion of the cap. The total cap area is around 27 ha [26]. The largest area of damage is associated with an aircraft impact, and this area is only around 0.5% of the total cap area.

Table 8.9: Consequences of intrusion events for repository evolution and performance

Event	Area of intrusion	Depth of intrusion	Consequences	Trenches	Vaults
House with a basement	75 m^2	~2.5 m Into cap gas collection layer	Repository evolution	Local increase in infiltration and possibly also ingress of oxygen through backfill around the basement walls. Negligible impact on water levels and flows at the trench scale. If oxygen reaches the waste, accelerated degradation of remaining organic waste, accelerated corrosion of metals, and less reducing below the house.	Enhanced infiltration may result in locally increased leaching of grout, lowering the pH to around 10.5 to 11. If oxygen reaches the waste, faster corrosion rates possibly leading to faster container settlement, and less reducing below the house.
			Releases in water	Locally enhanced leaching of radionuclides from the waste. If conditions are less reducing, redox sensitive radionuclides might be more mobile, i.e. Tc-99 and isotopes of U, but only a very small volume of waste affected.	As trenches plus potentially locally increased releases of C-14 from the matrix of corroding metals.

<table>
<tr><th>Event</th><th>Area of intrusion</th><th>Depth of intrusion</th><th>Consequences</th><th>Trenches</th><th>Vaults</th></tr>
<tr><td></td><td></td><td></td><td>Releases in gas</td><td>Potential for radon ingress into the house. Backfill may also provide a conduit for release of bulk gas and trace radioactive gases analogous to leaving the cap vent open. Backfill would capture gas from a smaller area than the cap vent.</td><td>As trenches plus potentially locally increased releases of C-14 from the matrix of corroding metals, and locally increased microbial activity generating C-14 gas. A portion of any additional C-14 gas might enter a kitchen garden next to the house, if present.</td></tr>
<tr><td rowspan="3">Septic tanks or sewage tanks associated with a house or houses</td><td rowspan="3">Septic tank
50 m^2
Sewage tanks
400 m^2</td><td rowspan="3">5 m
Through the cap resistive layers everywhere.
Into the top of waste where the profile material is at the 2 m minimum.</td><td>Repository evolution</td><td colspan="2" rowspan="2">Consequences like a house with a basement. Tanks will be barriers to infiltration of water and release of gas. There may be locally increased infiltration through backfill around the tank(s) and possibly also oxygen ingress. If the excavation(s) intersect the waste, the potential for oxygen ingress into the waste is increased compared with a basement excavation that terminates in the cap gas collection layer.</td></tr>
<tr><td>Releases in water</td></tr>
<tr><td>Releases in gas</td><td colspan="2">Enhanced releases of bulk and trace radioactive gases would have little impact as there would not be a house (or houses) over the tank(s), and it is unlikely the tank(s) would be located in a kitchen garden. Possibly enhanced releases of C-14 gas to a smallholding grazing area.</td></tr>
</table>

Event	Area of intrusion	Depth of intrusion	Consequences	Trenches	Vaults
Deep trial pits	2.3 m^2 each Total area of a campaign of 20 pits = 46 m^2	6 m Through the cap resistive layers everywhere. Into the top of waste where the profile material is around the 2 m minimum.	Repository evolution	Backfilled trial pits could provide conduits for locally increased infiltration into the waste and oxygen ingress. The implications for local leaching, pH, Eh, corrosion rates and radionuclide releases would be as described above for a septic or sewage tank. It is unlikely all the trial pits would be in one area. The magnitude of the local impacts associated with a single deep trial pit would be similar to or lower than the magnitude of the impacts associated with a single tank.	
			Releases in water		
			Releases in gas	Potentially a house or a smallholding could subsequently be built over a trial pit (or pits). The backfilled trial pits would then provide conduits for migration of radon and C-14 gas. This would be analogous to a house or smallholding over an open cap vent, although trial pit would capture gas from a smaller area than an open cap vent.	
Aircraft impact	1425 m^2 This is around 0.5% of the cap area	21.3 m Into waste or through full thickness of the repository, depending on location	Repository evolution	Consequences similar to the events described above but more extreme and affecting a larger volume of waste. However, the volume of waste affected would still be a small proportion of the total. If the impact site is not backfilled, waste would be exposed at the surface and therefore to rain and oxygen. In addition to direct precipitation, the crater could capture cap runoff from upslope. There might be sufficient infiltration to locally affect water levels in part of a trench, or a whole vault. However, the increased infiltration is not expected to result in water levels rising to the cap surface, forming a pond that	
			Releases in water		
			Releases in gas		

Event	Area of intrusion	Depth of intrusion	Consequences	Trenches	Vaults
				overspills onto the surrounding cap soils. The crater could be a significant conduit for release of gases. If the crater is backfilled a house or smallholding might be constructed over the backfill. This could be analogous to a house or smallholding over an open cap vent, depending on the nature of the backfill material.	
Borehole into waste Assume the borehole is not backfilled or sealed. If unlined, the sediments are likely to collapse into the hole, so it forms a linear higher permeability pathway. If a piezometer is installed, eventual silting / collapse of the slotted / screened section results in a similar situation.	$0.025\ m^2$ Total area of a campaign of 5 boreholes in one area = $0.125\ m^2$	Around 5 m to 18 m depending on location	Repository evolution	Local increase in infiltration but negligible impact on water levels or flows at the trench scale. Boreholes could provide a conduit for release of gases but are unlikely to be conduits for significant oxygen ingress. No significant impact on geochemical conditions.	Borehole likely to penetrate several containers. Potential for additional leaching of grout from these containers, decreasing pH and resulting in faster corrosion and possibly some microbial activity. <0.1% of containers in the repository affected.
			Releases in water	Locally enhanced leaching of radionuclides from a small proportion of the trench waste.	Enhanced leaching of radionuclides from the containers breached by the borehole. <0.1% of containers in the repository affected.

Event	Area of intrusion	Depth of intrusion	Consequences	Trenches	Vaults
			Releases in gas	Locally enhanced fluxes of radon and C-14 gas. A house or smallholding could be constructed over a borehole. This would be analogous to a house or smallholding over an open cap vent, but the area of the gas pathway and gas fluxes would be smaller. Potential for greater radon flux than an open vent if the borehole intersects wastes with elevated Ra-226 content, but this is unlikely.	
Borehole through repository	$0.025\ m^2$ Total area of a campaign of 5 boreholes in one area = $0.125\ m^2$	Around 11 m to 19+ m depending on location	Repository evolution	Consequences similar to boreholes that terminates in the waste. The thickness (and therefore volume) of waste impacted, and number of containers impacted, would be a little greater, but still very small proportions of the totals. Boreholes would provide high conductivity pathways for some water to drain through the vault bases, but this would be little different to flowing over the 1 m high vault walls into the underdrainage blankets.	
			Releases in gas		
			Releases in gas		

8.7.2 Potential Implications for Doses

Our pathway assessments can be used to understand the potential implications of the outcomes described in Table 8.9 for doses from radionuclides released in groundwater and gas.

Groundwater pathway

Our groundwater pathway assessment model does not take any credit for the containers, so our assessment model is equivalent to assuming all the containers are damaged, which is far worse than the container damage that could arise from an intrusion event.

Our early cap degradation and poor performance case (Subsection 5.5.5) has higher total infiltration into the repository than would likely be associated with the damage from a human intrusion event. It also includes a much higher solubility limit for uranium in the trenches than would be expected under reducing conditions. This solubility limit might be more representative of uranium solubility under less reducing conditions below a damaged area of the cap.

Even though the local infiltration rate through a damaged area of cap could be higher than the average infiltration rate in the early cap degradation and poor performance case, radionuclide releases to groundwater are expected to be lower than in the early cap degradation and poor performance case. The low risks calculated for the early cap degradation and poor performance case indicate that potential groundwater pathway doses following a human intrusion event would be low compared with the dose criterion. This includes the expectation value of doses from a well, assuming a well exists in the contaminated area of groundwater between the repository and the coast.

Gas pathway

Damage to the cap could provide a conduit for release of radon and C-14 gas. We have already assessed the potential doses from radon in a house with a basement that extends into the cap gas collection layer (Subsection 8.5.7). Calculated doses for a house with a basement are below the GRA dose guidance level.

We have also already assessed potential doses from radon and C-14 gas above an open cap vent (Subsection 6.7.3). The potential doses from damage to the cap providing a conduit for gas release are analogous to the doses from an open vent. An open vent would provide a more significant gas pathway than a borehole or the backfill round a house basement, septic or sewage tank. The total area of 20 backfilled deep trial pits would be similar to the area of an open cap mushroom vent that could be intersected by a house or smallholding [26]. However, it is unlikely all the trial pits would be in the same area. They are more likely to be distributed across the cap, so a house or smallholding would only intersect one or a few trial pits at most.

The peak doses from the open vent case are far below the dose guidance level (Table 8.10), therefore the doses associated with a borehole, backfill round a house or tank, or backfilled trial pits would also be far below the dose guidance level. As it is unlikely a house or a kitchen garden would be constructed over a sewage or septic tank, any doses would be downwind of the release area, with gases subject to a degree of dilution and dispersion. This would further reduce doses compared with the open vent case.

Table 8.10: Peak doses from the open vent case

Gas	Kitchen garden and house over the cap vent
Radon	1.2 µSv
C-14	25 µSv

The only situation that could lead to higher doses than the open vent case is where a house is built over a borehole that provides a direct pathway from high Ra-226 waste into the house. In this situation, the potential doses from radon would be similar to the potential doses from radon in a house with a basement that intercepts the cap gas collection layer. Therefore, doses would be below the dose guidance level.

Cumulative impacts of multiple events

Multiple intrusion events could lead to more damage than described above. The consequences of multiple intrusion events would be as follows.

- As cumulative damage to the cap increases, infiltration through the cap, and water levels and flows in the repository, would become increasingly similar to the early cap failure and poor cap performance case (Subsection 5.5.5). The early cap failure and poor cap performance case includes a much higher solubility limit for uranium in the trenches than would be expected under reducing conditions, so the case is broadly consistent with the more oxic conditions that might develop in the localised volumes of waste below damaged areas of the cap. Therefore, as cumulative damage to the cap increases, radionuclide releases to groundwater, and groundwater pathway doses would become increasingly similar to the early cap failure and poor cap performance case. Doses calculated for the early cap failure and poor cap performance case are far below the dose guidance level.
- Releases of bulk gas, radon and C-14 gas would be distributed between a larger number of damaged areas. Doses from radon would tend to be somewhere between the open vent case and the reference case with the geomembrane intact, i.e. far below the dose guidance level. Doses from C-14 gas would tend to be between the open vent case and the early geomembrane failure case, i.e. also far below the dose guidance level.

9 Assessment of Criticality

9.1 Introduction

The repository accepts waste that contains small quantities of enriched uranium and plutonium. The quantities are limited by the LLWR WAC, which comply with the requirements of the Environment Agency for fissile materials in low level radioactive wastes at a near surface disposal facility. These limits ensure the mass of fissile material is sufficiently small that a criticality is not possible.

A criticality occurs where there is a sufficient mass of fissile material for a self-sustaining nuclear reaction. This releases large amounts of thermal energy and radiation. The amount of energy released during a near-surface criticality would immediately physically disrupt the fissile material, ceasing the reaction. The consequences of a criticality would be a 'burst' of radiation and physical damage such as melted and displaced or ejected material.

We have assessed the potential for a criticality in the repository and concluded the possibility of a criticality is so remote that it can be discounted [27].

Our assessment considers the potential for a criticality during the operational and post-closure phases. This includes the potential for a criticality associated with the tumble tipped trench waste, individual containers in the vaults, and arrays of containers in the vaults. Our assessment also considers the potential for concentration or accumulation of fissile material in the long-term through container and waste degradation and settlement, and through aqueous transport of fissile material and subsequent deposition in a small area. Theoretically, aqueous transport could result in accumulation in a small area if dissolved fissile isotopes precipitate due to a change in geochemical conditions, or they strongly sorb a material, or particulate material settles or is filtered out.

Our assessment approach and results are described below, and in more detail in reference [27].

9.2 Approach

Our assessment utilises established good practice from the UK and world nuclear industry, including the International Organization for Standardization (ISO) standard on criticality safety, and from relevant standards and guides published by the International Atomic Energy Agency (IAEA). It follows our criticality safety principles developed for assessment of disposal of much larger quantities of fissile material to a Geological Disposal Facility (GDF).

Our assessment uses pessimistic assumptions and calculations to demonstrate the LLWR WAC provide substantial safety margins against three criteria:

- criticality is not credible for the existing or illustrative future inventory;

- criticality is not credible based on acceptance of wastes at the fissile limits specified in the WAC;
- the Safe Fissile Mass (SFM), i.e. the mass of fissile material below which a criticality is not possible, is significantly above the limits set out in the WAC and the illustrative inventory, for all credible situations.

Our assessment assumes the fissile limits in the WAC for LLW would also be applied to ILW if it is accepted for disposal in the future. Cautiously we have assumed the same limits would be applied to SWTC compatible strong boxes for ILW. In practice the limits might be reduced for these containers, so the maximum concentrations of fissile isotopes are the same as for half height ISO containers.

9.3 Results

We have calculated the potential for a criticality to occur in the trenches, a single container in the vaults, or an array of containers in the vaults.

We have then qualitatively assessed whether fissile material could be redistributed and accumulate in a small area, increasing the potential for a criticality, during closure and post-closure.

9.3.1 Waste as Disposed

Inventory data for the trenches (Table 9.1) indicates the inventory of Pu-239 is too small and widely distributed for any realistic possibility of criticality. A substantial quantity of uranium has been disposed to the trenches, but this is natural or depleted uranium, which is not fissile. There is no enriched (i.e. fissile) uranium present in the trenches.

Most of the uranium in the existing vault disposals is depleted below the natural isotopic ratio, but a small amount of material is present that has been enriched. A greater quantity of enriched uranium is present in the potential future vault inventory. Also, the quantity of Pu-239 in the potential future vault inventory is substantially greater than in the existing disposals. Therefore, we have calculated SFMs for the vault waste for comparison with the amounts of fissile material that could be present.

Table 9.1: Minimum critical mass with optimal geometry and moderation compared with the Stage 2 repository inventory

Isotope	Minimum critical mass (kg)	Trenches inventory (kg)	Existing vault inventory (kg)	Future vault inventory including ILW (kg)
Pu-239	0.51	0.70	0.25	15
Uranium (1.6% to 5% enrichment)	1.9	3540 Average 0.7% enrichment No enriched uranium	399 Average 0.4% enrichment	1172 Average 1.8% enrichment

The SFM varies with the situation being considered, for example the geometry of the fissile material and the characteristics of the waste. The characteristics of the waste are important because they affect moderation of neutron energies to the levels needed for a fission reaction. The ratio of the SFM to the mass of fissile material permitted by the vault WAC was calculated for a range of credible situations.

- Individual half height ISO containers.
- Arrays of half height ISO containers stacked five-high and eight-high.
- Arrays of SWTC compatible strong boxes.

The calculations for individual containers are cautious because they assume all the fissile material is concentrated into a single mass, not dispersed in a waste matrix, with optimal geometry and moderation for a criticality.

The calculations for arrays are cautious because they assume the amount of fissile material in every container is at the WAC limit, and the aerial extent of container stacks is infinite (i.e. in plan view there is an infinite number of container stacks). The WAC limits were not reduced for SWTC compatible strong boxes, to reflect their smaller size than half height ISO containers, which means the assumed concentration of fissile material is higher in these containers. Our calculations also include pessimistic assumptions about the distribution of fissile material in the containers, with fissile material assumed to be 'lumped' rather than evenly spread throughout the containers.

The ratio of the SFM to the mass of fissile material permitted by the vault WAC is always greater than one, indicating that a criticality is not possible.

9.3.2 Redistribution of Fissile Material

Process that could redistribute fissile material during closure and post-closure are:

- surcharging of the trenches and containers of existing designs during closure, prior to final capping;
- long-term settlement of the waste in the trenches and vault waste stacks;
- aqueous transport of dissolved, colloidal or particulate fissile material by water, followed by accumulation in a small area through precipitation, sorption or filtering (clogging).

The waste in the trenches is expected to be safe because it contains too little fissile material for a criticality to be possible. The whole Pu-239 inventory would need to accumulate to the same small volume, with optimum geometry and moderation for a criticality to occur. This is not credible.

In the vaults, our calculations for a single container assume the fissile material is present as a single mass. Surcharge and settlement cannot further concentrate the fissile material.

Theoretically a criticality could occur if the fissile material in a 2 x 2 x 2 array of third height ISO containers, all at the WAC limit, could be redistributed into a single mass. However, this is not a credible situation. It requires all the fissile material to be redistributed into a single mass at the centre of the array. This could not occur through surcharge or settlement. Even if the fissile material could be redistributed in this way, the mass of material would need optimal geometry and moderation, which is very unlikely to occur. An accumulation of fissile material and other materials would not be critical.

The potential for redistribution of fissile material by water is low due to the anticipated flow and geochemical conditions in the repository.

The geomembrane and the BES layer in the cap will reduce the rate of infiltration to the repository to a very low level [13]. While the geomembrane is intact, there will not be sufficient infiltration for significant transport of fissile material by water. The geomembrane is expected to remain intact and perform for hundreds of years to a few thousand years.

Infiltration will increase once the cap geomembrane fails but it should still be low since the cap BES layer is expected to be in good condition. Water will enter top of stack containers of current designs through the damaged lids (Figure 5.2), but much of the water is likely to subsequently flow down the gaps between container stacks. Most of the water infiltrating the cap is likely to flow down the gaps between stacks of future stronger containers (Figure 5.3). This limits water contact with the waste and the potential to transport fissile material. For all container types, the grout and containers provide physical barriers to transport of particulate material.

The geochemical conditions in the vaults mean that the solubility of plutonium and uranium will be low. When combined with the low infiltration through the cap, this means that little plutonium or uranium will be leached from the vault waste. Sorption of plutonium and uranium onto the grout will be a further barrier to transport. We don’t account quantitatively for sorption on iron corrosion products in our near-field and assessment models, but this could also add to retention in the waste and the containers.

Once infiltration rates increase sufficiently, mixing of waters in the bottom of the vaults will dilute concentrations of dissolved fissile material and any colloidal fissile material. Subsequent flows through the vault bases, and over the 1 m high future vault walls to access under vault drainage blankets, are expected to be too widely distributed to result in local accumulation of fissile material.

Decrease in the pH along the transport path from the containers into the geosphere is expected to increase the solubility of plutonium and uranium. Conditions in the geosphere are more oxidising than in the repository, and carbonate concentrations are higher than in the vaults. These changes are also expected to increase the solubility of plutonium and uranium. Therefore, plutonium and uranium are not likely to precipitate and accumulate.

Only a small fraction of the disposed inventory of fissile isotopes is expected to be released to the geosphere during the assessment timeframe. Plutonium and uranium are expected to sorb only moderately to the natural sediments in the B2 unit below the repository [27], limiting the potential for accumulation. Once dissolved fissile isotopes have been transported down through the B2 unit into the B3 unit, they will be substantially diluted, so it is not credible that fissile material could accumulate in the B3 unit.

Disruption of the repository by coastal erosion does not introduce any new processes that increase the potential for accumulation of fissile material. Only a proportion of the waste would be exposed to altered conditions in and immediately adjacent to the erosion front at any time. There is no conceivable mechanism to concentrate fissile material from eroding wastes into a small location. Any waves that penetrate the repository may redistribute material, however dissolved fissile isotopes would be massively diluted and then flushed out to sea, where further dilution would occur.

10 Assessments Relating to the Heterogeneous Distribution of Radioactivity

10.1 Approach

There is heterogeneity in the inventory of radioactivity in the waste, conditions and processes in the near field, and in the properties of the geosphere (and therefore groundwater flows). There will also be heterogeneity in post-PoA radionuclide concentrations in groundwater and in the biosphere, including heterogeneity in radionuclide concentrations in the areas occupied by people and in the areas needed to produce foodstuffs at the assumed consumption rates.

We address heterogeneity in several ways:

1) in detailed underpinning models of the near field and groundwater flows in the geosphere;
2) by representing spatial variations in our assessment models;
3) by exploring spatial variations within our assessments using variant calculation cases or side models;
4) in setting controls on the waste that is accepted and how we manage the repository.

This section focuses on heterogeneities that affect the spatial distribution of radioactivity.

10.1.1 Detailed Models

Conditions and processes in the near field

We have developed models of biogeochemical processes at the scale of individual containers, waste stacks, and the whole repository, using PFLOTRAN [6]. These models help us understand evolving conditions inside and outside of containers, how fast solutes are transported, and processes that release or attenuate contaminants. The outputs from these models feed into our conceptual models for radionuclide release and transport, understanding of the spatial (and temporal) variations that need to be included in our assessment models, and selection of parameter values for assessment models. The outputs used in our assessments are described in Sections 5 to 7, and more detail on spatial variations in the near field is provided in reference [6]. These spatial variations are not discussed further in this report.

Geosphere Properties and Groundwater Flow

Our 3D geological model (developed using LEAPFROG software) describes the spatial distribution of geological units, and subunits, with significantly different hydrogeological and geotechnical properties (lithofacies units). The geological model feeds into our groundwater flow model developed using ConnectFlow, to calculate the impacts of heterogeneity at the scale of these units and subunits on groundwater flows. The scales of heterogeneity included in our site-scale groundwater flow model are illustrated in Section 5.4.2.

There is heterogeneity in the properties of the individual lithofacies units and subunits represented in our geological and groundwater flow models, i.e. at smaller scales than represented in the LEAPFROG geological model. We know this from intrusive investigations. There is sufficient heterogeneity that, even after drilling hundreds of boreholes and conducting multiple geophysical investigations, it is not possible to completely characterise the heterogeneity within each unit or subunit.

We have determined the effective hydrogeological properties of each lithofacies unit and subunit by calibrating our ConnectFlow model, so it closely reproduces site hydrogeological data. We have then developed different realisations of the ConnectFlow model that include heterogeneity within the lithofacies units and subunits, particularly heterogeneity in the B3 unit. Some of the realisations do not adequately reproduce the site hydrogeological data, so they are not credible and have been screened out from further analysis. Pathline analysis has been applied to credible realisations to understand how heterogeneity affects groundwater flows and radionuclide transport (Subsection 10.8).

10.1.2 Assessment Models

Heterogeneity in the distribution of radioactivity in the waste and repository exists at all length scales:

- on or within a waste item;
- across individual waste items;
- within a waste container;
- across containers and wastes stacks;
- between sub-areas of the vaults and between sub-areas of the trenches;
- between the vaults and trenches.

The length scales of heterogeneity in the distribution of radioactivity in the waste and repository relevant to each pathway are described in Table 10.1. The length scales of interest depend on our conceptual understanding of each pathway and types of exposures that might occur.

