Nuclear Waste Services

ASSESSMENT OF IMPACTS ON NON-HUMAN BIOTA

2026 Environmental Safety Case for the LLWR

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Conditions of Publication

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.

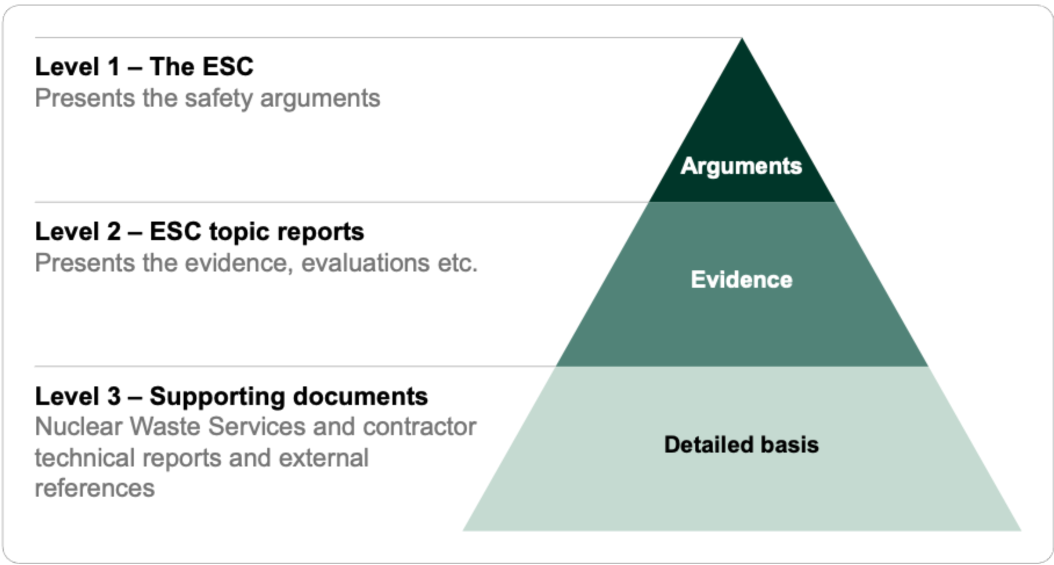
The 2026 ESC is issued under the authority of the Nuclear Waste Services' Executive Director of Sites and Operations.

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.

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



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

Near Field [6]	Describes our understanding of the chemical and physical evolution of the engineered disposal system
Hydrogeology [7]	Describes our understanding of the geology and hydrogeology of the site
Site Evolution [8]	Describes our understanding of how the site will evolve, with a focus on coastal erosion
Monitoring [9]	Presents our programme of environmental monitoring supporting the ESC
Optimisation and Site Development Plan	
Optimisation and Site Development Plan [10]	Describes our approach to optimising the design and management of the disposal facility and wider site, and sets out our Site Development Plan
Waste Management Plan [11]	Presents our plans for managing the wastes produced by previous uses and operation of the site
Assessments	
Safety Functions [12]	Presents our understanding of how the different aspects of the repository system and its management contribute to the safety of the facility
Engineering Performance Assessment [13]	Presents our analysis of how the various components of the engineered disposal system will perform, which is an input into our impact assessments
Environmental Safety During the Period of Authorisation [14]	Presents evidence that the LLWR is currently being operated safely and will continue to be so during the period that the facility is permitted
Assessment of Long-term Radiological Impacts [15]	Presents evidence that, if the LLWR is managed in accordance with the Site Development Plan, the site will remain safe in the long term
Hydrogeological Risk Assessment [16]	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 (this report)	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

As well as ensuring that humans are protected from exposure to radiation and radioactive materials originating from the Low Level Waste Repository (LLWR), we need to ensure that the environment is adequately protected. This is expressed as a regulatory requirement "*to protect ecosystems against radiation exposure that would have adverse consequences for a population* [of non-human biota]". This report assesses the radiological impacts of the disposal of radioactive waste at the LLWR on the environment and shows that the regulatory standards are met.

The LLWR site is bordered by farmland to the north and east. However, to the west and south the site is contiguous with or lies close to the Drigg Coast Site of Special Scientific Interest, which is also designated as a Special Area of Conservation. The repository is also close to the Cumbria Coast Marine Conservation Zone. There are two inland designated Sites of Special Scientific Interest within 5 km of the LLWR: Hallsenna Moor is approximately 2 km to the north-east of the LLWR, and Drigg Holme is approximately 2.5 km from the LLWR. The Morecambe Bay and Duddon Estuary Special Protection Area is south of the LLWR site.