Table 10.1: Length scales of heterogeneity in the distribution of radioactivity in the waste and repository relevant to each pathway

Length Scale	Pathway			
	Groundwater	Gas	Coastal erosion	Human intrusion
Waste particle	-	-	✓	✓*
Waste item or part of an item	-	-	✓	✓
Container	-	-	-	✓
Waste stack	-	-	-	✓
Trench and vault sub-areas including shielded modules	✓	✓	✓	✓
Trenches and vaults	✓	✓	✓	✓

*Relevant but as little waste is exposed the risk of encounter is low. More relevant to coastal erosion.

Coastal erosion pathway

People are expected to occupy the whole area in front of the eroding repository, i.e. the cliffs, storm beach and foreshore. Doses from the cliffs, storm beach and foreshore will depend on average radionuclide concentrations in each of these areas. However, spatial variations at the length scale of sub-vault and sub-trench areas are of interest to understand variations in dose as different wastes are eroded, and to understand the range of doses that could result from non-uniform occupancy.

Some waste items are made of resilient materials and may attract interest when they are exposed on the coast. People using the coast may pause to casually inspect such items. They might also inadvertently contact, inhale, or ingest particles of waste. Therefore, the radioactivity associated with particles and discrete items of waste is also of interest (Table 2.1).

Groundwater pathway

In contrast, spatial heterogeneity in waste radionuclide concentrations is only important at larger length scales for the groundwater pathway. Radioactivity released from individual waste items, containers, and sub-vault areas will be reasonably well mixed by flows within the repository, and then further mixing and dispersion in the geosphere. Spatial variations in radionuclide concentrations in the geosphere, resulting from spatial variations in radionuclide concentrations in the waste, will be at the length scales of individual vaults and the adjacent sections of individual trenches.

Spatial variations in the distribution of radionuclides along and across the contaminated area between the repository and the coast have little impact on calculated risks for the well biosphere pathway [65]. The calculated risks depend on the expectation values of the radionuclide concentrations in the contaminated area, i.e. the area-weighted average concentrations. Therefore, the total amount of each radionuclide in the contaminated area is the key control on risk, not the spatial distribution within the contaminated area.

Our groundwater flow models describe a water balance where a fraction of rainwater recharges the regional groundwater system between the repository and the coast. If flows are concentrated through certain areas of the regional groundwater system the water balance still needs to be respected. This implies that radionuclides released from the repository will tend to mix with the same volume of water in the regional groundwater system whether flows in the regional groundwater system are homogeneous or focused within small-scale preferential paths, so there will be limited impact on the expectation values of radionuclide concentrations. Even if radionuclides are preferentially concentrated in certain preferential paths, this might ?? not ??have any consequences for radiological risk: the expectation values of the radionuclide concentrations are higher, but this is offset by the lower probability a well exists within the preferential paths.

There will be further mixing and dispersion of radionuclides that migrate to the marine environment and then the estuary environment. Radionuclides will be widely spread over large areas. People using the marine and estuary environments are expected to occupy substantial areas, so their exposures will reflect average radionuclide concentrations across those areas.

Gas pathway

Radon doses in houses and doses from C-14 in foodstuffs will vary with location on the cap. We don't know where defects (small holes) in the cap geomembrane will be relative to Ra-226 bearing waste. However, there will be a degree of dispersion and mixing of radioactive gases as they migrate through the cap, and a smallholding will capture C-14 from a substantial area (0.5 ha to 1 ha). Therefore, heterogeneity at the length scales of trench and vault sub-areas, including shielded modules, is important for doses. Like the groundwater pathway, as calculated risks depend on the expectation value of radionuclide fluxes, the calculated risks are not sensitive to small scale spatial heterogeneities.

Human intrusion pathway

The full range of length scales are relevant to the human intrusion pathway, because a broad range of events are considered. These vary from exposing parts of waste items in borehole cores through to an aircraft crash exposing a large volume of waste, potentially from adjacent trenches and, or vaults. Therefore, all length scales are considered in the human intrusion assessment.

10.1.3 Potential Controls

Heterogeneities at smaller length scales are potentially important for the coastal erosion and human intrusion pathways. There are hundreds of thousands of waste items in the repository. It is not possible to know the activity of every individual waste item, where each item is in the repository, or to model every waste item. It is not possible to characterise heterogeneity within wastes that have not yet arisen, or to know exactly where they will be placed in the repository. There are also limitations in the records for existing disposals, although the most recent disposal records provide detailed information on the locations and inventories of individual containers. These are biases in our assessment models.

To address these biases, we assess generic waste items, generic containers, and consider individual waste stacks and vault sub-areas in our assessments using side models. The results of our main assessment models and side models are used to develop controls on the activity burden associated with particles and individual waste items, and potential controls on heterogeneity in individual containers and waste emplacement. These controls ensure potential risks and doses from exposure to individual waste items, waste stacks and vault sub-areas are not substantially different to the results of our main assessment models with homogeneous assumptions, and potential risks and doses remain consistent with the levels described in the GRA.

Risks for the gas pathway depend on the expectation values of radionuclide fluxes, so the calculated risks are not sensitive to small scale spatial heterogeneities. However, there may be significant spatial variations in potential doses, and we want to control these variations as far as practicable. The potential spatial variations are greatest while the cap geomembrane is intact, as there may be gas pathways to some houses and not others, and most of the radon flux through a defect in the cap geomembrane tends to come from the container immediately below the defect. Therefore, we propose to implement controls on the Ra-226 content of individual containers control heterogeneity in doses, even if they are not needed to control risk [17].

The following sub-sections describe how our assessments include heterogeneity in radioactivity, starting at the smallest length scale in Table 10.1 and finishing at the largest length scale. Waste items are divided into low activity sealed sources and discrete items, although sealed sources can be considered a specific class of discrete item. WAC and emplacement controls on heterogeneity are discussed in the '*Implementation'* report [17].

Subsection 10.8 describes the outcomes of our work using ConnectFlow to understand the potential implications of heterogeneity in the geosphere, and how the results have informed parameterisation of the assessment models for the groundwater pathway.

10.2 Particles

10.2.1 Background

Particles are small but may have high specific activities. The small size of particles means that exposures are likely to be inadvertent, e.g. they are unlikely to be deliberately scavenged, or displayed on a shelf. The relatively high activity in particles could lead to significant doses if they are encountered. However, their small size and limited number in the repository mean the probability of encounter is low. This means the risk from particles can also be low. Nevertheless, the potential risks associated with particles are sufficient that the current LLWR WAC require wastes that contain, or could potentially generate particles, to be assessed on a case-by-case basis, before they are accepted.

Subsequent to the 2011 ESC, we assessed the potential risks from particles in existing and future disposals [104]. The characteristics of the particles that may be present in the waste or may be develop as the waste degrades are uncertain, and the numbers of particles are uncertain. Therefore, we assessed a range of particle types based on those that have been identified in the environment in the vicinity of the LLWR and elsewhere. This is a hypothetical assessment as there is no evidence these types of particles are present in the LLWR, for example, we do not expect particles derived from spent fuel to be present in the LLWR. We anticipate that any particles that are present in the LLWR are considerably less hazardous than the particles assessed. Therefore, the assessment is cautious.

The characteristics of the particles include the radionuclide(s) present. For the relevant radionuclide(s), we have assumed the entire LLWR inventory is in the form of particles. This gives the maximum number of particles that may be present. This is very cautious as most of the activity is not associated with particles.

The potential risks are then assessed for each particle type, noting that the risk is equal to the dose from exposure to a particle multiplied by the frequency of encounter (which depends on the number of particles present). The calculated risks are expected to be cautious because it is not reasonable that all the activity of a given radionuclide in the LLWR is in the form of particles. Therefore, the number of particles, frequency of encounter and risk are overestimated.

If particles are present, a given particle type may bear a range of activities. However, this uncertainty does not affect the calculated risk, so long as the dose from exposure to a particle is not sufficiently high to lead to deterministic effects. Although particles bearing more activity than an average particle would lead to higher doses than an average particle, the frequency of encounter would be lower, so the risk is the same.

Much of the radioactivity in the LLWR is present as contamination on the surfaces of waste items. Conceptually, as these surfaces degrade large number of particles could be generated. These particles would contain little activity, but the frequency of encounter would be high. Multiple encounters per year would be expected. In this situation the risks tend to those calculated assuming activity is uniformly spread throughout the waste. Therefore, it is not necessary to undertake assessments for large numbers of particles with low activities, derived from the surfaces of degrading waste items.

Exposure to particles could occur as the repository is disrupted by coastal erosion and the waste is exposed, or if the waste is exposed through inadvertent human intrusion. Inadvertent human intrusion could occur before the repository is disrupted by coastal erosion, giving less time for radioactive decay. For certain radionuclides, e.g. Cs-137, Sr-90, inadvertent human intrusion could lead to exposure of particles bearing higher activity levels than disruption by coastal erosion. However, inadvertent human intrusion would only expose a small volume of waste. The number of particles that could be exposed and the frequency of encounter would be much lower than for disruption by coastal erosion.

The findings of our assessments led to the introduction of WAC controls on active particles (AP) in waste in 2014. Active particles are a subset of particles with certain characteristics. The current WAC [121] define active particles as being in the size range 0.6 to 2.0 mm, bearing of the order of 1 MBq or more of alpha-emitting radionuclides or 0.01 MBq or more of Ra-226. Typically, particles contain more than 100 MBq g^{-1} of most alpha-emitting radionuclides or 10 MBq g^{-1} of Ra-226. Particles may be present in the waste when it is disposed or may be generated as the waste degrades and breaks up.

Having assessed the potential impacts from particles during coastal erosion of the repository [122], and inadvertent human intrusion events [123], we reviewed the potential impacts from stored and disposed wastes at the LLWR in 2016 [124]. More recently, we have updated the assessment of potential impacts from particles during human intrusion events for the 2026 ESC, capturing updates to the projected inventory and repository design [24].

The coastal erosion and human intrusion assessment models represent the repository using a 2D (plan view) grid (Figure 3.12). The model grid was developed to capture the major spatial heterogeneities in the concentrations in the key radionuclides for coastal erosion and human intrusion. There is necessarily a degree of compromise between information on the locations of radionuclide disposals and modelling practicality. The grid has been updated and improved for the 2026 ESC, to reflect changes to the inventory, repository design and to better capture peak radionuclide concentrations in certain areas.

Our initial particle assessments [122] [123] used the assessment model grid, and associated inventory data from the 2011 ESC. Our updated assessment of potential impacts from particles during human intrusion events uses the assessment model grid and inventory data for the 2026 ESC.

10.2.2 Particles and Exposure Pathways

The size of the particle is important as it influences which exposure pathways are relevant [125]. The HPA has issued draft guidance on exposure pathways and relevant particle sizes [126]. The HPA considered that the largest size of particle that could be inadvertently ingested is 1 mm in diameter. Small particles, i.e. 1 mm or so, can remain on the same small area of skin but larger particles move around and therefore do not expose the same area of skin, leading to lower contact doses. The HPA assume that the largest particle size that could become trapped under fingernails is about 1 mm in diameter. The HPA also consider that the smallest particle that could be inhaled and reach the lungs is 10 μm in diameter (AMAD[16]).

We assessed the potential impacts to recreational users of the coast from particles during erosion of the repository. Recreational users of the coast are considered because they have the highest occupancy of the coast in front of the eroding repository. The following exposure pathways and particle sizes are considered in our assessment [122]:

- inadvertent ingestion of particles 1 mm in diameter;
- inhalation of particles 10 μm in diameter;
- skin dose from 1 mm diameter particle on the skin;
- skin dose for 1 mm diameter particle on clothing;
- skin dose for 1 mm diameter particle under fingernail and toenail;
- skin dose for 1 mm diameter particle in shoes;
- external dose at 1 m from a 1 mm diameter particle;
- skin dose from sitting on a stone-sized item;
- deliberate ingestion (pica) of stone-sized item by a child;
- external dose at 1 m from a stone-sized item;
- ingestion of 1 mm particles contained in shellfish.

The inadvertent intrusion events that could lead to the largest potential doses from particles are borehole intrusion, and occupancy of a site contaminated by an event such as an excavation for a septic tank or sewer system. The exposure pathways and particle sizes considered in our assessment are [123]:

- inadvertent ingestion of particles 1 mm in diameter;
- inhalation of particles 10 μm in diameter;
- skin dose from 1 mm diameter particle on the skin;
- skin dose for 1 mm diameter particle on clothing;
- skin dose for 1 mm diameter particle under fingernail and toenail;
- skin dose for 1 mm diameter particle in shoes;
- external dose at 1 m from a 1 mm diameter particle;
- effective dose from a particle embedded in a wound.

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[16] Activity median aerodynamic diameter (AMAD) represents the diameter at which 50% of the activity in the aerosol is associated with particles of aerodynamic diameter greater than the AMAD [140].

Our initial assessments were based on the 2011 ESC engineering design. The design included a 3 m thick cap underlain by a minimum of 1 m profile material increasing to around 10 m profile material. Excavations for a septic tank or sewer system could be up to around 5 m deep. In our 2011 ESC human intrusion assessment [90] we did not take credit for the profile material. We assumed that there could be 2 m of waste exposed at any location on the cap. This volume of waste exposed would be sufficient to contaminate an area of the site that could subsequently be used for housing or smallholding. Boreholes could penetrate deeper but would expose less waste.

Our updated design now includes a minimum of 2 m of profile fill. Future containers would also be covered by CPUs or the roofs of shielded modules. Deeper excavation such as for a septic tank would only exposure waste in a few locations around the perimeter of the repository (Figure 6.7). Deeper excavations might also expose future waste if there was significant erosion of the cap reducing its thickness. However, this is a low likelihood situation [13].

Probability of Encountering a Particle

The frequency or probability of encountering a particle depends on the number of particles per gram of material and the quantity of material ingested, inhaled or in contact with the skin, clothes, fingernails etc. during the intrusion event or beach activity (walking, leisure, angling).

The range of generic particle types we have assessed and the associated maximum number of particles that could be present in the waste are shown in Table 10.2. As noted above, the calculation cautiously assumes all the activity of the relevant radionuclides in the repository is in the form of each generic particle. This assumption leads to some double counting of the inventory; for example, we assume the whole inventory of Cs-137 is in particles of the types found on the beaches adjacent to Sellafield and in particles from of the types found on the beaches around Dounreay (from spent fuel). The calculation also cautiously excludes decay during the operating lifetime of the LLWR.

To put the results in context, the HPA assessment of the Sellafield particles [127], used the actual number of particles found on beaches to calculate the number of particles per gram on the beach. The particle densities for Drigg beach were 3 10^{-3} alpha-rich particles per tonne, 1 10^{-5} beta-rich or gamma (Co-60) particles per tonne. The HPA calculated that annual probabilities of encountering a particle, summed over all exposure pathways, for individuals with assumed high beach occupancy, range from 1 10^{-6} to 4 10^{-5} per year for alpha-rich particles, and from 2 10^{-7} to 2 10^{-6} per year for beta-rich particles. The exact value depends on the number of particles found on the particular beach and on the habits of the person (walker, leisure, angling) on that beach.

Table 10.2: Particle types assessed and numbers [122] [123]

Radionuclide	Particle activity (Bq)	Number of 1 mm particles per tonne in trench waste*	Number of 1 mm particles per tonne in vault waste*	Number of 1 mm particles per tonne on the coast**
Am-241 (Sellafield particle activity)	$6.3\ 10^{+05}$	$3.4\ 10^{+01}$	$2.4\ 10^{+00}$	$3\ 10^{-01}$
Co-60 (Sellafield particle)	$1.0\ 10^{+05}$	$2.2\ 10^{+02}$	$2.7\ 10^{+03}$	Not assessed
Cs-137 (Sellafield particle)	$1.0\ 10^{+05}$	Not assessed	Not assessed	Not assessed
Cs-137 (Dounreay fuel fragment)	$3.4\ 10^{+06}$	$3.5\ 10^{+01}$	$1.5\ 10^{+01}$	$6\ 10^{+00}$
Pu-238 (Sellafield particle activity)	$8\ 10^{+04}$	Not assessed	Not assessed	$4\ 10^{+00}$
Pu-239/40 (Sellafield particle activity)	$3.1\ 10^{+05}$	$1.4\ 10^{+02}$	$1.7\ 10^{+01}$	$2\ 10^{+00}$
Ra-226 (fleck of paint)	$2.3\ 10^{+04}$	$9.4\ 10^{+02}$	$3.9\ 10^{+00}$	$5\ 10^{+00}$
Ra-226 (Dalgety Bay)	$5.0\ 10^{+04}$	Not assessed	Not assessed	Not assessed
Ra-226 (typical sealed source)	$2.0\ 10^{+05}$	Not assessed	Not assessed	Not assessed
Sr-90 (Dounreay fuel fragment)	$2.7\ 10^{+06}$	$9.3\ 10^{+00}$	$7.8\ 10^{+00}$	$2\ 10^{+00}$
Th-232 (LLWR monazite sand)	$5.9\ 10^{-01}$	$1.5\ 10^{+08}$	$4.4\ 10^{+05}$	$6\ 10^{+05}$

Radionuclide	Particle activity (Bq)	Number of 1 mm particles per tonne in trench waste*	Number of 1 mm particles per tonne in vault waste*	Number of 1 mm particles per tonne on the coast**
U-234/35/38 (enriched uranium)	6.6 10^{+02} 2.8 10^{+01} 1.2 10^{+02} Sum 8.1 10^{+02}	4.8 10^{+07}	2.5 10^{+04}	4 10^{+03}
Alpha particle (Sellafield) Pu-239/40, Pu-238, Am-241	3.1 10^{+05} 8.4 10^{+04} 6.3 10^{+05} Sum 1.0 10^{+06}	Not assessed	Not assessed	Not assessed

* Based on the 2011 ESC inventory and the maximum activity concentration in the waste. The average activity concentration in the waste gives a smaller number of particles.

** Based on the 2011 ESC inventory and 2011 ESC coastal erosion assessment model.

10.2.3 Potential Impacts during Inadvertent Human Intrusion

2013 Assessment

Our initial assessments [122] [123] made a cautious estimate of the absorbed dose to skin, neglecting self-absorption in the particle. An absorbed dose to the skin of 2 Gy or greater received in less than two hours represents an unacceptable level of damage to the skin. It is unlikely that a particle would remain in contact with the same patch of skin for more than two hours, unless it is trapped under a fingernail.

The results showed that a fuel fragment containing Sr-90 or Cs-137, similar to those found in the vicinity of Dounreay [128], could give rise to an absorbed dose above the threshold for tissue effects in skin of 2 Gy in about 2 hours, if it is excavated from the last vault to be closed, i.e. with the minimum decay time before the intrusion event. No other particles were calculated to give rise to tissue effects. Intrusion 30 years later, or into the trenches or other vaults would not result in tissue effects from any particle. Only a small fraction of the total inventory of the LLWR could be in the form of such particles. Therefore, we concluded that it is very unlikely that a particle will be encountered that could give rise to tissue effects.

Allowing for radioactive decay between completion of disposals and the intrusion event, and using experimentally determined uptake value where available, the effective dose for encounter (ingestion, inhalation or external exposure) with any single particle is calculated to be equal to or below 20 mSv. The effective dose if an alpha-rich particle becomes embedded in a wound has been cautiously estimated by HPA to be 20 to 50 mSv [127].

The annual dose is calculated as the product of effective dose per encounter and the annual frequency of encounters, i.e. the expectation value of annual dose. This is compared with the dose guidance range for human intrusion of 3 to 20 mSv y^{-1}.

Accounting for radioactive decay, the calculated annual dose to a site occupier following intrusion is estimated to be below the lower guidance level of 3 mSv, for all particle types and exposure pathways except one: the exception is external exposure to thorium-bearing (monazite) sand excavated from the maximum activity concentration cell in the trenches. In this case, an annual dose of 10 mSv is calculated, still below the upper guidance level. However, this is a localised disposal at the north end of Trench 2, so the probability of an intrusion event into this waste is low.

Intrusion into only one other area in the trenches would give rise to an annual dose above the lower guidance level; the calculated annual doses from the other areas are below the lower guidance level. The annual dose based on the average activity concentration in the trenches cell is 0.6 mSv. Hence, we conclude it is unlikely that intrusion into the trenches would give rise to an annual dose above the lower guidance level of 3 mSv.

Calculated annual doses from intrusion into the vaults are below the lower guidance level, for all particles, even based on the maximum activity concentration in any cell. The number of particles per gram is inversely proportional to the assumed activity on the particle. Hence, assuming a different activity on the particle in the waste leads to a different number of particles per gram of waste, and a different frequency (probability) of encounter of the particle. However, it leads to same calculated value (expectation value) of annual dose.

There are several cautious assumptions in these calculations. The calculation of the number of particles per gram in the waste neglects radioactive decay between filling of individual trenches or vaults and filling of all trenches or vaults. It also assumes that all of the activity is present in the form of particles (1 mm particles for most exposure modes and 10 µm for inhalation) and that these particles are mobilised (i.e. not embedded within other materials). The maximum activity concentration in any cell is used to calculate the number of particles per gram of waste. The thickness of the cap at the point of intrusion is assumed to be the minimum value, leading to the minimum quantity of non-waste material excavated with the waste and a cautious estimate (overestimate) of the number of particles per gram of excavated material. This leads to a cautious estimate of the annual frequency (probability) of encounter of a particle during the intrusion event. The annual dose from ingestion assumes that the particle dissolves in the gastro-intestinal tract.

2026 Assessment

Our updated assessment for the 2026 ESC [24] confirms that particle skin contact is unlikely to lead to stochastic effects, e.g. skin ulceration. The shortest timescales for doses to the skin to exceed 2 Gy is several hours for a particle derived from a radium sealed source.

Updated dose calculations show that the expectation value of annual dose is below 3 mSv for all particles and all relevant areas of the trenches and vaults. Note that our 2026 assessment uses the Reference Inventory containing LLW and ILW, while our 2011 assessment only considered LLW. Therefore, the number of particles and frequency of encounter will tend to be higher in the 2026 assessment.

Overall, we conclude that, in all cases, the assessed impacts are consistent with the criteria set down in the GRA.

10.2.4 Potential Impacts during Coastal Erosion

The absorbed doses, effective doses and radiological risks from exposure to given particles were calculated using the methodology applied by the HPA to assess radiological impacts of radioactive particles found on beaches in the vicinity of Sellafield. This includes the calculation of doses and risks to adults and children engaged in a range of beach activities and applies habit data based on local surveys commissioned by the Environment Agency, Food Standards Agency (FSA) and Health and Safety Executive (HSE). Overall, the approach yields cautious estimates of the activity per particle, the number of particles on the beach during erosion of the LLWR, the occupancy of the beach, and absorption in the gastrointestinal tract. We have also made similar cautious estimates for larger, stone sized objects, analogous to those found on beaches adjacent to Sellafield [122].

Absorbed dose levels are such that no deterministic effects are expected as a result of likely contact times with skin and residence times in the gastrointestinal tract for any of the postulated radioactive particles. The effective dose, for encounter with any radioactive particles that could reasonably be present, is below 100 mSv for an adult, 10 year old child and 1 year old child.

Doses and risks via inhalation are very small on account of the smaller size and therefore lower activity of particles that might be inhaled. The highest calculated effective doses relate to inadvertent ingestion of small particles, about 1 mm diameter, and larger items, plus ingestion of shellfish, parts of which may contain beach particulates.

The highest calculated effective doses for any particle that could reasonably be present relates to inadvertent ingestion of an alpha-rich particle, such as previously found on the beaches in the vicinity of Sellafield. Effective doses of 20, 60 and 60 mSv are calculated for an adult, 10 year old and 1 year old child respectively, assuming the gastrointestinal uptake factor determined by HPA for such particles. Taking account of the probability of encounter, the corresponding risks to health are well below 10^{-6} per year for adults and children.

The effective dose to a 3 month old infant from inadvertent ingestion of the highest activity alpha-rich particle found on beaches in the vicinity of Sellafield is 310 mSv; although exceeding the 100 mSv value referred to above, deterministic effects would not occur at this level of dose. Even assuming that the total inventory of relevant radionuclides in the LLWR is entirely in the form of such particles, the probability of the infant finding this particle and ingesting it is very low: estimated to be less than $3\ 10^{-6}\ y^{-1}$. Although it is possible that such particles are present in the LLWR, they can only represent a small fraction of the inventory since the forms of disposals are known and relate mainly to surface contamination and dispersed forms. Hence, the actual probability of inadvertent ingestion of such a particle must be much lower.

Estimates of doses from particles containing Ra-226 are based on a 1 mm particle of radium paint and on a larger particle found at Dalgety Bay. Inadvertent ingestion of a 1 mm particle of radium paint is calculated to give rise to an effective dose of 3, 8 and 10 mSv to an adult, 10 year old and 1 year old child respectively. The corresponding results for the Dalgety Bay particle are a factor of five higher. SEPA assessed the dose to a 3 month old infant from ingestion of this particle as 83 mSv [129]. These calculations are cautious as they assume complete dissolution of the particles in the gastrointestinal tract. Even so, taking account of probability of encounter, the corresponding risks to health are well below 1 10^{-6} y^{-1} for both types of particles.

The annual frequency of contact with Th-232 bearing mineral sand particles by beach users is greater than one. The highest frequency of encounter is due to the presence of sand in the shoes of walkers and anglers and hence exposure to skin, leading to estimated exposure to 300 particles per year for adults and a few tens of particles per year for children. The annual frequency of encounter by inadvertent ingestion and inhalation is less than one in all cases. On the other hand, the estimated dose from skin contact, ingestion or inhalation are all very low on account of the low specific activity of Th-232 and, hence, thorium bearing particles. The highest calculated annual dose is for ingestion of particulates in shellfish by a high rate (95th percentile) consumer gaining their entire annual intake from the Drigg shore, and assuming complete dissolution of the particles in the human gastrointestinal tract. This case is based on the ingestion of sand particles retained by shellfish (mainly in the gut parts) and a calculated intake of about 30 thorium bearing particles in 36 kg of shellfish consumed per year, which leads to a calculated annual dose of 20 μSv and annual risk of 1 10^{-6}.

The assessment shows that, for all radioactive particles that could reasonably be present, and taking account of the probability or frequency of encounter, the overall risk to health to adults, 10 year old children and 1 year old children from inadvertent ingestion, inhalation, skin contact for 24 hours and external dose over a month, are in all cases less than the risk guidance level of 10^{-6} per year. Ingestion of Th-232 bearing particles incorporated into shellfish is calculated to give an overall risk to a high-rate consumer of shellfish of 1 10^{-6} y^{-1}, but this estimate involves multiple cautious assumptions.

Overall, we conclude that there is no potential for deterministic health effects and, in all cases, assessed risks are consistent with the risk guidance level.

10.2.5 Analysis of Active Particles in Existing Waste

10.2.5.1 Approach

We have used the results of our assessments to implement WAC to control disposal of waste that contains or could result in generation of active particles. Any disposals that could potentially contain or generate active particles are assessed on a case-by-case basis. We have also used the results of our assessments to review the potential impacts of particles associated with existing disposed and stored waste [124].

We have examined electronic and paper records of consignments that are considered most likely to contain active particles. We have also reviewed the RECALL interviews with former operators and consignors about the characteristics of waste disposed to the trenches to identify items of potential interest.

Electronic records for disposed and stored waste were searched. The potential presence of active particles was assessed based on the Ra-226, U or Pu activity and the keywords: 'Vials', 'Glassware', 'Brickwork', 'Luminising', 'Contaminated Soil', 'MoD' and 'PCM'.

Electronic records are available for a proportion of wastes disposed of in Trench 7 but not for the other wastes in the trenches. The approach for consignments for which there are no electronic records was to identify some waste disposal cells for further investigation and then to study the paper records.

Evidence from the RECALL interviews is mostly circumstantial or anecdotal; hence linking it with specific locations, activity levels and radionuclides proved to be difficult.

Consignments potentially containing active particles were assessed for compliance with the WAC in place at the time of our work (2016). A staged approach was taken, with a more detailed assessment being undertaken for consignments containing higher levels of activity. If the records indicated that a waste consignment containing a higher level of activity also appeared to contain active particles with activity levels in breach of the (2014) WAC, then disposal of this consignment was assessed against the dose criteria on which the WAC were based. If the calculated dose was less than the relevant dose criterion, then the consignment was considered suitable for disposal without further assessment. If the relevant dose criterion was not met, we undertook a Best Available Techniques (BAT) assessment to identify the optimal waste management option, bearing in mind that these wastes were all accepted for disposal or storage pending disposal, according to the WAC in operation at the relevant time. Appropriate BAT arguments were also used to identify the preferred option for consignments containing lower levels of activity.

For a small number of containers stored in Vault 9, the BAT outcome was not clear. Reference [124] recommended reviewing the BAT approach to managing these containers after a period of 5 years. This was done in 2021 [130]. The conclusions from the review are summarised in the following subsection.

10.2.5.2 Results

It is likely that some wastes containing active particles have been disposed in the trenches but consignments containing luminous dials or waste from luminising sites are rare (around 0.01% of the records reviewed). Options for retrieval were considered, based on previous work for the 2011 ESC [131]. The difficulty in identifying the location of particular waste consignments in the trenches makes targeted retrievals practically impossible and hence the retrieval options would result in disproportionate impacts. The options assessment therefore concluded that leaving all wastes in the trenches, without disturbance, is preferred.

21 containers were identified in Vault 8 that potentially contain active particles. Some of these are in top of stack positions. None of the containers could readily be retrieved. Retrieving the containers would require large numbers of aged containers to be removed from the vault to provide access. The risks involved in this operation would be disproportionate compared with the risks from potential active particles [124]. The containers have all been grouted, so once they had been retrieved from the vaults it would be difficult to remove the waste so it could be managed via another route. Overall, the preferred option was to leave the containers in Vault 8 [124]. The outcomes of recent work to optimise the Vault 8 closure approach [10] and the schedule for placement of the first strip of the final cap over Vault 8, continue to support this as the BAT outcome.

Ra-226 luminous paint active particles may be present in one grouted container in Vault 9. The BAT outcome for this container was not clear. It was reviewed in 2021 [130] and again in 2026 [132]. The latest review concluded that the container should continue to be stored in Vault 9. A long list of options to act now should be developed over the next year. The BAT assessment should be reviewed again at the earlier of relevant options being identified in the long list, the NDA reaching a decision on further near-surface disposal facilities, and five years.

10.2.6 Conclusions

Particles may be present in the waste disposed to the LLWR. Particles may also be generated as waste degrades. We have introduced controls in our WAC on the particles that have the greatest potential to cause harm based on their size and activity levels. These are termed active particles in our WAC. Historically the WAC did not provide a control on active particles. We have reviewed the disposal records, and while active particles may be present in the existing waste, the evidence is that the quantities are small. The risks associated with locating and retrieving the waste are disproportionate to the long-term hazard. WAC have been in place since 2014 to prevent active particles of certain sizes and with significant levels of activity being disposed.

We have undertaken cautious assessments of the potential risks from particles. The assessments are hypothetical and cautious because they consider high particle activities. We do not expect particles as hazardous as those assessed to be present in the LLWR. We also assume unrealistically high particle numbers.

The particle activities are not high enough to result in deterministic effects. Even with very cautious assumptions, the risks from particles during coastal erosion of the repository are consistent with the risk guidance level, and the potential doses from particles during and following a human intrusion event are consistent with the GRA dose guidance level.

10.3 Assessment of Low-activity Sources

10.3.1 Background

Low-activity radioactive sources typically arise as redundant sealed sources that have been used for instrument testing and calibration, or similarly small, low-activity items. Sealed sources contain a small mass of radioactive material that is sealed in a capsule, with the objective of preventing any loss or dispersion of radioactive material under normal conditions of use. Many sealed sources weigh no more than a few grammes.

Historically, low-activity sealed sources were disposed to the LLWR (Table 10.3). In 2005, as part of discussion over the UK Surplus Source Disposal Programme, the operator of the LLWR came forward with proposed Conditions for Acceptance for the disposal of low-activity sources, which, after clarification, were accepted by the Environment Agency. The conditions include, amongst other things: limitation to sources of not more than 1 MBq; that after removal of extraneous packaging sources should be mixed with cement grout into ‘paint-tin type’ containers; that each such container should declared as a specific waste stream.

We carried out an assessment of the disposal of low-activity sources at the LLWR in support of the 2011 ESC [103]. The assessment:

- reviewed the conditions for the disposal of low-activity radioactive sources at the LLWR that were in place at the time;
- reviewed the information on disposal of radioactive sources at the LLWR both prior to and after 2005 (Table 10.3);
- discussed the applicable regulatory guidance levels and appropriate assumptions for assessment, and developed a methodology to assess the impacts of disposal of low-activity sources at the LLWR that is cautious with respect to the regulatory guidance;
- applied the methodology:
    - to calculate the potential post-closure impacts from the disposal of low-activity sources as it has been practiced at the LLWR, including assessing whether the present conditions and radioactive limits for disposal of sources are protective;
    - to support the development of revised conditions and radioactive limits for the future receipt and disposal of low-activity sources at the LLWR that would offer some flexibility to consigners, while continuing to assure long-term radiological protection.

Table 10.3: 2011 ESC summary of sources disposed to the LLWR prior to 2005 and after 2005 [103]

Radio-nuclide	Half-life (y)	In consignments before 2005	In consignments after 2005
Ni-63	100	Uncommon 6 of 370 MBq, 1 of 550 MBq	None found
Sr-90	29.1	Hundreds of sources Mainly less than 0.1 MBq Frequently 0.1 to 1 MBq Sometimes > 1 MBq Twelve ca. 20 MBq Maximum 370 MBq (1 of)	Tens of sources Mainly less than 0.01 MBq Maximum 0.07 MBq
Cs-137	30.0	Hundreds of sources Mainly less than 0.1 MBq Frequently 0.1 to 1 MBq Sometimes > 1 MBq 5 of 300 to 600 MBq Maximum 1,000 MBq (1 of)	Tens of sources Mainly < 0.01 MBq Maximum ca. 0.5 MBq (8 of)
Ra-226	1600	Uncommon 8 of 0.2 to 0.4 MBq Several smaller	Uncommon 3 of ca. 1 MBq Several smaller
Th-232	$1.41\ 10^{10}$	None found	Very rare 2 of 0.002 MBq; 1 of 1 MBq
Np-237	$2.14\ x10^{6}$	Only one found 0.04 MBq	Very rare 4 of 1 MBq
Pu-239	$2.41\ 10^{4}$	Hundreds of sources Typically < 0.01 MBq Sometimes 0.01 to 0.1 MBq Maximum 3.7 MBq (1 of)	Tens of sources Always < 0.001 MBq Maximum 0.0002 MBq
Am-241	432	Hundreds of sources Typically < 0.01 MBq Frequently 0.01 to 0.1 MBq Maximum 0.54 MBq	Tens of sources Mainly < 0.01 MBq Several > 0.1 MBq Maximum 1 MBq

Our 2011 assessment was undertaken before the Environment Agency issued guidance to assessors, that is specific to the LLWR, on discrete items exposed by coastal erosion [59] and subsequently the GRR [28]. We define a discrete item as, a distinct item of waste that, by its characteristics, is recognisable as unusual or not of natural origin and could be a focus of interest, out of curiosity or potential for recovery and recycling or re-use of materials should the waste item be exposed after repository closure. Sealed sources are a specific type of discrete item.

We assessed cases of casual encounter with sealed sources, and taking away a sealed source, against an annual effective dose of 20 μSv. This corresponds to the risk guidance level of 1 10^{-6} y^{-1}, assuming exposure occurs. Our assessment recognised that the probability of these cases is less than one, but the probabilities are difficult to quantify. Therefore, our assessments calculated doses assuming probabilities of one, i.e. people using the coast would encounter and casually inspect sealed sources exposed by coastal erosion and take them away.

We have not updated the 2011 dose calculations for the 2026 ESC, as the results from the 2011 ESC are still relevant. They are described in the next sub-section. However, we have updated our dose calculations for generic discrete items, which underpin the discrete item activity limits in the LLWR WAC. Recognising that sealed sources are a specific class of discrete item, we have decided to:

- retain the current grouped radionuclide activity limits for sealed sources in the WAC;
- updated the radionuclide grouping for sealed sources to be consistent with updates to the radionuclide grouping for discrete items.

The basis for this approach is further described in reference [17].

Our dose calculations for generic discrete items are consistent with supplementary guidance from the Environment Agency [59] [28] that was issued after our 2011 assessment. A key change from our 2011 assessment is the approach to applying the risk and dose guidance levels (Subsection 2.13.4.2).

We assume casual encounters with discrete items exposed by coastal erosion will occur, so an annual effective dose of 20 μSv, which corresponds to the risk guidance level of 1 10^{-6} per year assuming exposure occurs, is appropriate. We expect there will be many discrete items on the beach, so the probability of a casual encounter with an intact paint tin type container, or an individual sealed source broken out of a paint tin type container, is less than one. However, in the context of calculating activity limits for a generic discrete item, assuming exposure occurs is a reasonable approach. It is not possible to determine which discrete items could attract attention, and a person might inadvertently be in proximity with an item even if they are not examining it. Therefore, updated activity limits for sealed sources assume people will have casual encounters with them (probability of one).

A person could take away a sealed source broken out of a paint tin-type container. The probability is less than one and cannot be determined. Therefore, the dose guidance level is relevant for calculating activity limits for a generic discrete item. We assume damaged and degraded, but otherwise intact, paint tin-type containers would not be attractive for taking away from the beach, so we only calculate limits for taking away individual sources.

10.3.2 2011 ESC Assessment

10.3.2.1 Limiting Cases

Our assessment for the 2011 ESC considered the limiting assessment cases will:

- occur after the facility has been impacted by coastal erosion, paint tin type containers have been distributed with other waste on the beach and may be broken open by wave action on the cobble storm beach;
- involve a beach user, e.g. walker or beachcomber, either interacting with a single source container (described as a small container or a paint tin type container in our WAC [121]) or finding and taking away one or a small number of individual sources.

As noted above, the 2011 ESC cautiously assessed these events on the basis that they occur, although this is by no means certain. Assuming an event occurs, the impacts to assess are:

- the effective dose due to handling and, or proximity to a source container on the beach (or similar situation);
- the effective dose due to prolonged proximity to an individual source taken away from the beach (or similar situation);
- the equivalent dose to a localised area of skin due to handling, or pocketing of an individual source (or similar situation).

The earliest time considered for start of disruption of the repository by coastal erosion was 300 years after present. The assessment cautiously assumed source containers and individual sources may be present on the beach or foreshore at that time.

10.3.2.2 Doses from Casual Inspection of a Source Container

The annual probability that an individual will encounter and take time to inspect a source container, or source is less than one but is difficult to estimate. Although we assessed the case cautiously, on the basis that exposure occurs, the risk will not be sustained. The same individual is very unlikely to also encounter and inspect a source container in subsequent years, since the number of such containers is low.

The calculated doses from a source container with the maximum loading expected to be found in practice were less than 20 μSv. If a source container could be loaded to the level meeting the definition of LLW, the calculated doses exceed 20 μSv. If the probability of exposure could be quantified, all the calculated risks would be less than 1 10^{-6} y^{-1}.

Table 10.4: Effective doses for inspection and proximity to a 15 litre source container (paint tin type container) calculated in the 2011 ESC

Case	Content	Effective doses for the event (μSv)	
		at time of disposal	at 300 years
Maximally loaded container meeting the definition of LLW	100 MBq Ra-226	29	26
	100 MBq Th-232	42	42
Reasonable maximum loaded container in practice	15 MBq Ra-226	4.4	3.9
	15 MBq Th-232	6.4	6.4
Maximally loaded container meeting the definition of LLW	100 MBq alpha 300 MBq other*	51.6	0.045
Reasonable maximum loaded container in practice	30 MBq alpha 90 MBq other**	15.5 82% Co-60, 17% Cs-137	0.013 20% Cs-137, 79% Am-241

* 100 MBq each of Co-60, Sr-90 and Cs-137, plus 50 MBq each of Pu-239 and Am-241.

** 30 MBq each of Co-60, Sr-90 and Cs-137, plus 15 MBq each of Pu-239 and Am-241.

10.3.2.3 Doses from Inspection or Keeping Individual Sources

We assessed the case of finding a single 1 MBq source on the beach that has broken out of a paint tin type container and grout. This may be put in a pocket and taken home. This is a low-probability event since the sources are generally very small and are liable to fall between the cobbles of the storm beach or become buried in the sand of the foreshore.

The limiting concern was the localised equivalent dose to skin that might be received from handling a source or placing it in a pocket. Such doses are difficult to estimate and depend on the form of the source, its condition at the time of discovery and the duration of close contact time. The equivalent dose values calculated in Table 10.5 should be regarded as probably overestimating doses for a typical handling event by about an order of magnitude.

On this basis, the localised skin dose from handling and pocketing a 1 MBq disposed source at 300 years after disposal are in the range zero to a few mSv depending on the radionuclide, form of the source (e.g. contained source or surface source), and its condition. These doses are substantially below the ICRP recommended annual limit for equivalent dose to skin of 50 mSv for public exposure, which relates to the prevention of deterministic effects.

Annual effective doses due to taking home and prolonged exposure to a low-activity (1 MBq at disposal) source kept in the home are low for most radionuclides; the largest doses (in the range 10 to 30 µSv) are for Ra-226, Th-232 and Np-237 sources. Following the Environment Agency guidance issued after our 2011 assessment [59] [28], these doses may be compared with the dose guidance level of 3 mSv for prolonged exposures, where the probability of exposure cannot reasonably be quantified. The calculated doses are far below the dose guidance level.

Table 10.5: Equivalent tissue (skin) doses for close contact with a 1 MBq source for 1 hour calculated in the 2011 ESC

Radionuclide	Tissue (skin) dose at time of disposal, mSv			At 300y, mSv
	Photon	Beta x 0.2*	Photon + beta	Photon + beta
C-14	$0.00\ 10^{+0}$	$3.08\ 10^{+1}$	31	30
Cl-36	$1.94\ 10^{-4}$	$3.69\ 10^{+1}$	37	37
Co-60	$3.07\ 10^{+0}$	$4.54\ 10^{+1}$	49	0.000
Ni-63	$0.00\ 10^{+0}$	$1.26\ 10^{+1}$	13	1.6
Sr-90	$5.94\ 10^{-5}$	$7.42\ 10^{+1}$	74	0.058
Cs-137	$7.70\ 10^{-1}$	$4.60\ 10^{+1}$	47	0.046
Ra-226	$2.70\ 10^{+0}$	$1.47\ 10^{+2}$	150	132
Th-232	$3.74\ 10^{+0}$	$1.62\ 10^{+2}$	166	166
Np-237	$1.30\ 10^{+0}$	$7.20\ 10^{+1}$	73	73
Pu-239	$4.00\ 10^{-2}$	$5.57\ 10^{+0}$	5.6	5.6
Am-241	$3.53\ 10^{-1}$	$2.46\ 10^{+1}$	25	15

* Relates to geometric considerations, see [103].

10.3.2.4 Conclusions

There are issues over the probability of events, and uncertainties that are difficult to quantify concerning the source form and condition, and exposure geometry and duration. Nevertheless, assessing on the cautious basis that source containers and individual sources will be found, inspected and, in the case of individual sources taken away, effective doses were calculated that are consistent with an annual effective dose of 20 µSv, which corresponds to the risk guidance level of 1 10^{-6} per year assuming exposure occurs. The doses associated with taking away a sealed source are far below the dose guidance level of 3 mSv, for prolonged exposures. Equivalent doses to skin were calculated that are consistent with the ICRP recommended annual limit for equivalent dose to skin of 50 mSv for public exposure.

In the 2011 ESC, we concluded that the conditions for acceptance and disposal of low-activity sources, as agreed with the Environment Agency in 2005, are highly protective in the case of sources from nuclear fuel cycle operations and are protective in practice in the case of disposals of Ra-226 and Th-232 sources.

The results of the 2011 ESC assessment were used to update the WAC for low-activity sources and continue to underpin the current low-activity source WAC [121]. Our current WAC require waste producers to disfigure sources that may be attractive as far as possible. This reduces the potential for casual inspection of a sealed source or taking it away, even though the benefit cannot be quantified in assessment calculations.

10.3.3 Updated Calculations for Low-activity Source Limits

We have calculated updated activity limits for low activity sources, treating them as a specific type of discrete item. The calculations consider similar, but updated, potential exposure situations to those described above, and use updated assessment model data. For example, updated data include internationally recommended dose factors [57].

The earliest timescales for start of disruption of the repository by coastal erosion have increased compared with the 2011 ESC [48]. Given the large uncertainties in future greenhouse emissions, and the large uncertainties when calculating potential rates of erosion, the earliest timescales for disruption of the repository by coastal erosion are liable to further change in future assessments. It is not desirable to have activity limits that are dependent on such uncertain timescales.

The updated approach is to assume that the earliest time exposures could occur is 300 years after the last waste emplacement, i.e. around 2430 AD. This is based on a maximum credible duration of active institutional control of 300 years. The reference assumption is that active institutional control would last around 100 years [10]. However, if erosion is observed to occur faster than currently expected, then the period of institutional control could be extended. Institutional controls might still be in place and prevent exposures when erosion of the waste begins. Therefore, 400 years of radioactive decay are assumed in the assessment calculations, i.e. 2442 – 2026 = ~400 years.

Wastes disposed in the future may be subject to less than 400 years radioactive decay before exposures could occur. However, the updated activity limits will be appropriate for the period until the next major review of the ESC in 10 to 15 years. They would need to be updated for the next major iteration of the ESC, potentially considering a slightly shorter period of radioactive decay.

Cases involving casual encounters with sealed sources and taking away (scavenging) sealed sources are assessed.