Radioactivity from the LLWR site can enter the environment from releases of radionuclides to surface water or groundwater, releases of radionuclides to air, from coastal erosion or from disturbance of waste during human intrusion. Different pathways are relevant at different times and in different locations. Consequently, we assess the following pathways:

- gaseous releases:
    - tritium (during the Period of Authorisation only);
    - C-14;
    - Rn-222;
- migration of leachate in groundwater with releases to marine foreshore and local coastal waters and, after the Period of Authorisation, abstraction from a well and use for irrigation;
- liquid discharges:
    - managed leachate discharge via marine pipeline (during the Period of Authorisation only);
    - one-off managed leachate discharge via the stream (during the Period of Authorisation only);
- historical contamination in the Drigg Stream;
- direct and scattered radiation from uncovered vaults;
- coastal erosion of the repository.

We also explore the potential implications of the release of radioactive particles.

Our assessments of radiological impacts to people include contaminant transport models. These models calculate activity concentrations of radionuclides resulting from each pathway in different environmental media (soil, sediment, water and air). These activity concentrations are then used to assess doses to people. Our assessment of radiological impacts to people during the Period of Authorisation also uses measured activity concentrations.

To assess dose rates to non-human biota, we take measured and modelled concentrations of radionuclides in environmental media from the assessments of radiological impacts to people. This approach ensures we use the same understanding of release and physical transport processes for assessments of radiological impacts to people and non-human biota.

We use the ERICA tool, an internationally recognised approach, in this assessment. This uses simple, stylised models, both for the ecosystems being assessed and for the organisms in those ecosystems. The models used represent a broad range of potential ecosystems and organisms and, thus, account for potential future variations in the ecosystems and organisms present around the LLWR.

We identify representative organisms around the LLWR based on species that are found at and near the site, including on nearby protected areas. We assess the radiological impacts to these representative organisms that would arise from disposal of the LLWR Reference Inventory, which is predominantly Low Level Waste but also includes a component of Intermediate Level Waste. Transfer of radionuclides from the environment to the non-human biota is modelled using concentration ratios, which estimate the internal whole-body activity concentration in organisms assuming equilibrium has been established between the media and organism. External and internal dose rates to the representative organisms are then calculated.

We compare the assessed dose rates to a screening dose rate of 10 µGy h^{-1}. Dose rates below this are likely to have negligible impacts on populations of non-human biota. We use the maximum concentration of each radionuclide, which will occur at different times, to calculate the assessed dose rate for this screening; this is a cautious approach.

Where the screening dose rate is exceeded, we consider the

- time-dependency of radionuclide concentrations and, thus, calculated dose rates;
- radiosensitivity of the assessed organisms, e.g. with reference to International Commission on Radiological Protection derived consideration reference levels;
- size of the contaminated or affected area relative to the area required to support a population of the affected organisms;
- notable uncertainties and pessimisms associated with the modelling approach.

Most assessed dose rates are below the screening dose rate. Where assessed dose rates exceed the screening dose rate, we have considered the potential for these dose rates to have detrimental consequence for populations of non-human organisms. We conclude that even if there is potential for radiological detriment to some individual organisms, it is highly unlikely that there would be harm to the local populations of these species either during the Period of Authorisation or afterwards.

During the Period of Authorisation, all assessed dose rates are lower than our screening dose rate except for direct and scattered radiation from one vault (Vault 11) when it is uncapped. For all other exposure pathways during the Period of Authorisation, the highest dose rate to non-human biota is for insect larvae and arises from historical contamination of the Drigg Stream. It is about half the screening dose rate. Assessed dose rates for the other pathways are all at least one order of magnitude below the screening dose rate.

Although the direct dose rate from Vault 11 (to all organisms) exceeds the screening dose rate, the affected area would be very small and dose rates in most of the affected area would be lower than calculated in our assessment. Therefore, we conclude that these dose rates will not result in detriment to populations of non-human organisms.

After the Period of Authorisation, all assessed dose rates are below the screening dose rate except for dose rates to organisms on the cliff and storm beach during erosion of the repository.

For many of the organisms on the cliff and storm beach during erosion of the repository, considering the time-dependency of the dose rates and the radiosensitivity of the organisms shows that detrimental effects to populations of organisms are unlikely. Moreover, the areas of the cliff and storm beach affected are limited to the areas in front of the eroding repository. These will be relatively small areas, constantly disrupted by weather, wave and tidal action. They will not be large enough to sustain populations of most organisms, and the organisms present will typically be transient, spending only a small proportion of time in the areas. Therefore, we conclude that these dose rates will not result in detriment to populations of non-human organisms.