Casual encounters

Our dose calculations to derive activity limits for discrete items assume casual encounters last for a maximum of one hour at a distance of 0.3 m. We consider that a maximum exposure duration of one hour is cautious. Most encounters of this nature would last seconds, and a few might last a several minutes. Therefore, although the assumption of an exposure lasting one hour is somewhat arbitrary, it means that even in the unlikely event that a person spends a longer than expected amount of time inspecting the highest dose rate discrete item, the conditional risk to them would be low.

Over a year, a person may have multiple casual encounters with discrete items at a range of distances. Doses decrease rapidly with increasing distance. Therefore, multiple encounters at a range of distances are assumed to be equivalent to spending one hour inspecting discrete items at close range.

These arguments are relevant to sealed sources. It is unlikely that anyone would spend an hour casually inspecting a degraded paint tin type container at a distance of 0.3 m. Also, a person making recreational use of the coast could have casual encounters with paint tin type containers and a range of other discrete items.

Scavenging

Our dose calculations to derive discrete item activity limits consider two exposure events. Both events assume people involved in scavenging activities are exposed to discrete items including sealed sources. Therefore, they receive doses from the discrete item and other waste they encounter during scavenging, including activity dispersed across the coast in front of the eroding repository.

- Event 1. A scavenger picks up a small discrete item (such as a sealed source) and places it in their pocket. It remains in their pocket for 16 hours. One item is retrieved during 50 hours spent scavenging, i.e. one item per year.
- Event 2. A scavenger picks up a discrete item and places it on a shelf, one metre from where they sit for a few hours per day, for example in the evenings. Assume four hours of exposure to the sealed source per day. One item is retrieved during 50 hours spent scavenging, i.e. one item per year.

The scavenger is exposed to activity on the coast through external irradiation, inadvertent ingestion, inhalation and dermal contract while scavenging. They are exposed to the discrete item by external irradiation and inadvertent ingestion [24] as these are expected to be the dominant exposure pathways [60]. This is a cautious approach for sealed sources as the ingestion pathway is unrealistic for a sealed source [60]. Exposure by inadvertent ingestion is only considered to occur while handling the sealed source, not while it is on a shelf. Skin beta doses are not considered, as reference [122] shows that doses from external irradiation and inadvertent ingestion are more limiting.

Calculations and results

Dose calculations are implemented in GoldSim and the same mathematical models and data are used to calculate doses from casual encounters and scavenging. Doses are calculated for discrete items of a range of weights, with lower weight items being relevant to individual sealed sources and an intact paint tin containing grouted sealed sources.

For each discrete item weight, doses are calculated for a discrete item with unit activity of each radionuclide to give the dose per unit inventory for each radionuclide. The relevant regulatory criterion is then divided by the dose per unit inventory to give the maximum activity of each radionuclide for that discrete item weight.

For scavenging events, the combined doses from the discrete item and undertaking scavenging are considered. Therefore, the maximum acceptable dose from the discrete item is equal to 3 mSv minus the dose incurred during scavenging. The dose from scavenging is uncertain, for example the future inventory is uncertain, and the dose from scavenging depends on which wastes are scavenged. Nevertheless, this approach ensures the activity limits calculated for discrete items account for potential exposures from other waste in addition to discrete items. A cautious approach is adopted because doses from other waste may be notable (Figure 8.9 and Figure 8.10) and orders of magnitude higher than the doses from recreational use of the coast.

Doses from Event 1 are more limiting than from Event 2. The scavenger is near the sealed source for longer in Event 1 than Event 2. Event 2 involves a prolonged exposure but at a further distance. This shows that proximal interactions provide tighter constraints on sealed source limits than longer exposures at greater distances.

The activity limits calculated for scavenging sealed sources (Event 1) are similar to the activity limits calculated for casual encounters with sealed sources [24]. The differences in the assumed exposure durations and maximum acceptable doses broadly cancel out. The activity limits are detailed in reference [17].

10.4 Discrete Items

10.4.1.1 Background

Some wastes by their nature may remain largely intact upon erosion and encourage investigation either for curiosity or with intent to recover. This could be especially so for discrete items that remain recognisable as human artefacts, e.g. durable metal items, in a range of sizes. People using the coast are expected to encounter and casually examine discrete items, including at close range. It is also possible that people could scavenge discrete items from the coast, leading to additional exposures, or encounter discrete items exposed by human intrusion. The probability of scavenging discrete items or exposing them by human intrusion cannot be determined. As described in the previous subsection (10.3), sealed sources are a specific type of discrete item.

Only small discrete items, or parts of larger items could be exposed by borehole drilling. It might be difficult to remove larger discrete items from an excavation for a septic tank or sewer system. Even if larger discrete items are excavated, arguably they are less likely to be left onsite exposing subsequent site occupants compared with smaller discrete items. Discrete items of all sizes would be exposed by coastal erosion.

The activity associated with discrete items was included in the coastal erosion and human intrusion assessment calculations undertaken for the 2011 ESC. Close encounters with discrete items could potentially result in additional doses that are not included in the assessment models, if the average activity of the discrete items is higher than the average for containerised waste.

It is not possible to include exposure to individual discrete items in assessment models. The UK Radioactive Waste Inventory (UKRWI) does not include information on discrete items in potential future disposals, and the characteristics of future waste streams are uncertain and potentially very variable. Categorisation of a waste item as a discrete item is often subjective, and there are many thousands of candidate discrete items in the waste. It is not possible to calculate how many discrete items would be exposed by intrusion or on the beach at any given time, what their condition would be, what the activity distribution would be, and which ones (if any) might attract interest. This is a bias in the assessment models.

Following discussion of the issues with the Environment Agency, and the Environment Agency's additional guidance [59], we concluded that an additional control was needed on the potential doses from discrete items in the form of activity limits for discrete items in the WAC. The discrete item activity limits control the potential heterogeneity in activity in a waste consignment, and any potential additional doses from discrete items. Therefore, the control addresses the bias of not representing discrete items explicitly in the assessment models.

The discrete item WAC aim to limit potential doses from casual encounters with discrete items, so that they do not disproportionately add to the doses from occupancy of the coast in front of the eroding repository. The additional dose from casual encounters with discrete items is disproportionate if it is similar to or greater than the dose from occupancy of the coast in front of the eroding repository and results in calculated annual effective doses around or above the dose equivalent to the risk guidance level assuming exposure occurs (i.e. 20 μSv).

Discrete item activity limits were developed in 2013 [60] and introduced into the WAC in 2014. The activity limits only apply to waste items that meet the definition of a discrete item. Containers of waste that could be intact when exposed by coastal erosion, and then durable in the coastal environment, such as thick-walled concrete boxes, are also considered to be discrete items, in addition to relevant waste items within them.

The discrete item activity limits vary with item weight and radionuclide. Items are divided into three weight categories: ≤ 1kg, >1 kg to <100 kg, ≥100 kg. Radionuclides are placed into four groups (A, B1, B2 and C). The activity limits for each radionuclide group depend on the item mass. The activity limits for the lowest weight category align with those for low-activity sources. A Sum of Fractions (SoF) calculation is undertaken to determine whether a discrete item is acceptable (the method is detailed in reference [121]).

Group C includes radionuclides with short half-lives and, or low radiotoxicity. The activity limit for group C was set higher than the 4 GBq t^{-1} alpha and 12 GBq t^{-1} beta/gamma limits for LLW. However, it was truncated below the maximums safe limit to a level considered to be appropriate to a facility Permitted to dispose LLW. The truncation also limits potential doses to LLWR operators, other people onsite and people offsite during the PoA from short lived radionuclides such as Co-60. Potential exposure calculations were not undertaken to inform the truncation as other WAC control potential doses during transport and the operational phase.

The WAC include an additional control on the radiological burden from multiple discrete items in the same container. The SoF for all the discrete items in a consignment must not exceed ten calculated using the discrete item limits for items of mass 100 kg or greater.

The GRR [28] indicates, amongst other things, that the Environment Agency would accept an assessment of encounter with discrete ‘visually identifiable objects’ against a significance test of an effective dose guidance level in the range of around 3 mSv/year to around 20 mSv/year. This is by analogy with the GRA Requirement R7 for assessment of human intrusion, which is to account for situations that ‘cannot reliably be assessed in terms of a numerical value of probability’.

Given that we expect the repository to be disrupted by coastal erosion, and people making recreational use of the coast will have casual encounters with discrete items, we concluded that it is cautious to assess casual and short duration encounters with discrete items against an effective dose of 20 µSv. This dose analogous to the risk guidance level assuming exposures occur. Individuals may examine or remain in close proximity with multiple items over the period of a year, but only one or a few will bear radionuclides at higher levels, approaching the discrete item (activity) limits (DILs), i.e. SoF close to one. This is a more cautious approach than required according to the GRR, but we consider it is appropriate given that we expect casual encounters with discrete items will happen.

It is uncertain whether individuals would deliberately seek out, collect, take away or attempt to disrupt discrete items exposed on the coast. Consistent with the GRR, these activities are judged against the effective dose guidance range of around 3 mSv/year to around 20 mSv/year. The 3 mSv/year applies to an individual that was carrying out such activities over several years and 20 mSv/year applies to cases where the activity extended over about a year or less. The assessment underpinning the current WAC [60] found that DILs for people casually encountering discrete items while making recreational use of the coast are lower (more restrictive) than for people inadvertently intruding into the waste and deliberately scavenging items from the coast. Therefore, DILs for people casually encountering discrete items while making recreational use of the coast are included in the current WAC.

In the case of discrete items excavated during human intrusion, the effective dose guidance range of around 3 mSv/year to around 20 mSv/year applies. Contact times of two to three orders of magnitude greater than we consider for the casual and short duration beach encounter would be required for exposure to items excavated during human intrusion to be limiting. In addition, the number of discrete items that could at any time be exposed on the beach and foreshore is much greater than could be excavated through any of the assessed human intrusion events. Thus, the likelihood (or frequency) of encounter with a discrete item bearing activity near to the DILs during human intrusion is much lower than following coastal erosion. Therefore, we did not assess cases of human intrusion as part of the determination of DILs.

Introduction of the DILs in the WAC placed additional characterisation requirements on waste producers. These have had implications for waste producer worker doses, for example resulting in additional exposures where discrete items need to be moved from high background areas to low background areas for characterisation. They have also had implications for producer waste management plans.

Around 15,000 discrete item assessments have been undertaken over the last ten years since the DILs were introduced to the WAC. However, some waste producers take a cautious approach and assess all waste items as potential discrete items, so the actual number of potential discrete items is expected to be smaller. We have also undertaken a retrospective assessment of potential discrete items in the trenches and existing vault waste [124]. We have reviewed the results of the discrete items assessments to understand the performance of the DILs and inform updates to the DILs and WAC.

We have calculated updated DILs considering experience of applying the WAC, the results of the review of the existing discrete item assessments, and updates to calculation data. Updated DILs have been calculated for people who make recreational use of the coast, and people who deliberately scavenge materials from the coast. The arguments presented above against calculating DILs for human intrusion cases are still relevant, as discussed in reference [24]. Therefore, our updated activity limit calculations only consider cases associated with coastal erosion. The results of these calculations are taken forward by the '*Implementation'* report [17].

The following subsections summarise:

- the findings from our review of existing discrete item assessments;
- the findings from our review of potential discrete items in the trench and vault wastes;
- the results from our updated activity limit calculations.

10.4.1.2 Review of Existing Discrete Item Assessments

Nature and materials

The masses of discrete items disposed to date have a log-normal distribution (Figure 10.1). The modal mass is in the range 10 kg to 100 kg. Most discrete items fall within the existing DIL 1 kg to 100 kg WAC weight category, with most of the remainder in the DIL >100 kg category.

The SoF results have a truncated log-normal distribution (Figure 10.2). The distribution is truncated at high SoF values by the discrete item activity limits.

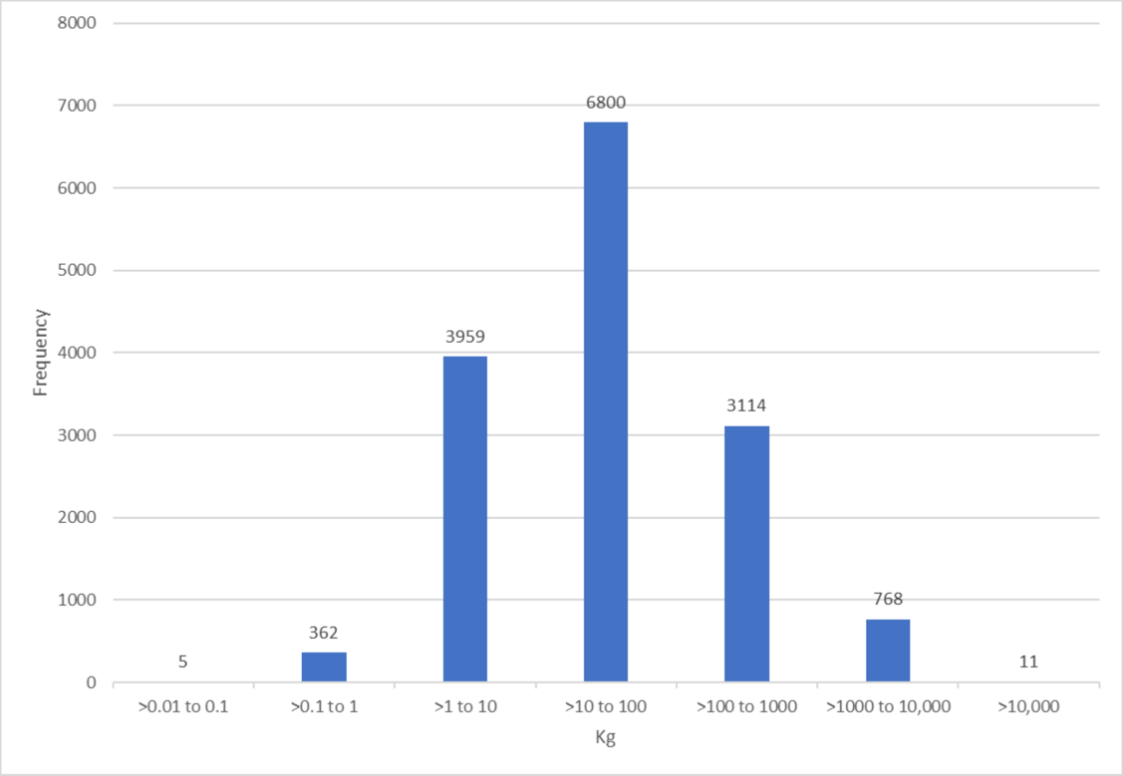

Figure 10.1: Distribution of mass of discrete items

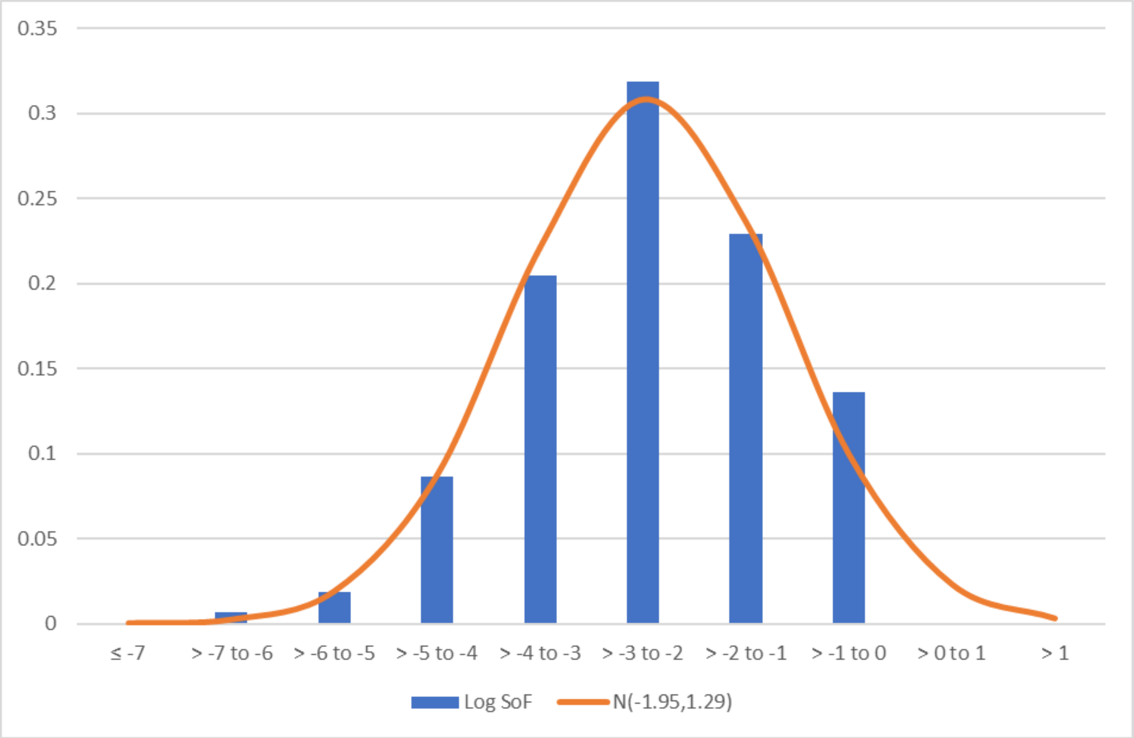

Figure 10.2: Distribution of log SoF compared to normal distribution of log SoF values, N (mean, standard deviation). Note the normal distribution is not truncated.

The data set was randomly sampled to determine the expectation value (mean) of the SoF. A discrete item randomly selected from the population would be expected to have a SoF of 0.040, which corresponds to a dose rate of 0.80 μSv hr^{-1}. This is a small overestimate because the Group C activity limit is truncated below the level that leads to a dose of 20 μSv, so the SoF is overestimated. The expectation value of the dose rate is well below 20 μSv hr^{-1}. The expectation value of the dose rate is not much different from the mean activity of the waste. Therefore, casual encounters with most discrete items on the coast would result in no additional dose to that from waste exposed on the coast, or only a small additional dose.

There is not a strong relationship between mass and SoF. Only a small number of very heavy items have been disposed. These all have high SoF results. There might not be enough items for this to be a statistically significant result. Very heavy items are expected to have higher SoF as the discrete item activity limits intentionally[17] become increasingly restrictive with increasing mass above 100 kg.

Effectiveness of the discrete item WAC

The key points from the preceding discussion are:

- there could be many discrete items exposed on the coast at a given time, leading to multiple encounters during a year;

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[17] To reflect the increasing probability of encounter, and that large items might attract more interest.

- most items would have a low dose rate, substantially below 20 µSv hr^{-1}; and
- casual encounters last seconds to a few minutes at most.

Although it is not possible to undertake a radiological risk assessment for discrete items, the discrete item assessments undertaken to date, combined with data from recent habits surveys, and observations from analogue sites are used to review the effectiveness of the current discrete item WAC. The approach taken is to review the additional dose that could potentially be received from discrete items that is not accounted for in the main assessment model. No account is taken of the timing of exposure of individual discrete items, the duration of exposure to individual discrete items, their condition, persistence on the coast, potential interest to users of the coast, or the number of discrete items on the coast at any given time.

It is assumed there are always many discrete items exposed on the coast, and the time spent casually inspecting discrete items is independent of the numbers, types and condition of the discrete items, because the reason for occupying the coast is to undertake recreational activities, not to inspect discrete items. Therefore, the expectation value of the dose rate of the population of discrete items is an appropriate metric for the potential additional risk from casual inspection of discrete items, and therefore also the potential additional dose, assuming exposure occurs.

A cautious approach is taken whereby the potential dose from discrete items is assumed to be completely additional to the doses from occupying the coastline in front of the eroding repository. This is likely to be a cautious approach because:

- discrete items with external irradiation dose rates that are lower than those from general waste exposed on the eroding coastline may not result in any additional dose, depending on the exposure geometry (e.g. exposure to an item on the beach where there may be no additional dose, compared with picking up an item where there may be an additional dose); and
- the activity associated with discrete items is already included in the main assessment model, so there is some double counting.

Two PRPs are considered:

- high occupancy recreational users of the coast (e.g. dog walkers) as represented by the Recreational PRP in the main assessment model; and
- regular visitors to the coast, who have significantly lower annual occupancy compared with the Recreational PRP but might be preferentially interested in discrete items.

There is limited basis for the second PRP from local habits surveys. However, this PRP is conceptually possible based on observations at analogue sites, and it is useful to build further confidence that the discrete item activity limits are achieving their aim. The habits assumed for this second PRP are deliberately weighted towards having an interest in discrete items to help build confidence in the effectiveness of the discrete item limits.

Casual encounters with discrete items by high occupancy recreational users (dog walker)

There is not a clear dividing line between the number of casual encounters that would be associated with recreational use of the coast, and the number of encounters that would be associated with deliberately seeking out and scavenging discrete items. Here we are only interested in casual encounters. Given that the highest occupancy recreational users, such as dog walkers, would be present on the coast multiple days per week, they are assumed to inspect many discrete items, with encounters having different durations, and at different distances. Note that the distance has a large impact on the dose rate. Over a year, these multiple encounters are assumed to be equivalent to spending one hour inspecting discrete items at close range (0.3 m).

Casual encounters with discrete items by visitors

Interactions of people with demolition waste exposed by coastal erosion at Formby [25] and concrete anti-tank blocks on the coast at Alnmouth [25] [48] provide insight into the some of the potential interactions of people with manmade objects on beaches. Regular users of the coast, such as dog walkers, are likely to be familiar with the presence of manmade items on the coast. Although their occupancy is high, and their frequency of casual encounters with discrete items could also be high, the duration of the encounters would likely be low. In contrast, occasional visitors to the coast might have greater interest in manmade items, including playing with them (Formby analogue) or preferentially occupying the adjacent area because large manmade objects provide or support a windbreak (Alnmouth analogue).

A Visitor PRP is conceptualised that spends a day on the beach several times per year. There are only limited national and local habits information to parameterise the behaviour of the PRP, so the assumed behaviour is informed by the analogues and experience of the assessment team. As noted above, the assumed habits are deliberately weighted towards having an interest in DIs. Therefore, the assumed behaviours likely have bias, but the results are informative.

A family is assumed to make 6 visits to the beach over the course of the summer. Each visit is assumed to last 5.5 hours, for example this would be from 10 am to 3.30 pm. This gives a total occupancy of 33 hours per year. Reference [109] suggests that a leisure related occupancy of 30 hours per year is reasonable for children and infants but note the lack of data. This suggests an occupancy of 33 hours per year may be representative of the most exposed people. However, 33 hours per year is around half the time the most exposed, local recreational users of the coast are expected to spend in front of the eroding repository, which suggests it may be an overly cautious assumption. Also, it does not consider that occupancy might be reduced by the potential amenity impacts of the eroding repository.

For each visit, the family is assumed to spend 3.5 hours on the sandy foreshore (e.g. playing ballgames, paddling at the water edge, rockpooling, building sandcastles), 1.5 hours on the cobbly storm beach, e.g. a ‘base’ with windbreaks might be set-up there, and 0.5 hour clambering on the cliffs / talus and finding discrete items. A total of 2 hours is spent playing with the discrete items they find. This includes using large discrete item to provide or support a windbreak.

The items most likely to form a windbreak are large items such as fuel transport flasks and large, thick-walled, concrete containers such as WAGR boxes. Many such items would provide substantial shielding, but this is not accounted for in these simple, exploratory calculations. It is less likely that people would shelter behind a container when it is has lost its lid and its contents are being eroded. Stainless steel drums might not provide substantial shielding, unless they are annular grouted, however they are too small to provide usable windbreaks.

Potential doses to high occupancy recreational users (dog walker)

The expectation value of the dose from an individual casual encounter with a discrete item, at close range, is 0.8 µSv hr^{-1}, multiplied by the duration of the encounter. Assuming the Recreational PRP (dog walker) encounters discrete items for the equivalent of 1 hour per year at close range, the total annual effective dose would be 0.8 µSv (Table 10.6). This compares with an annual effective dose of 10.5 µSv from general occupancy when the vaults are eroding, including the shielded modules. The dose from discrete items would account for 7% of the 11.3 µSv total dose. The total dose would be lower prior to exposure of the shielded modules (Table 10.7), but with the dose from discrete items accounting for a higher proportion of the total dose (11%). This assumes all the potential dose from discrete items is completely additional to the dose from general occupancy, which as noted above, is unlikely to be the case.

Table 10.6: Combined potential annual effective doses from discrete items and general occupancy during erosion of the vaults including the shielded modules

Location	Dose rate	Occupancy hr		Dose µSv		Assumptions
	µSv hr^{-1}	Dog walker	Visitor	Dog walker	Visitor (adult)	
Discrete items	0.80	1.0 (a)	12	0.8	9.6	Dog walker assumed to examine discrete items for 1 hr y^{-1}.
Cliffs	0.86	4.8	3	4.1	2.6	
Storm beach	0.43 (b)	11.6	9	5.0	3.9	
Foreshore	0.03	49.2	21	1.4	0.6	
In situ foreshore	0.0	0.0	0	0	0.0	
Total	-	65.6	33.0	11.3	16.7	
Fractional dose from discrete items	-	-	-	7.1%	57.6%	

Notes

a. This is the equivalent time spent in close proximity to discrete items. It is not additional to the time on the cliffs, storm beach and foreshore.

b. Includes doses from the cliffs and talus while occupying the storm beach.

Table 10.7: Combined potential annual effective doses from discrete items and general occupancy during erosion of the vaults prior to exposure of the shielded modules

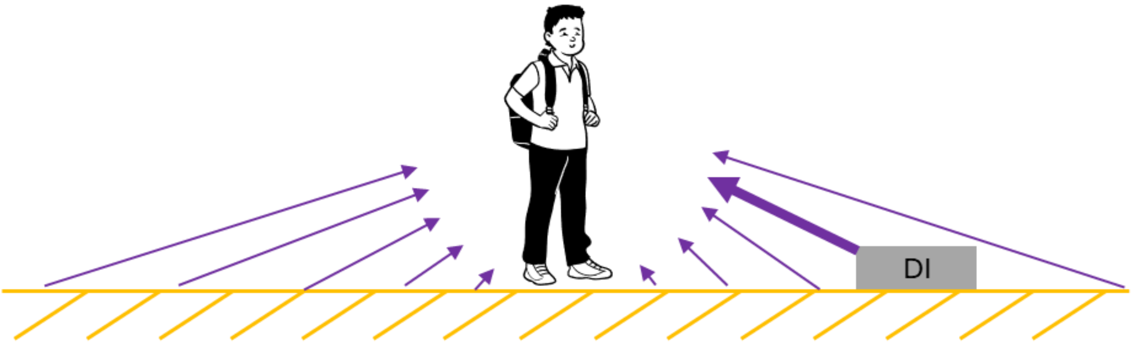
Location	Dose rate	Occupancy hr		Dose µSv		Assumptions
	µSv hr^{-1}	Dog walker	Visitor	Dog walker	Visitor (adult)	
Discrete items	0.80	1.0 (a)	12	0.8	9.6	Dog walker assumed to examine discrete items for 1 hr y^{-1}.
Cliffs	0.60	4.7	3	2.8	1.8	
Storm beach	0.28 (b)	11.5	9	3.2	2.5	
Foreshore	0.01	51.3	21	0.60	0.2	
In situ foreshore	0.0	0.0	0	0	0.0	
Total	-	67.5	33.0	7.4	14.2	
Fractional dose from discrete items	-	-	-	10.7%	67.8%	

Notes

a. This is the equivalent time spent in close proximity to discrete items. It is not additional to the time on the cliffs, storm beach and foreshore.

b. Include doses from the cliffs and talus while occupying the storm beach.

As most discrete items weigh 10 kg or more (Figure 10.1), they are unlikely to be moved by people or mobilised by anything other than the strongest storm waves. Therefore, the greatest spatial density of discrete items will likely be in the cliffs, talus and on the storm beach. The spatial density of discrete items would decrease with increasing distance from the cliffs, due to degradation, break-up, burial and dispersal of discrete items by coastal processes including wave action. Given that the dose rates from waste exposed in the cliffs, talus, and on the storm beach, are of the same order as the expectation value of the dose rate from discrete items, the dose rate from discrete items would only be a little above the 'background' dose rate. Therefore, the additional dose from discrete items is likely to be less than 0.8 μSv (e.g. Figure 10.3).



'Background' dose rate from the storm beach = X μSv h^{-1}
Dose rate from the DI = Y μSv h^{-1}
Additional dose rate from the DI = Y - X

Figure 10.3: Schematic illustration of the proportion of the potential external irradiation dose from discrete items that is additional to the 'background' dose rate

In the analysis above the expectation value of the dose rate from discrete items is for existing disposals, i.e. LLW, while the average dose rate from the waste assumes disposal of LLW and ILW. The potential magnitude of this bias is explored by considering the additional dose from discrete items for the case where ILW is not disposed (Subsection 7.5.3). For the no ILW case, the dose from discrete items could be a greater proportion of the dose from occupancy of the coastline, which is around 2.1 μSv. The additional dose from discrete items could increase the total dose to the Recreational PRP by a factor of (2.1 μSv + 0.8 μSv) / 2.1 μSv = ~1.4. However, the total dose (2.9 μSv) would still be significantly lower than the reference case including ILW disposal, and significantly below 20 μSv.

Disposal of ILW could increase the expectation value of the dose rate from discrete items, so it could be higher than the value of 0.8 μSv hr^{-1} assumed in Table 10.6 and Table 10.7. Therefore, the additional dose from close inspection of discrete items could be greater than shown in these tables. Crude extrapolation of the results for LLW only case to the reference case, prior to erosion of the shielded modules (Table 10.7), implies the total dose to the dog walker could increase from 7.4 μSv to around (7.4 μSv – 0.8 μSv) x 1.4 = 9.2 μSv, with an additional 9.2 μSv – 6.6 μSv = 2.6 μSv due to close inspection of discrete items.

The potential additional dose from casual inspection of discrete items is similar to uncertainty in the dose calculated in the main assessment model resulting from uncertainty in the proportion of total time on the coast that is spent in front of the eroding repository, and the distribution of that time across the cliffs, talus, storm beach and foreshore.

Children and infants have also been considered in our assessments. Local habits data, e.g. [133] [106] [107], shows children and infants spend considerably less time on the coast than adults. Their occupancy is more weighted towards the sandy foreshore than the stoney storm beach. Children and infants would have fewer casual encounters with discrete items than adults due to their lower time on the coast, and weighting of occupancy towards the foreshore, where the spatial density of discrete items would be lower compared with the cliffs, talus and storm beach. Children and infants might have higher inadvertent ingestion rates from handling discrete items than adults [57], however loose surface contamination would be low on discrete items on the foreshore due to repeated wave washing. Therefore, we conclude that adult recreational users of the coast would have similar or higher doses from occupancy of the coast and encounters with discrete items than the children and infants with the highest occupancies.

Potential doses to visitors

The calculated total annual effective dose for adults is a little higher than the Recreational PRP (dog walker, Table 10.6 and Table 10.7), but still less than 20 μSv. This difference is not significant given the conjectural nature of the Visitor PRP, and cautious habit and exposure assumptions. The Visitor PRP dose from general occupancy is lower than the Recreational PRP. However, they receive a larger dose from encounters with discrete items and the dose from discrete items is a larger percentage of the total dose compared with the Recreational PRP. This is expected as the assumed habits are deliberately weighted towards having an interest in DIs.

Doses to visitor children might be higher than to adults, as they might have higher inadvertent ingestion rates. It could be misleading to quantify the difference given the conjectural nature of the Visitor PRP. Doses to infants might be lower than to adults and children, as they are unlikely to be clambering on the cliffs and talus.

Conclusions

The outcomes of our review build confidence that the discrete item WAC are limiting potential doses from casual encounters with discrete items to the highest occupancy users of the coast (the Recreational PRP), so that they do not disproportionately add to the doses from occupancy of the coast in front of the eroding repository, and risks remain broadly consistent with the risk guidance level assuming exposure occurs. People with alternative habits, such as visitors to the coast, might have greater exposures from discrete items. However, their habits are more uncertain. Even with arguably extreme exposure assumptions, the total dose from occupancy of the coast in front of the eroding repository and discrete items is less than 20 μSv.

The expectation value of the dose rate from discrete items is likely to be similar to, or a little higher than the average dose rate from LLW and ILW exposed on the coast. The expectation value of the dose rate from discrete items could increase from the current value if ILW is disposed. If this occurred, the potential additional dose from casual encounters with discrete items would become more significant. This is an important consideration for ongoing monitoring of the effectiveness of the discrete item WAC if ILW is disposed in the vaults. It is further considered in the '*Implementation*' report [17].

10.4.1.3 Analysis of Discrete Items in Existing Waste

In Subsection 10.2.5 we describe the findings of our retrospective assessment of active particles in existing waste, and our preferred (BAT) management options. The potential for discrete items in existing waste was assessed at the same time, and preferred management options identified [124]. Our findings are summarised below.

The activity levels in some trench disposals indicate they waste could contain discrete items with activities above the DILs (i.e. SoF >1). However, information in the disposal records is limited, so it is not possible to determine whether individual waste items would exceed the DILs. Given that it is uncertain whether individual waste items would exceed the DILs, and the difficulties in locating and retrieving specific disposals (as previously discussed for active particles), the preferred option for managing these disposals is to leave them in the trenches.

Most consignments disposed in Vault 8 are unlikely to contain discrete items or contain discrete items that meet the WAC. Our review identified 15 consignments containing a higher level of activity that probably contain discrete items that exceed the DILs, for waste items weighing 100 kg or more. Some of these containers are in top of stack positions. Given the difficulties in retrieving the containers, and that they have already been grouted so discrete items cannot readily be consigned to a different route, the preferred option is to leave the containers in their current positions in Vault 8.

Three grouted containers, and three ungrouted containers in Vault 9 do not meet the discrete item WAC. These all contain pond skips from Bradwell. The BAT outcomes for these containers were not clear.

Disposal to a commercial landfill Permitted to accept Low Activity LLW was not possible because the container activities are too high. Retrieval and repackaging were ruled out. Although three of the containers are not grouted, the pond skips had been filled with grouted desiccant to make efficient use of the container volume. This makes retrieval and repackaging of individual items impracticable.

Our assessment concluded that the containers should either be disposed of to the existing LLWR vaults or stored onsite pending disposal an alternative facility to be constructed at the LLWR site. It was inconclusive as to which of these options was preferred. It was concluded that the containers should be grouted, to reduce operational doses and improve containment, monitored and protected from the elements in the interim.

This outcome was reviewed in 2021 [130] and again in 2026 [132]. The latest review concluded that the containers should continue to be stored in Vault 9. A long list of options to act now should be developed over the next year. The BAT assessment should be reviewed again at the earlier of relevant options being identified in the long list, updates to the DILs in the WAC, the NDA reaching a decision on further near-surface disposal facilities, and five years.

10.4.1.4 Results of Updated Activity Limit Calculations

Casual encounters with discrete items during recreational use of the coast

Discrete item activity limit calculations were updated based on the results of the reviews, using updated data, e.g. dose coefficients [57]. The results of the reviews indicate the discrete item activity limits are successfully fulfilling their objectives. There is no need to change the calculation approach or the weight categories. The earliest potential time for exposure of discrete items on the coast, and therefore the amount of radioactive decay assumed, is as previously described for low-activity sealed sources (Subsection 10.3.3).

Radionuclides groups A, B1, B2 and C were retained, and the existing activity limits were considered to still be appropriate. Some radionuclides could be moved to less restrictive groups. Only Cl-36 might be moved to a more restrictive group. This change is due to updates to the internationally recommended dose coefficients.

If the repository is Permitted to receive ILW, it might be appropriate to remove the activity limit truncation from Group C. From the perspective of long-term safety, the activity limits for Group C could be increased and some radionuclides in Group C could be moved to a new Group D with even higher activity limits. However, these changes would have implications for doses to workers, other people onsite and people offsite during the PoA, and site infrastructure requirements. These factors are being evaluated separately. They are controlled by other aspects of the WAC.

Discrete items exposed by human intrusion and scavenged from the eroding repository

Inadvertent human intrusion into the repository cap might exhume discrete items. However, the volume of waste that could be exhumed by boreholes, deep trial pits or other deep excavations would be small, limiting the potential size of discrete items to be exposed.

While boreholes could potentially exhume waste from anywhere in the repository, deep trial pits or deep excavations, such as those for a sewer system or septic tank, would only intercept the top of the waste stacks in certain areas. Again, this limits the potential for discrete items to be exhumed.

Boreholes could only exhume small discrete items or parts of discrete items. Deeper excavations might exhume larger items than boreholes but are unlikely to exhume large discrete items. Any discrete items that are exhumed are likely to be damaged and not attractive, limiting the potential for close inspection. Therefore, we do not calculate discrete item activity limits for human intrusion into the repository cap. In Subsection 10.5.3 we consider container and sub-container scale heterogeneity in the waste, and the potential doses to intruders from 'hotspots'. Damaged discrete items that are not attractive and do not invite close inspection are equivalent to a hotspot.

We do calculate activity limits for discrete items retrieved while scavenging material from the eroding repository. Three situations are assessed:

- A scavenger picks up a small item (up to 100 g) and puts it in their pocket or bag. They are exposed to the object at close proximity (0.3 m) for 16 hours.
- A scavenger picks up a small item (up to 100 g) and displays it in their home. They are exposed at a distance of one metre for 4 hours a day throughout the year (total 1461 hours).
- A scavenger finds a medium size item (100 g to 10+ kg). These object sizes correspond to small instrumentation objects or machinery components. The item is taken home and closely examined in a workshop or garage for two hours. The object is then placed on a shelf in the garage or workshop, after noting that the object has no functional value or ability to be repaired. The scavenger continues to be exposed at a distance of one metre for two hours per day throughout the year (total 730.5 hours).

The first two events are especially unlikely because:

- the proportion of discrete items in this weight range is very small (Figure 10.1);
- waste items from operational plant and decommissioning will not have direct value or be particularly interesting or novel;
- waste will have experienced a degree of degradation in the repository, which will accelerate upon exposure, particularly for small items, further reducing the potential for scavenging.

We assume that a person could undertake scavenging on multiple years, so the dose guidance level of 3 mSv is appropriate. It is unclear whether a person would take a discrete item home every year or keep a discrete item for a prolonged period of time. Therefore, the dose guidance level of 20 mSv might be appropriate for these events.

Simple logical arguments show that the activity limits calculated for casual encounters with discrete items, while making recreational use of the coast, will tend to be more restrictive than those for retrieving items while scavenging:

- The calculated discrete item activity limits for casual encounters limit the dose from a close encounter (at 0.3 m), lasting 1 hour, to 20 µSv.
- For 1 hour exposure, the activity limits for retrieving discrete items while scavenging would be a factor of 20 mSv / 20 µSv = 1,000 higher than those for casual encounters.

- The exposure duration is only more than factor of 1,000 higher in the second event, i.e. 1,461 hours / 1 hour = 1,461. However, the exposure distance in the second event is 1 m rather than 0.3 m. Dose rates at 0.3 m are a factor of 3 to 10 higher than at 1 m [25].
- Therefore, we would expect the activity limits for the second event to be a factor of 0.1 * 1,461 / 1,000 = 0.15, to 0.3 * 1,461 / 1,000 = 0.44, of those for casual encounters.

If a scavenger takes a discrete item home every year or keeps discrete items for a prolonged period, then the dose guidance level of 3 mSv for long-term exposures is more appropriate. The scavenger would receive a dose from the 50 hours a year spent scavenging and from the item taken home. Therefore, we have undertaken some cautious discrete item activity limit calculations for the three situations described above. These calculations limit dose from discrete items to 3 mSv less the dose from the 50 hours spent scavenging.

The dose from the 50 hours spent scavenging is uncertain, as the inventory in future disposals is uncertain. However, the significance of this uncertainty is limited because this is a cautious approach to a stylised exposure situation. The dose from 50 hours spent scavenging assumes disposal of ILW in the vaults, which leads to higher doses than disposal of LLW only [24].

The '*Implementation'* report [17] compares the activity limits from these more cautious scavenging calculations with the activity limits calculated for casual encounters [25]. The calculated activity limits are typically similar, or the limits for casual encounters are more restrictive. The '*Implementation'* report provides more details and describes the activity limits that will be included in the WAC.

10.5 Container Scale Heterogeneity

Heterogeneity at the container scale is important for the potential impacts from radioactive gas and inadvertent human intrusion, so it is addressed in these assessments.

10.5.1 Radon Gas Assessment

The risk from radon depends on the area-weighted averaged dose across the repository cap. Heterogeneity at the sub-trench and sub-vault scale is relevant to calculating the area-weighted average dose. We also assess the effects of heterogeneity at smaller length scales, to understand the potential spatial variation in doses to individual houses. We use the results to determine whether controls on heterogeneity at smaller length scales are needed to limit heterogeneity in doses to individual houses. The relevant length scale for potential controls is an individual container.

Radon gas has a short half-life, so it decays considerably as it migrates through the profile fill and cap. When the cap geomembrane is intact, radon can only migrate through small defects (pinholes, holes, tears) in the geomembrane or open gas vents. The radon gas flux varies significantly with the amount of bulk gas that advects radon, and pathlength from the container to the surface of the cap via a defect or vent.

The mathematical model for transport of radon gas to a defect or open vent, and supporting transport calculations [26], show the contributions of individual containers to the radon flux. The contribution decreases within increasing distance from the defect or vent.

It is not practicable to model every individual trench bay or every container. Also, existing containers in Vault 9 are not in their final disposal positions, the contents of individual future containers are unknown, and the locations of small defects in the cap geomembrane are unknown.

To address this, we have identified potential controls on heterogeneity to limit the potential impacts from radon gas. These potential controls include potential limits on the Ra-226 content of individual containers, to limit the potential radon flux from a container under a defect. These options are taken forward by the '*Implementation'* report [17].

10.5.2 C-14 Gas Assessment

The trenches contain little C-14. Most of the C-14 is in the vaults. The vault wastes are isolated by the containers, container protection measures and passive drainage arrangements, which direct infiltration through the cap away from the waste, and limit mixing of waters between adjacent vaults. It is not possible to model the evolution of the waste and associated C-14 gas generation in every individual container. Therefore, the containers are grouped into several types, reflecting the waste type, rate and forms of release of C-14.

Heterogeneity from container to container is not important for migration of C-14 gas to the cap surface or potential doses. Larger scales of heterogeneity are relevant to migration of C-14 and potential doses. They are discussed in Subsection 10.6.

10.5.3 Human Intrusion Assessment

The current WAC [121] require the total specific activity of a waste consignment to be a reasonable reflection of the activity of the waste across the volume of the waste. This requirement precludes consciously 'diluting' high activity waste items with lower activity waste, so the specific activity of a consignment meets the 4 GBq t^{-1} alpha and 12 GBq t^{-1} beta/gamma limits for LLW.

We have explored the types of quantitative controls that could be placed on specific volumes of waste within an individual container to more rigidly control heterogeneity. For example, these controls might comprise activity concentration limits for all radionuclides, or total alpha and total beta/gamma in each cubic metre of waste. Discrete item activity limits already provide controls on heterogeneity within a container for relevant waste items. Additional controls would only be needed for situations where a person could be exposed to a small volume of waste with a relatively high activity. Only human intrusion events could lead to these situations.

Most of the potential human intrusion events would lead to exposure and mixing of waste from multiple containers. However, borehole analysts and coastal scavengers could be exposed to small volumes of undiluted waste with relatively high activities. This could lead to higher doses than implied by the container activity limits.

- Although a borehole drilled into the vaults would exhume waste from multiple containers, a laboratory analyst examining borehole core or exhumed material could be exposed to material from individual containers [134] [135].
- People scavenging waste from the eroding repository could potentially spend some of their time in proximity to some small volume, relatively high activity waste that is not a discrete item. For example, cemented sludges or degraded PCM waste disgorging from a container.

We have calculated sub-container scale activity limits for these events. The limiting event is radionuclide specific [135].

These limits would introduce extra characterisation requirements for waste producers, increasing complexity, cost and potentially worker doses. Approaches to treating multiple small waste items within a cubic metre and larger items with volume greater than a cubic metre would need to be defined. We have not discussed the practicalities with waste producers at this stage.

If such limits were introduced, they might need to be aligned with those for discrete items (Subsection 10.4.1.4), as the discrete item activity limits already provide controls on heterogeneity within a container for the relevant waste items. Alignment could include removing short-lived radionuclides from discrete item control and removal of the current truncation of the discrete item activity limit for Group C radionuclides. The requirement for controls on heterogeneity within a container is further discussed in the '*Implementation'* report [17].

10.6 Sub Trench and Sub Vault Scale Heterogeneity

Heterogeneity at the sub trench and sub vault scale is important for the potential impacts from coastal erosion, radioactive gas and inadvertent human intrusion, so it is included in these assessments.