We also consider the potential for harm to non-human biota from contact with high-activity particles during coastal erosion or human intrusion. We find that exposure to particles would not have significant impact on the overall health of individual organisms (and, therefore, would not have impacts on the health of populations).

The assessment reported here is based on an assessment of the potential impacts of the reference inventory. However, we have derived waste acceptance controls based on impacts to people; we conclude that these controls will also be sufficient to ensure that non-human biota are protected. Therefore, we conclude that the LLWR will meet the requirement "*to protect ecosystems against radiation exposure that would have adverse consequences for a population* [of non-human biota]", even if the inventory accepted differs from the reference inventory.

Table of Contents

1 Introduction

1.1 Objectives

As well as ensuring that humans are adequately protected from exposure to radiation and radioactive materials originating from the LLWR, we need to ensure that the environment is adequately protected. This is stated in Requirement R9 of the environment agencies' '*Guidance on Requirements for Authorisation'* (GRA) [19] and in Requirement R14 of their '*Guidance on Requirements for Release from Radioactive Substances Regulation'* (GRR) [20].

GRA Requirement R9: Environmental radioactivity

The developer/operator should carry out an assessment to investigate the radiological effects of a disposal facility on the accessible environment both during the period of authorisation and afterwards with a view to showing that all aspects of the accessible environment are adequately protected.

GRR Requirement R14: Protection of the environment

Operators shall assess the radiological effects of the site on the environment with a view to showing that all aspects of the environment are adequately protected, both during the period of, and after release from, radioactive substances regulation.

These guidance documents provide further detail about these requirements. The GRA states that discharges and migration of radionuclides from a disposal facility might have a detrimental effect on the environment, through effects on non-human species or more general environmental effects such as damaging habitat quality. The requirements aim to ensure that all aspects of the environment are protected.

The guidance explains that assessment of radiological impacts on the environment is required because, although there is no specific evidence that there might be a threat to populations of non-human species from the authorised release of radioactive substances if people are protected, there may be times when there are no people near a disposal facility. Environmental damage might also occur to areas and habitats that are not extensively exploited by people.

Some non-human species and sites are protected under legislation related to conservation, for example that derived from the EC Habitats Directive [21], and there are a range of statutory provisions relating to habitat, biodiversity and conservation matters. The environment agencies will only authorise disposals and release sites from regulation when they are satisfied that these provisions are met.

As is recognised in the guidance, however, there are no statutory criteria for determining radiological protection of the environment. In the absence of specific criteria, the environment agencies expect the developer or operator to carry out an assessment and to draw conclusions about the effects of a disposal facility, and the site as a whole, on the environment using the best available information at the time of the assessment. Nearby protected sites should be considered.

The GRR explains that, for their own assessments, the environment agencies use the *Environmental Risk from Ionising Contaminants: Assessment and Management* (ERICA) approach [22], implemented through the ERICA Tool [23]. They use a dose rate screening value of 10 μGy h^{-1} for assessments of designated conservation sites. This is a screening value, not a limit; where it is exceeded, they do more detailed, site-specific assessments.

The developer or operator is required to provide their assessment as an integral part of their Environmental Safety Case (ESC) for the facility and Site-wide Environmental Safety Case (SWESC) for the site. The environment agencies expect the extent and complexity of the assessment to be proportionate to the radiological hazard presented by the waste in the facility. The SWESC must consider all sources of radiological hazard on a site. At the LLWR, the waste in the disposal facility is the main radiological hazard on the site, with other less significant areas of other waste and contamination resulting from the use of the site for the storage and disposal of radioactive waste [11].

The objective of this assessment is to show that the radiological effects of the LLWR on the environment for all pathways of exposure, both during the Period of Authorisation (PoA) and afterwards, are sufficiently small that all aspects of the environment are adequately protected. We explicitly consider protected sites and species as part of this assessment.

1.2 Scope

We assess radiological impacts on the environment from radionuclides released from the repository. There are other smaller sources of contamination from other site activities, but the only such source we have included in this assessment is historical contamination in the Drigg Stream (which derives from past discharges from the repository). The inventory associated with these sources of contamination is much smaller than that in the repository, and we will assess them in a future SWESC.