10.6.1 Coastal Erosion

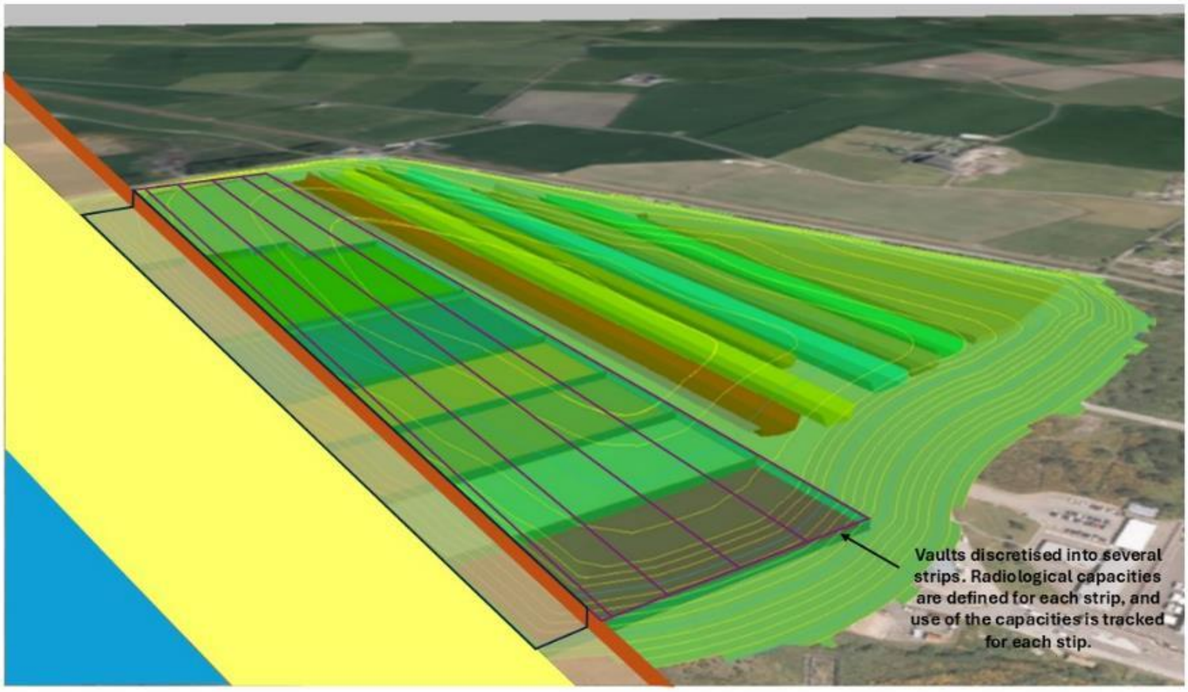
As the coast erodes it is expected to retain its current alignment [48]. The coastal erosion front will be orientated approximately parallel with the long axis of the vaults and trenches. Therefore, the eastern sides of the vaults will be exposed first, followed by the western sides, and then each trench in turn.

In our assessment model the trenches and vaults are represented using a 2D (plan view) grid. The grid is discretised to respect the locations of key radionuclides for the potential impacts from radon gas (i.e. Ra-226), coastal erosion and inadvertent human intrusion. The grid also respects variations in container stack heights in the vaults, and therefore variations in the amount of waste exposed and eroded as the erosion front passes through the repository.

People making recreational use of the coast are expected to occupy the full length of the coast adjacent to the eroding repository. There are no reasons to assume preferential occupancy of certain areas. Therefore, risks depend on the average activity exposed in along the erosion front. However, the model provides insight into the spatial variations in environmental radioactivity, and therefore the potential variability in risks associated with heterogeneous occupancy. Comparing the north, mid and south cliff sections, the reference case peak concentrations of Nb-94 and Pu-239 in the eroding vaults, and Th-232 in the eroding trenches, vary between around a factor of three higher than the length weighted mean to much more than a factor of three below the mean.

Heterogeneity perpendicular to the erosion front (i.e. in the direction of erosion) is more important, as it affects the wastes that are being exposed and eroded at a given time, and therefore risks from occupying the coast adjacent to the repository, and the fluxes of radionuclides to the marine environment (marine foodstuffs pathway). This is included in our assessment model and is reflected in changes in the calculated risks from coastal erosion as the erosion front moves through the repository.

Radiological capacities are calculated assuming uniform activity concentrations in the vaults. Therefore, they include the effects of waste stack height variations, but do not include the effects of heterogeneity in activities in the direction of erosion. The potential to dispose of certain ILW in the vaults increases the potential heterogeneity in activities in the direction of erosion, compared with disposal of only LLW. We have identified that a potential control for the post important radionuclides is to divide the total radiological capacities between several coast parallel strips and then manage capacity use in each strip (Figure 10.4). This is described in more the detail in the *'Implementation'* report [17].



Vaults discretised into several strips. Radiological capacities are defined for each strip, and use of the capacities is tracked for each stip.

Figure 10.4: Discretisation of the vaults into coast parallel strips to manage sub-vault scale spatial heterogeneity in the inventory and radiological hazard for key radionuclides in the coastal erosion pathway assessment

10.6.2 Radon Gas

As described previously, the post-PoA radon gas assessment uses the same assessment model grid, and gridded inventory, as the coastal erosion and human intrusion assessments. This reflects the major spatial variations in the Ra-226 inventory and bulk gas generation. It is not practicable to model smaller scale variations because future container contents and locations, and the numbers and locations of potential defects (small holes) in the cap geomembrane cannot be known.

The potential receptors are people occupying houses built on the cap. While the cap geomembrane is intact, there will only be radon pathways to houses above defects in the geomembrane or houses over an open vent. There will not be radon pathways to houses in other locations on the cap. The assessment model grid captures the major spatial variations in the radon gas fluxes and enables the potential doses from radon gas to be calculated for houses in each grid cell that are above a defect, and for houses that are above an open vent (radon gas from multiple grid cells may migrate to an open vent).

With the vent(s) closed, the expectation value of dose for the repository is the area-weighted average dose for the whole cap, including houses with a radon pathway and houses without a radon pathway. The expectation values of doses for the whole area of the cap, and the doses for houses that are above an open vent, are used to calculate the risks from radon gas with the vent(s) closed and open, respectively. The risk also includes the probability of the exposure situation.

10.6.3 C-14-bearing Gas

The trenches contain little C-14. Most of the C-14 is in the vaults. Therefore, the C-14 gas assessment model discretises the repository into the trenches and individual vaults. Releases of C-14-bearing gas are expected to be different from existing and committed containers and future containers. If ILW is disposed in the vaults, then shielded modules are anticipated to have relatively high concentrations of C-14. Therefore, the vaults are further discretised to represent existing and committed containers in Vault 9, and future containers in Vault 9 and future vaults, and shielded modules.

The potential receptors are smallholdings on the cap. Smallholdings are expected to include a house (75 m^2), adjacent kitchen garden (500 m^2) and a grazing area. Total area 0.5 ha to 1 ha. Therefore, the total area occupied by a smallholding is a substantial fraction of a vault. Noting that a smallholding, or smallholdings, could be located anywhere on the cap, and the layout of each smallholding is uncertain, the assessment model captures the potential major spatial variations in C-14 gas fluxes and doses to smallholdings in different locations on the cap.

Doses are calculated for smallholdings located over the trenches, each vault, existing and committed containers in Vault 9, future containers in Vault 9 and shielded modules in each vault. The area weighted average dose gives the expectation value of the dose for the whole repository. The expectation value of the dose for the whole repository underpins calculation of the repository risk from C-14 gas. The risk also includes the probability of the exposure situation.

10.6.4 Human Intrusion

Potential human intrusion events would only expose a small fraction of the waste in a trench or vault. Aircraft impact could expose a larger amount of waste, but this is a very unlikely event. As noted previously, all the potential human intrusion events would lead to exposure and mixing of waste from multiple containers. Some events would lead to exposure and mixing of waste from more containers than others. The total thickness of the cap and profile fill varies over the repository. Some events could only potentially exposure waste in the areas where the total cover thickness is at or close to the minimum.

The assessment model grid developed for the human intrusion, radon and coastal erosion assessments describes the major spatial variations in the key radionuclides for these pathways. It also describes spatial variations in the total thickness of the cap and profile material (Figure 6.7). Therefore, the assessment model grid underpins the description of which events are plausible for each sub-vault and sub-trench area and gives the dose from intruding into that area.

There will be spatial variations in radionuclide concentrations within the area represented by each grid cell. While individual consignments within a grid cell might have higher concentrations of more than one radionuclide, the pattern of spatial heterogeneity within a grid cell are typically expected to be different for each radionuclide. Therefore, the calculations give the expectation value of dose for intrusion into the grid cell.

Vault 8 disposal records are available for each container, and the location of each container is known. The disposal records were used to calculate the inventory in each waste stack and each section of the vault (the vault was divided into small sections to manage disposal operations). The inventory in each waste stack and section was used to explore the potential bias in the main assessment model due to averaging inventories over grid cells.

Borehole intrusion is the only event that could penetrate the full thickness of the waste. We have calculated doses to borehole workers and laboratory analysts from intrusion into each section. Doses to laboratory analysis are dominated by one exposure pathway and one radionuclide (Th-232), while borehole workers are exposed by a wider range of pathways. Therefore, doses to laboratory analysts are more sensitive to heterogeneity (hotspots) than borehole workers.

The peak doses to laboratory analysis and borehole workers from intrusion into all sections are far below the 20 mSv level for short duration exposures. For the laboratory analyst, the peak dose calculated by the main human intrusion assessment (Subsection 8.5.2) is above the 75th percentile of the doses calculated at a sub-vault scale. It is close to the mean of the results by Vault 8 section. For the borehole worker, the peak dose calculated by the main human intrusion assessment is close to the median dose over the individual vault sections.

We have also calculated doses to laboratory analysts from a single borehole intruding into a waste stack. However, it is not plausible that all the boreholes drilled into the repository would intersect the same stack. The results show that if one borehole intersects the waste stack giving rise to the highest doses in Vault 8, the total dose from all the boreholes is unlikely to be above the total dose from drilling boreholes into the worst section.

Overall, while it is possible to intrude into specific containers (or trench disposals) that would give rise to higher doses than calculated in the main assessment model, the probability of this is low. The main assessment model results provide a better indication of the doses that would be expected from intrusion into different areas of the vaults and trenches. Even in the worst case, the calculated dose remains consistent with the dose guidance level.

10.7 Trench and Vault Scale Heterogeneity

Trench and vault scale heterogeneity is explicitly included in the coastal erosion and gas pathway assessments, and in the human intrusion assessment, as described in the previous subsection. In these assessments, the trenches and vaults are represented separately, as are sub-areas of the trenches and vaults.

Discretisation of the groundwater pathway assessment model is described in Subsection 5.4. The trenches and vaults are represented individually in the assessment model. The discretisation of the repository (Figure 5.10, Figure 5.11) reflects the engineering design and represents the major heterogeneities in potential water levels and flows within the repository. The water saturations, flow rates and flow patterns in the repository affect the locations of releases and fluxes to the groundwater pathway. The release locations and fluxes affect the spatial distributions and concentrations of radionuclides in groundwater, and the length of the groundwater pathway to the coast.

Representing the vaults individually captures the major spatial heterogeneities in their radionuclide inventories. Mixing of waters in the vaults, drainage blankets and groundwater pathway will reduce heterogeneities in radionuclide concentrations, so there is no benefit to representing vault disposals in more detail, even though more detailed inventory information is available for existing disposals.

In the assessment model, the radionuclide inventory in each trench is distributed uniformly along its length. Post-PoA, flows are predominantly downwards through the bases of the trenches, so the groundwater pathlength to the coast is different for each trench. Each trench is discretised along its length into sections corresponding to the adjacent vaults (Figure 5.10). This is to support representation of strip capping and flow calculations using the CFM, rather than to calculate spatial heterogeneities in radionuclide releases to the groundwater pathway.

The groundwater flow and transport paths included in the assessment model (Figure 5.12) explicitly represent the major heterogeneities in the radionuclide concentrations in the geosphere, resulting from spatial heterogeneity in the radionuclide inventory, spatial and temporal evolution of water flows and radionuclide fluxes out of the repository, and evolution of the groundwater pathway (flow areas, velocities and pathlengths). Therefore, the assessment model represents the variations in radionuclide fluxes to the marine environment over time.

Spatial variations in the distribution of radionuclides along and across the groundwater pathway between the repository and the coast have little impact on calculated risks for the well biosphere pathway [65]. The calculated risks depend on the expectation values of the radionuclide concentrations in the contaminated area, i.e. the area-weighted average concentrations. Therefore, the total amount of each radionuclide in the contaminated area is the key control on risk, not the spatial distribution within the contaminated area.

There are small scale heterogeneities in the geology and hydrogeological properties of the geological units and subunits. The effects of small-scale heterogeneities in the geosphere on flows are represented implicitly in the assessment model, as the flow rates output from the ConnectFlow model are based on a single average hydraulic conductivity for each lithofacies unit. The effects of small-scale heterogeneities on contaminant transport are represented explicitly by the longitudinal dispersion length specified in the assessment model, and the cross-sectional areas of the pathways derived from the groundwater flow model, which include the effects of transverse dispersion. The effects of small-scale heterogeneities in the geosphere are discussed further in the next Subsection (10.8).

10.8 Heterogeneities in the Geosphere

There are heterogeneities in the properties of the lithofacies units at smaller length scales than cannot be fully characterised by site investigation or represented in the assessment model. We have investigated the small-scale heterogeneities that could be present and their potential implications for groundwater flow, radionuclide transport and concentrations, and well biosphere pathway risks.

We have investigated the potential implications of small-scale heterogeneity for groundwater flow using two approaches. In the 2011 ESC [136] we used lithological data to calculate distributions of the hydraulic conductivity of the B3 unit at length scales of one metre, and then one hundred metres. Similar calculations were undertaken to provide distributions of hydraulic conductivity for the B2 and C lithofacies units. The B2, B3 and C units in the ConnectFlow groundwater model were discretised using a 100 m 100 m grid (in plan) and hydraulic conductivities assigned to each grid cell by randomly sampling the relevant distribution.

Ten realisations of the ConnectFlow model were developed and used to calculate steady state groundwater flows. The key findings were as follows.

- Including heterogeneity at the scale of 100 m 100 m affected the model calibration against observations. The modelled groundwater heads in two of the realisations were sufficiently different from observations that potentially these realisations are not credible and could be screened out.
- Inclusions of smaller scale heterogeneity (i.e. at a scale of 100 m 100 m) resulted in focussing of flows through areas where the more permeable deposits are connected. This is illustrated by pathline analysis (Figure 10.5).

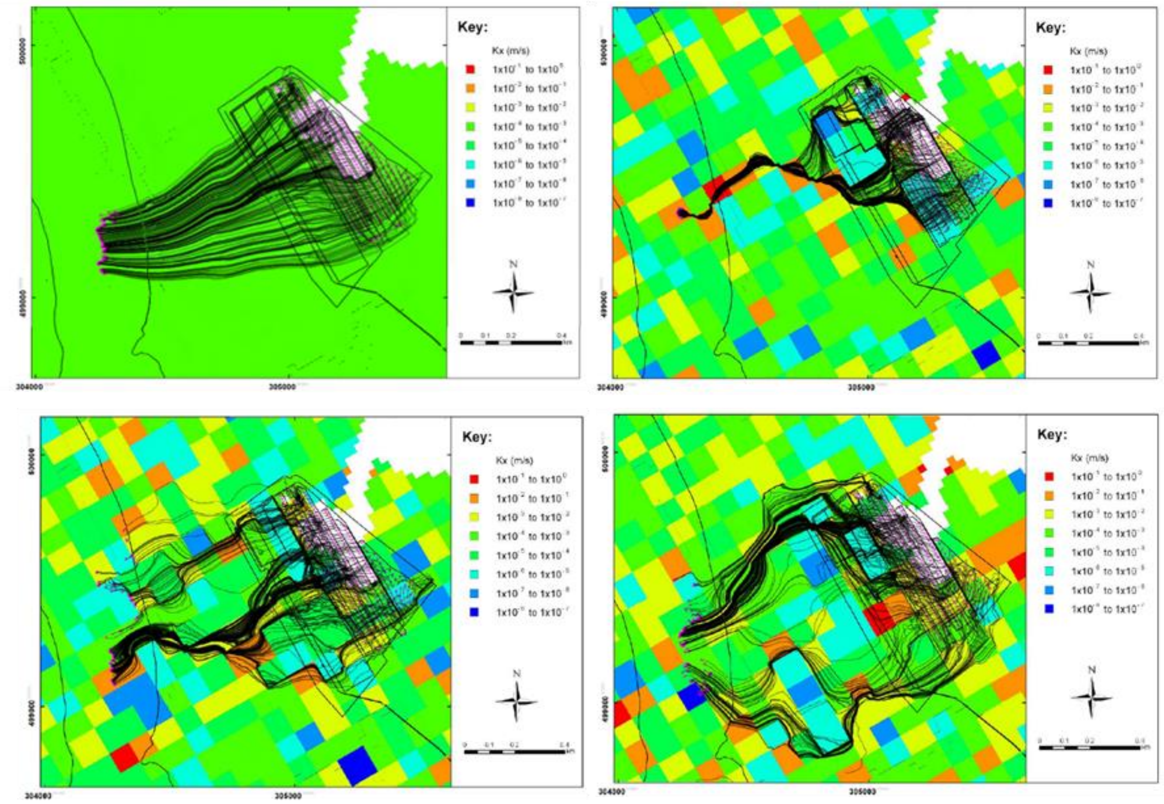


Figure 10.5: 2011 ESC groundwater flow model pathlines from the trenches with homogeneous B2, B3 and C unit properties (top left), and three ConnectFlow model realisations with heterogeneity at the scale of 100 m 100 m [136]

Our understanding of the environments the geological units were deposited in has improved since the 2011 ESC. This improves our understanding of the types of small-scale features that could be present in each unit. We developed geostatistical models for the B3 unit that are consistent with the types of small-scale features that could be present in the unit. We then explored the implications for groundwater flow using ConnectFlow [137].

The B3 unit is subdivided in the geological model into subunits, including a particularly permeable subunit composed of sands and gravels (B3L), which is more prominent close to the LLWR site, and a less permeable unit composed of sands and some silts (B3SSU) lying above this, which is more prominent offshore (Figure 5.8).

Five geostatistical models were used to generate an ensemble of 1,000 spatial random fields for the B3 unit, including the B3L and B3SSU subunits. Each of these random fields was then mapped into the LLWR steady-state regional groundwater model to give a corresponding ensemble of 1,000 ConnectFlow models. Ensemble runs were performed, and the resulting output files were post-processed and collated together to give a probabilistic set of results.

The key findings from this work are broadly consistent with the key findings from our work for the 2011 ESC, building additional confidence in our understanding. Small scale heterogeneity in the B3 unit can focus flows through areas and features with higher permeability, for example coarse grained deposits along the courses of palaeo-channels. However, models with high permeability features in the B3L outwash only match the observed groundwater head data when a relatively low permeability is assumed for the B3SSU sands. Otherwise, the transmissivity of the B3 unit is too high, resulting in calculated groundwater heads that are lower than observed.

This means that in models which closely reproduce the observed groundwater heads the bulk flow rates are not significantly different to a reference case model where the properties of the geological units are homogeneous. Consequently, uncertainty in the average permeability of the B3 unit has a bigger impact on the calculated groundwater flow rates that are passed to the groundwater pathway assessment, than the effects of small-scale heterogeneity that are not explicitly represented in the ConnectFlow model [137].

Flow focusing may increase spatial variability in contaminant concentrations in the B3L unit compared with a homogeneous model. However, as discussed previously, spatial heterogeneity in groundwater concentrations is of limited importance for well biosphere pathway risks [65].

Patterns of geosphere heterogeneity that result in the most extreme flow focussing situations, such as top right in Figure 10.5, may increase the expectation values of radionuclide concentrations compared with a homogeneous unit, if heterogeneity reduces the amount of dilution. However, the impact on calculated risks will be small, as the increases in concentrations (and therefore doses) tend to be offset by the smaller contaminated area, and lower probability of a well existing in the contaminated area [65].

We have also quantitatively assessed the contributions of variability and uncertainty to the distribution of radionuclide concentrations (and doses) about their expectation values [65]. In order of greatest effect to smallest effect, these are:

- variability of radionuclide concentrations in a contaminated area with homogeneous properties;
- uncertainty in parameterisation of the assessment model;
- uncertainty in small scale heterogeneity in the B3 unit;
- variability in the location of the inventory in the vaults.

This shows that the potential impacts of small-scale heterogeneity on the range of doses that could potentially be received from well water is smaller than other factors.

The outcomes of these studies have been used to inform parameterisation of the probabilistic reference case for the groundwater pathway (Subsection 5.5.4).

10.9 Summary

Our assessments address the effects of spatial heterogeneity in radioactivity at relevant length scales. The chosen approach and degree of spatial discretisation depend on the characteristics and projected evolution of the waste, pathway and receptor. For example, there is no benefit in attempting to model every waste item or container individually if the receptor is expected to be exposed to radioactivity originating from many waste items and containers, that becomes well mixed and is distributed over a large area in the biosphere.

Small-scale heterogeneities are important for the coastal erosion and human intrusion pathways, but they cannot be represented in assessment models of the whole repository. Information such as the activity on individual future waste items, the contents of individual future containers, and where containers would be placed in the vaults cannot be known. Evolution cannot be projected or modelled for every item, container or waste stack. Even if the required information was available, the models would be so complex they would not be tractable.

In these situations, side models have been used to assess generic waste items, generic containers, and to consider individual waste stacks and small vault sub-areas in Vault 8. We have used the results of our main assessment models and side models to identify potential repository management and waste acceptance controls, that would ensure that smaller scale heterogeneities could not results in risks that are significantly higher than calculated by our main assessment models. In this context, significant refers to calculated doses and risks that both, exceed the range of uncertainty in our best estimate doses and risks, and exceed the risk and dose guidance levels.

Our WAC already include quantitative controls on radionuclide activities in active particles, sealed sources, and discrete items, and a qualitative control on heterogeneity across all waste items in a consignment. We plan to introduce an additional control on the maximum amount of Ra-226 in a container, and to manage heterogeneities in the inventories of key radionuclides for the coastal erosion pathway at the sub-vault scale (Figure 10.4). We are also considering whether quantitative control on heterogeneity across all waste items in a consignment would be practicable and beneficial. Updates to the existing controls and new controls are discussed in the *'Implementation'* report [17].

11 Combined Pathway Risks

Our pathway assessments identify people who could be exposed to radionuclides. There are instances where the same people could be exposed to radionuclides from more than one pathway. This section assesses the potential risks from exposure to radionuclides from more than one pathway.

11.1 General

Our individual pathway assessments calculate risks to a representative member of the people at greatest risk. Habits are based on the 95th percentile of local and national habits data where available and relevant (Subsection 2.8). Adding the risks calculated for different pathways implies the people have habits that make them representative of the most exposed people for more than one pathway. These would likely be extreme habits, for example people making maximum recreational use of the coast in front of the eroding repository and consuming maximum amounts of marine foodstuffs, all caught locally. While one or two individuals might be exposed by more than one pathway, and might even have such extreme habits, it is less likely that a representative person would be exposed by more than one pathway or have such extreme habits. This is consistent with guidance from the ICRP [47] which states that extreme practices should not be adopted when determining the characteristics of the representative person. It is also noted that the ICRP biokinetic models are based on the characteristics of an average person and would not be applicable to a person consuming high intakes of multiple foodstuffs [138].

When appropriate, calculated risks included the probability of the exposure situation. Exposures are assumed to occur (probability equals one) for the coastal erosion pathway but are not certain to occur for the groundwater well and gas pathways (probability less than one). The probability that a person is exposed by more than one pathway is expected to be lower than the probability that they are exposed to any of the individual pathways. Therefore, while exposure to more than one pathway can result in higher doses, the lower probability of exposure means the risk might be lower than the highest risk from the individual pathways. It is not possible to determine the probability of exposure to more than one pathway when calculating combined risks. Therefore, the combined risks calculated below are expected to be an overestimate.

For all pathways, radiological capacities and other controls in the WAC will limit the potential risks to $1\ 10^{-6}\ y^{-1}$. In the very unlikely situation that the full radiological capacity of the repository is used for two pathways, and a person is exposed via both pathways, the maximum combined risk would be $2\ 10^{-6}\ y^{-1}$. Once the low probability of exposure by both pathways is accounted for, the risk would be $<2\ 10^{-6}\ y^{-1}$. Although mathematically the risk could be in the range $1\ 10^{-6}\ y^{-1}$ to $2\ 10^{-6}\ y^{-1}$, the difference compared with the risk guidance level would likely be negligible given the uncertainties in the pathway assessments.

The following subsections:

- systematically identify situations where people could be exposed by more than one pathway;
- then calculate the combined risks for credible pathway combinations.

We do not include the human intrusion pathway in the analysis as the probabilities of these events cannot be determined and doses from these events dominate over the potential doses from the groundwater, gas and coastal erosion pathways.

11.2 Exposure Situations

It is not possible to calculate and justify the probability of exposure by more than one pathway. However, we can qualitatively identify situations where exposure by more than one pathway is credible, and situations where exposure by more than one pathway is so unlikely they do not need to be considered further. A simple matrix exploring pathway combinations is presented in Table 11.1. The potential combinations are discussed below.

For simplicity we have not included the estuary groundwater pathway in Table 11.1. Risks from the estuary pathway are many orders of magnitude below the risk guidance level, and substantially lower than the other pathways. We have also not included the stream pathway as we consider this pathway to be very unlikely, and as explained in Subsection 5.4.7.3, our assessment model for this pathway is very cautious.

Table 11.1: Potential pathway combinations (note the table is reflected about the diagonal)

	Groundwater (well)	Groundwater (marine)	Gas	Coastal erosion
Groundwater (well)	N/A	Credible	Unlikely as well would be a considerable distance from the house	Not physically possible
Groundwater (marine)	Credible	N/A	Credible	Credible
Gas	Unlikely as well would be a considerable distance from the house	Credible	N/A	Credible
Coastal erosion	Not physically possible	Credible	Credible	Recreational use of the coast and consumption of marine foodstuffs

A person cannot be exposed by the well pathway and the coastal erosion pathway at the same time. When the repository is eroding, the groundwater pathway between the repository and the coast has already been completely disrupted by coastal erosion.

Although there is no groundwater pathway while the repository is eroding, a person could be exposed to environmental radioactivity released[18] via the marine groundwater pathway and environmental radioactivity due to erosion of the repository.

A person living in a house or smallholding on the cap could potentially be exposed to radioactive gases, drink water from a well, or use water from a well to irrigate their smallholding or to provide water for animals. If the well was drilled into the facility this would be a human intrusion event, so it would need to be assessed against the dose guidance level. However, if a driller attempted to drill a well into the LLWR itself, it is expected they would recognise that they were drilling into a man-made structure, which would look like a landfill, and they would stop drilling and move to a different location. The well would not be completed.

A person living on the cap could take water from a well located between the repository and the coast, i.e. within the groundwater pathway. However, this is unlikely due to the distance between the house and the well. It is more likely the house would be close to the well.

The smallholder's house could be located between the cap and the coast, with an adjacent well and a grazing area that extends onto the cap. The smallholder's kitchen garden would likely be located next to the house [26], i.e. the kitchen garden would not be on the cap. As a major portion of the dose from C-14 gas comes from kitchen garden foodstuffs, and there might not be a radon pathway to a house that is not on the cap, the combined risks would likely be lower than for the gas pathway alone.

It is credible that people living in a house (or houses) and, or smallholding on the cap could be exposed to radioactive gas and groundwater pathway releases to the marine environment.

It is also credible that a people could be living on the cap when the repository is being disrupted by coastal erosion. The people would likely make recreational use of the coast and be exposed to radioactivity on the coast, in addition to radon and C-14 gas.

The only credible situation where people could be exposed to three pathways is the same as above, i.e. exposure to radioactive gases and radionuclides released by coastal erosion, plus exposure to radionuclides released to the coast by the marine groundwater pathway, prior to the onset of repository disruption.

The coastal erosion assessment considers the risks to people who make recreational use of the coast and risks to high-rate marine foodstuff consumers. It is conceivable that some people who spend substantial time making recreational use of the coast are also high-rate

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[18] i.e. releases that occurred before the start of disruption by coastal erosion.

marine foodstuff consumers. We have examined the results or habits surveys to see if this is likely.

The 2019 survey [106] identifies one individual (Person ID number 2912/1/1) along the West Cumbrian coast who is in the high-rate group of sea-food consumers and has high occupancy of the coast. The are a few other individuals who have high occupancy of the coast and consume marine foodstuffs, but they are not high-rate consumers. From this data and logical arguments, we have concluded the following.

- At a given time, there may be none, one or a few individuals along the West Cumbrian coast who have high occupancy of the coast and are high-rate consumers or marine seafood.
- It is uncertain whether such a person would be present at the LLWR. The probability is less than one, and this would need to be considered when calculating a risk.
- These people are unlikely to be representative persons for both occupancy of the coast and marine foodstuff consumption. They would be an outlier.

11.3 Combined Risks

11.3.1 Summing Peak Risks

Peak risks have been calculated for each of the individual pathways. For each credible pathway combination identified in Table 11.1, we consider whether it is appropriate to sum the peak risks.

The peak risk calculated for the groundwater pathway to the marine biosphere is 8.7 10^{-12} y^{-1} in the deterministic reference case, while the peak expectation value of risk is 3.4 10^{-8} y^{-1} in the probabilistic reference case. These risks are sufficiently low that the marine biosphere pathway would not significantly add to the peak risks from any other pathways, relative to the risk guidance level.

For the coastal erosion and gas pathways, the peak risks are expected to arise at different times, so it is not appropriate to simply add the peak risks. The time varying risks for the gas and coastal erosion pathways are summed in the next subsection.

The final potential pathway combination identified is recreational use of the coast and consumption of marine foodstuffs. As discussed in the previous subsection, a person who has high occupancy of the coast and is also a high-rate marine foodstuff consumer is unlikely to be a representative member of both groups, as this would imply an extreme set of habits. Therefore, it is not appropriate to sum the risks. This is consistent with guidance from reference [47].

The peak risks for recreational users of the coast and high-rate marine foodstuff consumers occur at different times. We do look at the combined risks in the next subsection to show that even if the risks are combined, they are still broadly consistent with risk guidance level.

11.3.2 Summing Time Varying Risks

Combined doses from the gas and coastal erosion pathways

Adding the risks for the gas and coastal erosion pathways implies a very cautious set of habits:

- 90% of time in a house on the cap being exposed to radon.
- 3% of time on the coast being exposed to radionuclides eroding from the repository.
- 7% of time elsewhere, e.g. working on the smallholding.

For this situation to be credible, a large portion of time would likely need to be spent working on the smallholding and less time spent indoors, reducing the risk from radon.

Notwithstanding the above, the combined reference case risks from the gas and coastal erosion pathways are shown in Figure 11.1 and Figure 11.2. Note these risks include the probability of a person living on the cap and conditionally assume that if a person is living on the cap, they make recreational use of the coast.

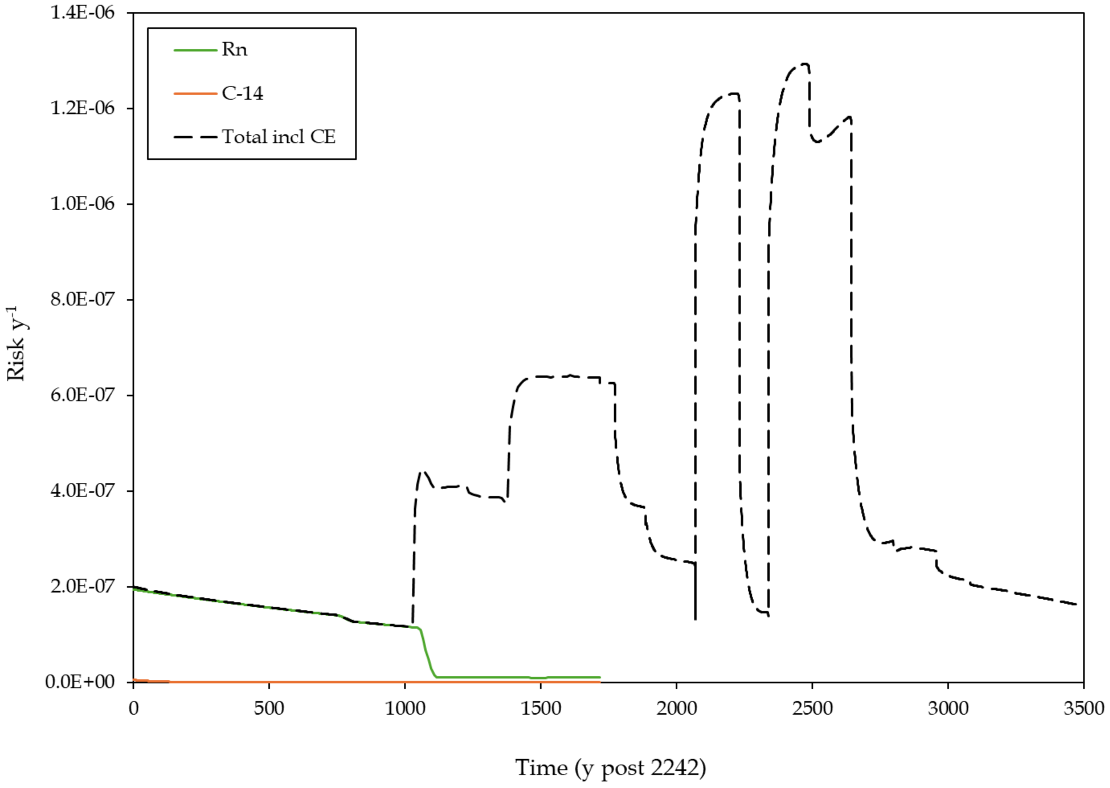

Figure 11.1: Combined risks from the gas and coastal erosion pathways with the geomembrane intact

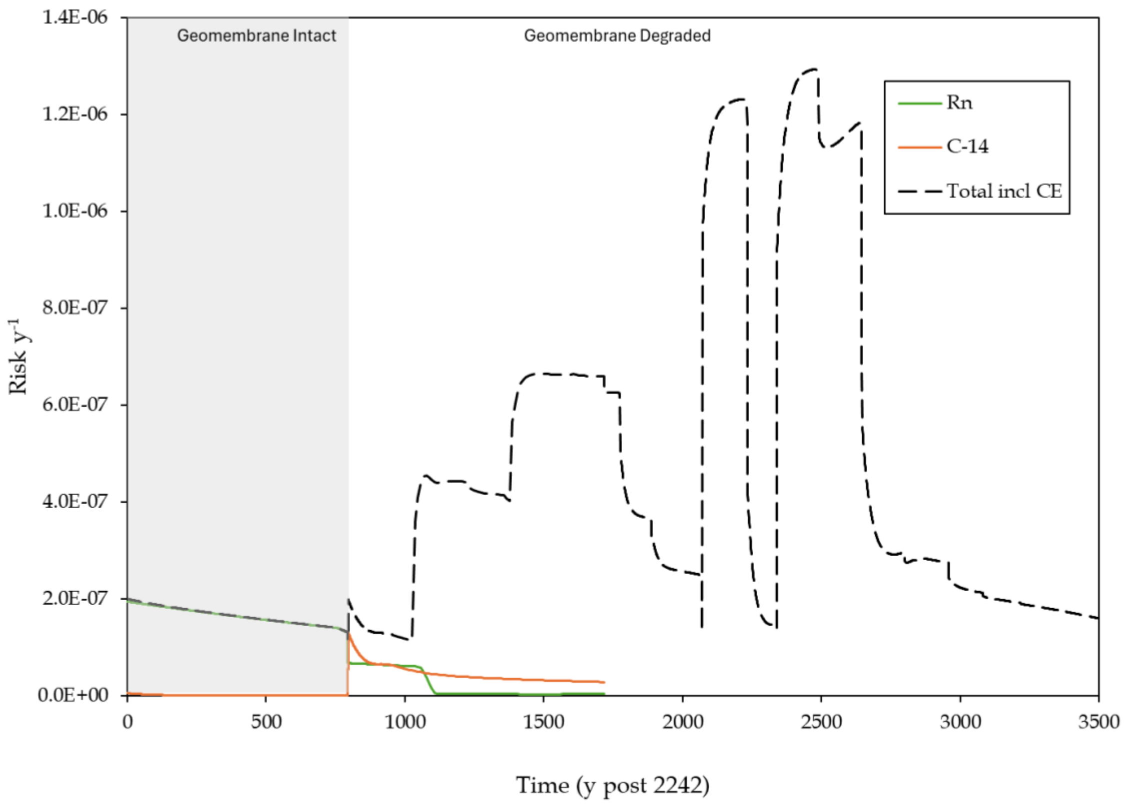

Figure 11.2: Combined risks from the gas and coastal erosion pathways with the geomembrane degraded

The peak risks from the gas and coastal erosion pathways occur at different times. The peak combined risk is far below the risk guidance level and far below the peak risk from coastal erosion. Combined peak risks are not shown for the case that the cap geomembrane is intact, and the vent is left open at the end of the PoA, because they would be lower than the cases shown (Figure 6.28).

The gas pathway assessment [26] qualitatively assessed the risks from radioactive gases during erosion of the repository, i.e. beyond 1,700 years post-2242 AD. It concluded that gas pathway risks would continue to decrease beyond 1,700 years post-2242 AD.

Breach of the repository could significantly reduce the gas overpressure, as bulk gas could readily migrate between vault container stacks, and within the cap gas collection layer, to discharge at the cliff face. C-14 gas discharging at the cliff face would be diluted and dispersed by the wind, so doses from C-14 gas are expected to fall. Similarly, doses from radon are anticipated to fall significantly, because less radon would be transported towards the surface of the cap, and advection upwards through the cap layers would be a much less significant transport process.

The erosion front is expected to be parallel with the long axis of the repository. Following breach of the repository, gas pathway doses from the trenches would fall less than doses from the vaults. The greater distance to the cliffs, and relatively low permeability of the trench wastes and natural sediments between the trenches compared with the gaps and void fill between vault container stacks, are expected to limit discharge of gas from the trenches at the cliffs. However, as erosion continues and the distance between the trenches and the cliff face decreases, the doses from the trenches would decrease.

Superimposed on the above would be the long-term decreases in doses that would occur even without disruption by coastal erosion, due to the diminishing radium and C-14 inventories remaining in the repository.

Combined doses from recreational use of the coast and high-rate consumption of marine foodstuffs

As noted previously, it is not appropriate to add the calculated risks for recreational use of the coast and high-rate consumption of marine foodstuffs. However, here we show the combined risks to provide assurance that even if the risks are added they are still broadly consistent with risk guidance level.

The combined risks from recreational use of the coast and high-rate consumption of marine foodstuffs are shown in Figure 11.1. The peak combined risk from the trenches and vaults are both a little below 2 10^{-6} y^{-1}. The probability of an individual with such extreme habits being present at the LLWR at any given time cannot be quantified but is less than one. If this probability could be quantified and included in the risk calculation, the peak risk to an individual with such extreme habits would be broadly consistent with the risk guidance level.

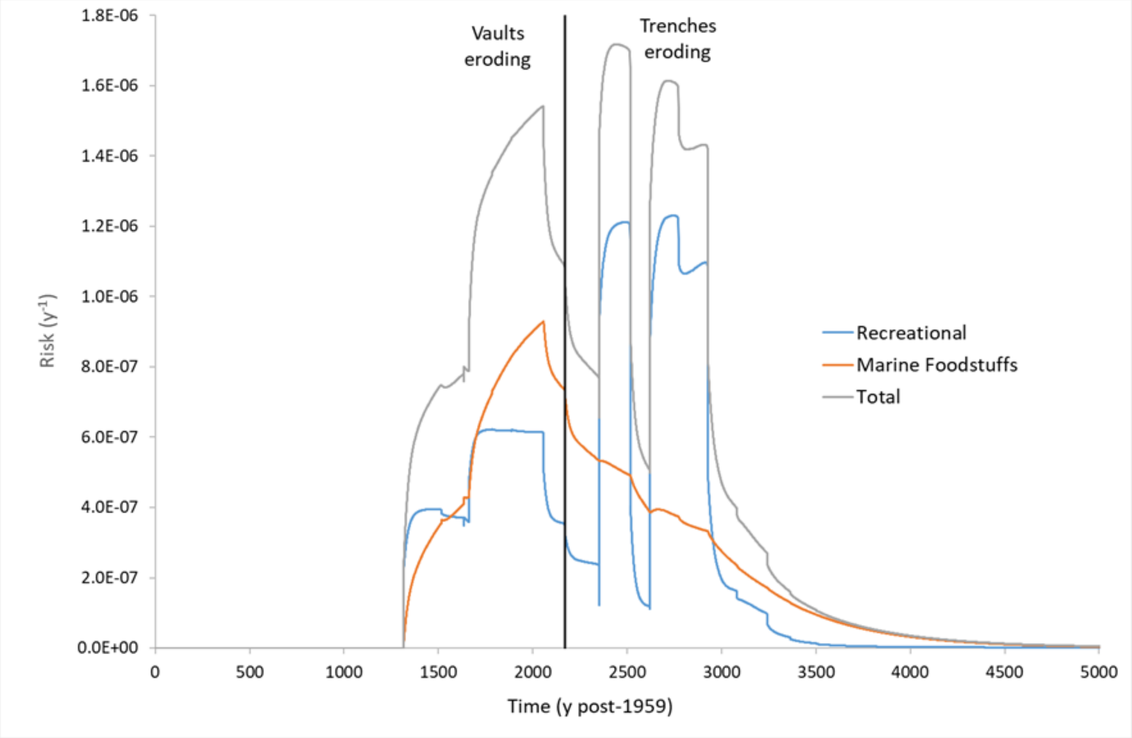

Figure 11.3: Combined risks from recreational use of the coast and high-rate consumption of marine foodstuffs

11.4 Summary

We have identified several credible situations where people could be exposed by more than one pathway. In all situations, the peak combined risks are little greater than the highest of the individual pathway risks because:

- the risk from one pathway dominate; or
- the peak risks occur at different times.

If the probability of being exposed by more than one pathway could be quantified and included in the risk calculation, then the peak combined risks might be lower than the highest of the individual pathway risks.

Summing the risks from individual pathways has negligible implications for safety. However, it makes the calculation of radiological capacities and management of use of capacities far more complicated. Therefore, we manage the repository using radiological capacities and WAC derived from the individual pathway assessments [17].

12 Conclusions

The LTRA for the 2026 ESC is mainly an evolution of that developed for the 2011 ESC, albeit with considerable updates to the conceptual and assessment models, and improved assessment, data and QA approaches. Updates to the assessments reflect changes to the future inventory, optimised repository design and development plans, and improvements to understanding of near-field conditions and long-term engineering performance. A completely new assessment has been developed for radon in response to our updated understanding of the final evolution and performance [13].

This section summarises how we build confidence in our assessments and then summarises the assessed radiation risks and doses and their consistency with the GRA Requirements R6 and R7. The key uncertainties that could most affect the outcome of our assessments are identified. Finally, this section considers the key Safety Functions (SFs) and implications for WAC, which are covered in more detail in the '*Safety Functions*' and '*Implementation*' Level 2 reports, respectively [12] [17].

12.1 Building Confidence

It is essential that we, and our stakeholders, have confidence in our assessment results. We build confidence in our assessments by:

- ensuring the people undertaking the work are suitably qualified and experienced (Subsection 2.11);
- using multidisciplinary teams to ensure all relevant factors are considered (Subsection 2.4) and then auditing our understanding against FEP lists (Subsection 2.11);
- ensuring our assessments use comprehensive, robust and consistent best practice approaches [33] and data [34]);
- using multiple lines of evidence to support our conceptual models and data, including 'real world' data such as data on the relationship between radon in houses and radon in soils, analogues for the behaviour of materials exposed in the coastal environment, and results from detailed interpretative groundwater flow and near-field biogeochemical models;
- implementing models using established software that is verified (Subsection 2.10);
- developing reference cases which describe our best understanding of the system but taking a cautious approach where the most realistic representation cannot be robustly quantified, i.e. being cautiously realistic (Subsections 2.9 and 2.5.1);
- representing spatial and temporal variability (Subsection 2.6), and important heterogeneities, in our assessment models, and developing WAC and other controls

on heterogeneities that cannot readily be projected or practically be included in our assessment models (Subsection 10);

- using a systematic approach to comprehensively identify biases and uncertainties (Subsection 2.5);
- using variant calculation cases to explore biases and uncertainties, the roles and performance of barriers which provide safety functions, and to ensure model results are robust to changes (Subsection 2.9);
- building on existing assessments and understanding the reasons for changes in our assessment results (Subsection 2.3);
- exploring a wide range of model outputs to understand the key controls on the system and ensuring models have been correctly implemented (Subsection 2.13);
- assessing risks to people who are representative of the small numbers of people at greatest risk, underpinned by local and national habits surveys, and developing narratives for the activities people might undertake on the site in the future, e.g. smallholding;
- using multiple QA approaches to check calculation and model results, including undertaking independent calculations using simpler models (Subsection 2.11);
- developing QA plans prior to undertaking the work, auditing the work against the plans, and sharing learning and best practice across our assessment teams (Subsection 2.11).
- independent peer review of our assessments (Subsection 2.11);
- ongoing, iterative maintenance and development of our assessments and the wider ESC between major submissions, including ongoing dialogue with the Environment Agency (Subsection 2.2).

Our approach is to use good science and accepted models and data to build confidence that calculated doses and risks are representative, and the facility is safe. We use monitoring data to:

- inform our conceptual models and parameterise our assessment models;
- calibrate our underpinning detailed groundwater flow model, which provides boundary conditions and flow rates for our groundwater pathway assessment models [25];
- compare against the results of our detailed biogeochemical near-field models, to build confidence in our understanding of materials inventories, repository conditions, relevant near-field processes and their rates [6];
- where possible, build confidence that environmental concentrations calculated by our assessment models are cautiously realistic.

12.2 Radiation Doses and Risks in the Long term

We have assessed the potential radiation doses and risks to people and protection of groundwater in the long term (after the period of management control) for the broad range of generally expected processes, including natural degradation and release pathways, and human intrusion.

The assessments focus on a broadly expected evolution of the local environment, taking account of uncertainties in:

- future emissions, climate and sea-level and local erosion processes;
- the properties of the waste, engineered barriers and hydrogeology;
- radionuclide release, migration and exposure processes;
- future human actions that could disturb the repository (human intrusion).

Our assessments consider the people at greatest risk from radionuclide releases in gas and groundwater, and disruption of the repository by coastal erosion, or who could receive the doses in the event of inadvertent human intrusion.

Consistency with Requirement R6: Risk guidance level

The GRA [19] requires that after the PoA, 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 1 10^{-6} y^{-1} (i.e. 1 in a million per year).

Table 12.1 presents a summary of the assessed risk via each of the 'natural' pathways, which may be compared to the risk guidance level of 1 10^{-6} y^{-1}. Results are presented for reference cases. Human intrusion and activities involving retrieval or seeking out waste exposed by coastal erosion are assessed against a dose guidance level range, so they are not included in the table.

For the groundwater well biosphere pathway, the peak risk from the deterministic reference case is more than two orders of magnitude below the risk guidance level. A well could be located anywhere in the area between the facility and the coast. Therefore, the risk depends on the expectation value of radionuclide concentrations in the contaminated area between the repository and the coast. It is not certain a well would intersect the contaminated area. Therefore, the risk also includes the annualised probability a well exists in the contaminated area.

We have also developed a probabilistic reference case for the groundwater pathway that includes parameter uncertainty, as described in Subsection 5.5.1. The expectation value of risk (i.e. the mean peak risk) for the well biosphere pathway is higher than calculated in the deterministic case but still below the risk guidance level. The peak expectation values of risk for all biosphere pathways are much higher than the peak median risks. The peak median risks are more similar to the peak risks from the deterministic reference case. This is expected given the shapes of the log-triangular PDFs used to describe uncertainty in some of the parameter values and the bimodal nature of the PDF for cap infiltration.

The differences between the means and medians of the log-triangular PDFs compound in the calculations resulting in expectation values (means) of risk that are substantially greater than the medians. The probabilistic case also includes realisations where the cap geomembrane fails during the assessment timeframe, leading to a notable increase in infiltration into the repository and releases to groundwater (this does not occur in the deterministic case). These realisations also contribute to the higher mean risks compared with the medians.

Realisations with cap performance towards the lower bound lead to releases to the stream biosphere pathway. Radionuclides released to the stream are then transported into the estuary and the marine environment, also contributing to increased mean risks for the estuary biosphere pathway and the marine biosphere pathway compared with the medians.

Risks for the gas pathway are based on a deterministic calculation. People could live anywhere on the cap, and it is not certain the potential exposure situations would occur. The probability of the exposure situation is included in the calculated risk.

People are expected to be exposed to the radioactivity remaining in the repository when it is disrupted by coastal erosion. Therefore, for the coastal erosion pathway we calculate annual doses and compare them against a dose of 20 μSv, which is equivalent to the GRA risk guidance level assuming exposure occurs [19].

For all pathways, doses and risks are calculated to individuals with locations and habits such that they are estimated to be representative of those at greatest risk (cf. GRA Requirement R6, see Subsection 1.1).

Peak risks are all consistent with the risk guidance level of $1\ 10^{-6}\ y^{-1}$. Numerically the peak annual dose from the coastal erosion pathway is slightly above 20 μSv for the trenches. However, the difference is small given the uncertainties. Risks from the trenches are dominated by disposals of Th-232. We have undertaken BAT assessments considering the potential to retrieve selected waste from the trenches, including thorium bearing waste [139]. These BAT assessments have concluded that long-term risks a little above the risk guidance level would not be sufficient to justify waste retrieval given the conventional and radiological risks of the retrieval operations.

We have also considered the potential for the same person to be exposed by more than one pathway. We have identified several credible situations where people could be exposed by more than one pathway. In all situations, the peak combined risks are little greater than the highest of the individual pathway risks because:

- the risk from one pathway dominates; or
- the peak risks occur at different times.

Table 12.1: **Peak doses and radiological risks from reference cases**

Pathway	Case	Peak risk (y^{-1}) or dose (µSv)	Key radionuclides	Notes
Groundwater well	Deterministic reference Probabilistic reference (mean)	$2\ 10^{-9}\ y^{-1}$ $4\ 10^{-7}\ y^{-1}$	I-129 Th-232, Ra-228, Th-228	Risks dominated by the trenches. Probabilistic case includes log triangular PDFs, including for inventory uncertainty. Combined with the bimodal PDF for cap infiltration, these distributions result in an expectation value of risk (mean) that is much higher than the median.
Gas (total risk from C-14 and radon)	Reference (Cap geomembrane intact)	$8\ 10^{-8}\ y^{-1}$	Rn-222	Receptor could be anywhere on the cap.
	Cap vent is not closed (Cap geomembrane intact)	$1\ 10^{-9}\ y^{-1}$	C-14	Probability of receptor being over the vent is lower than being somewhere on the cap.
Coastal erosion (recreational use of the coast)	Reference Vaults Reference Trenches	11 µSv 22 µSv	Nb-94, Pu-239 Th-228 & Ra-228 ingrown from Th-232, and U-238	None

The cap is the key barrier isolating and containing radioactivity (and non-radiological contaminants) in the waste. We have assessed the impacts of early failure of the cap geomembrane and poor performance of the cap (Table 12.2). The calculated risks do not include the probability of poor cap performance.

Poor performance of the cap could lead to water in the repository rising to the level where it discharges to the ground surface and enters surface streams. Calculated risks for the stream biosphere pathway are higher than calculated risks for the well biosphere pathway with poor cap performance. Risks for the well pathway include the annualised probability a well exists in the contaminated area, and therefore the probability that people are exposed, while risks for the stream pathway assume exposures occur (probability of one). This is a cautious assumption. Our assessment model for the stream pathway also includes other cautious assumptions (Subsection 5.4.7.3). Therefore, calculated risks for the stream pathway are expected to be an overestimate.

Table 12.2: Summary of assessed radiological risks from deterministic poor cap performance cases

Pathway	Case	Peak risk (y^{-1})	Key radionuclides	Notes
Groundwater well	Early geomembrane failure and poor performance - deterministic	1 10^{-7}	U-238, U-234, Cl-36, Tc-99, I-129	Includes the annualised probability a well exists in the contaminated area
Groundwater stream*		4 10^{-7}	Cl-36	Assumes exposures occur
Gas (total risk from C-14 and radon)	Early geomembrane failure	1 10^{-7}	C-14	Includes the probability of exposure

* The peak risk for the probabilistic reference case is lower at 2 10^{-7} y^{-1}. This includes the probability of releases to surface streams.

The probabilistic reference case results include the probability of poor engineering performance resulting in discharges to the stream pathway. Poor engineering performance results in discharges to the stream pathway in 6% of realisations. The conditional risk calculated for the stream biosphere pathway in the early geomembrane failure and poor cap performance case (Table 12.2) can be multiplied by a probability of 0.06 (i.e. 6%) to give an indication of the radiological risk, i.e. 0.06 x 4 10^{-7} y^{-1} = 2 10^{-8} y^{-1}.

These variant cases provide confidence that, even with poor cap performance, the risks are consistent with the risk guidance level. Overall, our assessment results show peak risks for the Stage 2 Reference Inventory (Subsection 3.4) are consistent with the risk guidance level. Radiological capacities and WAC derived from our assessment will ensure risks from the disposed inventory are consistent with the risk guidance level.

Consistency with Requirement R7: Human Intrusion

The GRA [19] requires that human intrusion into a near-surface disposal facility after the end of management control of the site is to be assessed on the basis that it occurs, and against a dose guidance level in the range of around 3 mSv y^{-1} for exposures continuing over a period of years to around 20 mSv y^{-1} for exposures that are of limited duration.

We have identified a range of possible human intrusion events and more prolonged exposure situations, based on characteristics of the repository and credible future uses, assuming the site is no longer protected under planning procedures, and the presence and nature of the disposal facility is forgotten. We have assessed these events qualitatively and selected for quantitative assessment those events that we consider are representative and have potential to cause radiological exposures. This includes events that occur while the repository is still intact, and events that could occur when the facility is eroding and wastes are more directly accessible at the coast.

The events include:

- aircraft crash into the repository;
- geotechnical investigations and laboratory analysis of samples;
- exposure of waste by deep excavations, followed by occupancy of a contaminated site;
- exposure to radon in a house with a basement that breaches the cap low permeability layers and intercepts the gas collection layer;
- informal and organised scavenging of waste exposed by coastal erosion.

All transitory exposure events are below the relevant regulatory dose guidance level (20 mSv y^{-1}). The highest doses from existing and potential future disposals are to the clean-up crew following an aircraft crash into a shielded module in Vault 11 (5.8 mSv) and from analysing borehole core samples from localised deposits of monazite sand waste in the trenches (3.7 mSv). Although the probabilities of these events cannot be quantified, they are both very low.

All prolonged exposure events (the occupiers and the informal scavenger) have a maximum dose below the relevant regulatory dose guidance level (3 mSv y^{-1}). The highest calculated annual dose is 2.1 mSv from occupying a building constructed on soils contaminated by waste excavated from Vault 11. The thickness of the cap and profile fill over the most hazardous trench wastes, including the monazite sands, is too great for these wastes to be exposed by the intrusion events leading to a contaminated site.

The highest annual dose from radon in a house with a basement is 1.3 mSv above future disposals in Vault 9. The gas assessment [26] notes that most of the Ra-226 in the future Vault 9 inventory is in Ra-226 sealed sources. The available information indicates some, maybe all these sealed sources would not meet the current WAC. Therefore, they might not be accepted for disposal, and the peak calculated annual dose is too high.

If these Ra-226 sources are accepted for disposal, calculated doses could be reduced by up to an order of magnitude by placing these wastes in bottom of stack positions [26]. Cautiously, the calculations take no credit for decay of radon while it migrates up the gaps between vault waste stacks, or the propensity of radon gas to sink due to its high density.

The peak annual dose from scavenging is 1.8 mSv from informal scavenging, again from the monazite sands. However, it is unlikely this waste would be attractive for informal scavenging (or organised scavenging).

Consistency with Environment Agency guidance on protection of groundwater

The Environment Agency has produced guidance on protection of groundwater [20] [21]. Part of the guidance relates to the potential impacts on groundwater after the end of the PoA. The calculated groundwater pathway impacts are consistent with the relevant parts of the guidance, showing groundwater is being protected.

Overall assessment outcomes

Overall, our assessments demonstrate the LLWR is safe from the end of the PoA until the repository has been fully disrupted by coastal erosion. Thereafter, concentrations of radionuclides in surface coastal and marine sediments decrease as long-lived radionuclides are buried in marine sinks.

The results presented above assume LLW and ILW are disposed in the future. We have also assessed the potential impacts if only LLW is disposed in the future. The peak potential groundwater pathway impacts are from existing disposals to the trenches, while the peak impacts from radon are from existing disposals of Ra-226 to the trenches and vaults. These impacts will not change if ILW is not disposed. The potential impacts from future vault disposals are most significant for risks from C-14 gas, potential doses to people using the coast in front of the eroding repository for recreational use, and people consuming marine foodstuffs at a high rate while the repository is eroding. The peak area-averaged dose from C-14 gas over the vaults decreases by a factor of ten if only LLW is disposed, while the peak risk from the repository decreases by a factor of five. The peak doses from the vaults to recreational users of the coast and high-rate marine foodstuff consumers decrease by a factor of five. Peak doses to high-rate marine foodstuff consumers from the repository decrease by a factor of three.

12.3 Key Uncertainties

We have used a comprehensive, systematic approach to identify biases and uncertainties in our assessments. We have identified the most significant biases and reduced them where possible during development of the ESC. We have identified the priority uncertainties and remaining biases and assessed them qualitatively through logical arguments, and quantitatively in variant calculation cases. In some instances, qualitative arguments are used because the potential consequences are simple, in others the consequences are complex, but are not amenable for quantification through variant calculations.

The major uncertainties are listed in Table 12.3. Understanding the major uncertainties improves our understanding of the repository and the factors that need to be considered in ongoing optimisation and management decisions.

Table 12.3: Summary of key uncertainties for each pathway

Uncertainty	Management of uncertainty in assessments	Implications of uncertainty
All pathways		
Future inventory.	A best estimate projection of the LLW inventory that will be disposed to the LLWR is used as the basis for assessments. Assessments also consider impacts from potential disposal of ILW. See further discussion in Subsection 3.4.1.	The potential risks from the facility are generally directly proportional to the radionuclide inventory or concentrations of radionuclides in wastes. The calculated doses and risks from the assessments are therefore useful to develop WAC, radiological capacities and other controls to limit potential impacts from the LLWR.
Groundwater pathway		
Long-term cap performance.	The time of onset of cap degradation and infiltration rates through the cap are treated as uncertain in the probabilistic reference case. Variant cases also study poorer than expected or better cap performance.	Failure of the cap geomembrane results in higher infiltration into the repository (although still expected to be low) and higher risks. With high percentile infiltration rates, the increase in infiltration following failure of the cap geomembrane could be sufficient to result in overtopping to the perimeter of the cap, with subsequent releases to the stream and estuary. Even with early degradation of the cap geomembrane and a high rate of infiltration, risks remain below the risk guidance level.

Uncertainty	Management of uncertainty in assessments	Implications of uncertainty
Long-term performance of future stronger LLW containers and new containers for ILW.	Cautiously, we take no credit for the containers providing any physical containment function in the groundwater pathway reference case.	The assessment approach is likely to overestimate releases of contaminants to groundwater. The reference case assumes the containers do not to provide any barrier to flow of water through the waste, or release of contaminants by advection or diffusion.
Gas pathway - Radon		
Types and numbers of defects present in the cap geomembrane, and geomembrane failure time.	50th percentile values are used for the number of defects present in the cap geomembrane. Uncertainty in the number and capture radius of defects is assessed qualitatively. A variant case considers the earliest time for complete degradation of the geomembrane.	The types and numbers of defects in the cap geomembrane included in the radon assessment are described by reference [26]. More detail on the potential numbers of defects is provided by reference [95]. The effects of uncertainty in the numbers of defects (small holes) in the cap geomembrane, the spatial distribution of defects, and heterogeneities in the gas pathways, are bounded by the geomembrane degraded and open vent cases. They show that fewer defects lead to higher doses from the radon pathway through a defect. However, with few defects the probability of radon pathways to houses on the cap is lower, so the risks are little changed, and vice versa.

Uncertainty	Management of uncertainty in assessments	Implications of uncertainty
Whether cap vent(s) will be closed at the end of active institutional control, and the vent design.	A variant case assesses the impact of leaving the cap vent open after the end of the PoA. A reference mushroom cap design is assumed for the assessment. Future gas management strategy and vent design optimisation work is planned.	Ra-226 disposed directly below the vent is the main source of radon gas. This uncertainty has low consequence for risks because most of the radon decays before reaching the vent (sensitive to the amount of bulk gas) and the probability of housing on the cap vent is significantly lower than the probability of housing somewhere on the cap.
Bulk gas generation rates	Bulk gas generation rates are calculated using SMOGG. Uncertainty is not assessed quantitatively, but sensitivity to the amount of bulk gas is demonstrated by differences between the trenches and vaults, and by considering different gas generation rates in side calculations.	Higher bulk gas generation rates would lead to faster advection of radon, resulting in reduced decay before it reaches a defect (small hole) in the cap geomembrane. The flux of radon from the mushroom vent at the crest of the cap is also sensitive to the amount of bulk gas. Earlier calculations with unreasonably high bulk gas generation from the trenches (not reported) resulted in higher doses all cases, but especially for a house above an open vent. However, risks remain below the risk guidance level.

Uncertainty	Management of uncertainty in assessments	Implications of uncertainty
Bulk gas flow paths and travel times	Variant cases examine alternative flow paths in the profile and gas collection layer, considering radial flow in the profile fill, local gas spreading in profile fill, and wider pressure equilibration and gas spreading in the gas collection layer.	Gas spreading would result in advection of radon along a longer path, leading to increased decay before it reaches a defect. The reference assumption of no spreading is a little cautious. Radial flow in the profile fill would reduce the pathlength to a defect and therefore increase fluxes of radon. Radial flow has greater impact on doses with cohesive profile fill (i.e. over the trenches) than with granular profile fill (i.e. over the vaults) but is less likely to occur with cohesive profile fill than granular profile fill. Given that radial flow is possible in the profile fill over the vaults but is unlikely in profile fill over the trenches, this uncertainty is unlikely to lead to risks above the risk guidance level.
Gas pathway - C-14		
Cap geomembrane failure time and rate of transition in conditions as the cap geomembrane fails.	The reference case assumes the cap geomembrane does not fail before the repository is disrupted by coastal erosion, but a variant case considers earlier than expected degradation of the cap geomembrane.	Peak releases of C-14-bearing gas are higher after geomembrane failure, due to increased cap infiltration allowing C-14-bearing small organic molecules in solution to be transported out of the containers and then microbially metabolised forming gas.

Uncertainty	Management of uncertainty in assessments	Implications of uncertainty
Long-term performance of future stronger LLW containers and new containers for ILW.	Future stronger containers and new ILW containers are likely to be intact at the time of cap geomembrane failure, and the assessment considers that they provide an important barrier by limiting water contact with the waste and diffusion of C-14 in solution out of the containers.	The future container designs provide an important barrier to release of C-14 following geomembrane failure. The impact can be quantified by comparing with calculated releases from existing containers (see below).
Impact of surcharging on the barrier performance of containers of existing designs.	Cautiously, we take no credit for containers of existing designs in the gas pathway assessment.	The assessment approach is likely to overestimate peak releases of C-14 gas from existing container designs, since the containers are assumed to provide no resistance to transport.
Coastal erosion pathway		
Elevation of the erosion front when the repository is being disrupted (linked to uncertainty in future greenhouse gas emissions and the amount of sea-level rise).	Variant cases consider lower and higher sea-level rise projections than the reference value, corresponding to High and Low Emissions Scenarios.	The elevation of the erosion front affects the expected mode of erosion of parts of the repository (undercutting, direct erosion or inundation). The reference case gives higher peak doses than the relevant variant cases, except for the vaults in the High Emissions Scenario. Peak calculated risks are just above the risk guidance level in the High scenario.

Uncertainty	Management of uncertainty in assessments	Implications of uncertainty
Radionuclide - particle size associations.	Variant cases test the impact of upper and lower bounds for the transport of radionuclides through the coastal environment. The reference case treats radionuclides as being transported at the same rate as bulk materials on average.	Slower radionuclide transport could result in more accumulation of activity on the beach and foreshore, resulting in higher doses. However, key radionuclides are associated with a mix of waste types so it is unlikely that they would all be transported substantially faster or slower than bulk materials as assumed in the variant cases.
Sorption distribution coefficients for the coastal and marine environment.	Variant cases test the impact of maximum and minimum sorption coefficients for key radionuclides.	Maximum sorption distribution coefficients result in lower doses than the reference case. Minimum sorption distribution coefficients result in doses around a factor of two higher than the reference case, and a little above the dose corresponding to the risk guidance level assuming exposure occurs, at 36 μSv. However, the variant cases consider extreme values for all radionuclides.

The key uncertainty for the human intrusion pathway is the types of events that should be included in the assessment. We have used a systematic approach to ensure that we have included a representative and robust set of possible events. The uncertainties associated with each event are substantial, which is why we use simple, stylised conceptual and assessment models.

12.4 Treatment of Heterogeneity

Radionuclides are heterogeneously distributed in the waste. Heterogeneity exists at all length scales, ranging from the distribution of activity on, or within, a waste item, to differences in the radioactivity associated with individual waste items, and then to differences between containers, waste stacks, sub-vault areas or modules and sub-trench areas, vaults and trenches. This heterogeneous distribution of activity is included in our assessments. We do this through two approaches:

1) Explicitly representing heterogeneity in our assessment models.
2) Developing controls on heterogeneity to ensure it could not lead to potential impacts which exceed the risk guidance level or dose guidance level.

Some heterogeneities cannot be known; for example, we cannot know the activities of individual future waste items, which items would be in which container, and where each container would be in the vaults. Even where such information is available for existing disposals, it is not tractable to model individual waste items. This results in biases in our assessment models.

Where biases could affect the potential doses or risks, we have undertaken additional assessments to identify controls to limit the bias and ensure the potential impacts remain consistent with the GRA. These controls include activity limits that can be included in the WAC (active particles, low-activity sealed sources, and discrete items) and possible additional controls on heterogeneity of activity within individual containers (potential doses to borehole analysts from 'hotspots'), and the spatial distributions of key radionuclides in the repository (coastal erosion pathway).

12.5 Implications for WAC

The assessment calculations have been undertaken with a Reference Inventory that includes the existing disposed inventory and our best estimate of future disposals, based on the UKRWI. The future LLW inventory includes waste that could be disposed at the LLWR following diversion (including new build waste), and secondary waste from waste treatment. We have used radiological capacities and consignment activity limits derived from earlier environmental and nuclear safety assessments, and waste producer plans and priorities, to identify the ILW streams that might be disposed at the LLWR (in full or in part).

There is uncertainty in the Reference Inventory for future disposals. There is uncertainty in the UKRWI, for example, due to the inherent uncertainty in the inventory that will arise from future decommissioning, and uncertainty over the effects of diversion including the potential to dispose of some wastes at other disposal facilities. If ILW were to be disposed to the LLWR in the future, there would also be decisions to be taken about what ILW would be accepted.

The doses and risks calculated using the Stage 2 Reference Inventory are consistent with the GRA dose and risk guidance levels, including the relevant parts of guidance on protecting groundwater [20] [21]. This demonstrates that an overall assurance of safety can be provided by developing WAC and controls on the radionuclide activities and concentrations that can be disposed to the repository. The reference and uncertainty cases explore how the system performs, inform and build confidence in the WAC and these controls.

Updated radiological capacities and concentration limits have been calculated by each pathway assessment. As noted in the previous sub-section, we have addressed heterogeneity at all length scales and derived additional activity limits and other controls that can be used to ensure that heterogeneities that cannot be represented in our assessment models could not lead to doses or risks above the levels described in the GRA, GRR and guidance on protection of groundwater.

Updated radiological capacities, activity limits and other potential controls are integrated, together with wider considerations (e.g. relevant to transport and repository operations), by the '*Implementation'* report [17]. The '*Implementation'* report forms the basis for updating the WAC, so the potential impacts of the repository remain consistent with the levels described in the GRA, GRR and guidance on protection of groundwater.

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Nuclear Waste Services Limited
Pelham House
Pelham Drive
Calderbridge
Cumbria CA20 1DB
UK

t +44 (0)300 369 0000
w www.nuclearwasteservices.uk

Nuclear Waste
Services

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is registered in England and Wales with the registered office located at
Pelham House, Pelham Drive, Calderbridge, Seascale, CA20 1DB.

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