In this report, we assess the radiological impacts to non-human biota (NHB) of the various pathways by which radionuclides could be released from the LLWR. To do this, we use the monitoring data and mathematical models used to assess radiological impacts to people (see references [9], [14] and [15]) and the ERICA Tool [23]. The assessments of radiological impacts to people calculate radionuclide concentrations in environmental media as an intermediate output. These concentrations are used as input to the ERICA Tool to evaluate the radiological impacts on NHB. The assessment also considers direct dose rates to biota from uncovered vaults and the radiological impacts on biota of radioactive particles, which are not based on environmental radionuclide concentrations.

We assess impacts both during the PoA and after the PoA until the disruption of the site by coastal erosion.

1.3 Structure

This report is structured as follows:

- Section 2 describes the geographical context of the site and its environs and our understanding of the types of ecosystems that exist at the present day and that may develop in the future.
- In Section 3 we discuss the background to and context of assessing radiological impacts to non-human biota. We outline our use of Representative Organisms and the dose rate screening criteria that can be used to demonstrate that they are protected. We explain how we evaluate the significance of exceedances of these criteria.
- Section 4 sets out our approach to this assessment. It describes the pathways we assess and describes and justifies our use of the ERICA Tool. It also discusses how we account for climate change and how we manage the uncertainties in the assessment.
- Section 5 summarises whole-body dose rates to Representative Organisms during the PoA, and comments on the significance of those dose rates for each Representative Organism.
- Section 6 provides a similar description and evaluation of assessed dose-rates after the PoA.
- Section 7 summarises our findings and draws conclusions from the assessment.

The assessment is described in more detail in an underlying report [24].

2 Geographic and Ecological Context

This section describes the geographical and ecological context of the LLWR site. Pathways by which radionuclides are released and interact with this environmental setting are described in Subsection 4.3.

2.1 Geographic Context

The LLWR is located on the West Cumbrian coastal plain, close to the village of Drigg and approximately five kilometres south-east of Sellafield. Apart from 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 site and its surroundings are shown in Figure 2.1.

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 (SAC). The coastline near the site is designated as a Marine Conservation Zone (MCZ), known as the Cumbria Coast MCZ [24].

Along the north-eastern boundary is the Carlisle to Barrow-in-Furness railway line, a siding from which enters 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 2.1: The LLWR Site and its immediate surroundings

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 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 topography surrounding the site varies from 25 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.

Near the site, the Regional Groundwater generally flows sub-horizontally from the Cumbrian mountains towards the coast, discharging to the sea. At the site, the Upper Groundwater flows generally downwards from the surface to the Regional Groundwater. There are localised discharges from the Upper Groundwater to nearby streams; however, these are from regions of the Upper Groundwater that do not have elevated contaminant concentrations [7]. A component of the Regional Groundwater passing under the south-eastern part of the site discharges to the River Irt and the Ravenglass Estuary [7].

The Drigg Stream 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 northeast, draining farmland. 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 River Mite and the River Esk, which discharges to the sea near the village of Ravenglass.

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

2.2 Current Ecological Context

The LLWR site is bordered by farmland to the north and east. However, to the west and south the site is contiguous with or lies close to the Drigg Coast SSSI, which is designated as both an SSSI and a SAC. It is also close to the Cumbria Coast MCZ. The nearby conservation areas are shown in Figure 2.2. Most of the releases from the LLWR are expected to the west and south-west of the site. The species present on and around the LLWR site (including on nearby designated conservation sites) are summarised in Subsection 3.2.

2.2.1 Inland Designated Conservation Sites

There are two inland designated SSSIs within 5 km of the LLWR: Hallsenna Moor and Drigg Holme [24]. Being inland from the LLWR, there are no direct groundwater contaminant transport pathways to these sites and, while airborne emissions from the LLWR may reach the sites, these will have a lesser impact than locations closer to the LLWR which are considered in more detail in our assessment.

- Hallsenna Moor SSSI is situated approximately 2 km to the north-east of the LLWR and is just over 30 hectares in area. The habitat includes wet and dry heath, nutrient-poor fen, basin mire and woodland. The mix of open moorland and woodland provides habitat for a wide range of bird species.
- Drigg Holme SSSI is approximately 2.5 km from the LLWR and consists of 9 hectares of lowland neutral and acidic grasslands on the flood plain of the River Irt. The grasslands have 150 different flowering plant species.

2.2.2 Coastal and Inshore Designated Conservation Sites

There are three nearby coastal and inshore designated conservation sites [24]. The Drigg Coast SSSI and SAC (Subsection 2.2.2.1), the Cumbria Coast MCZ (Subsection 2.2.2.2) and the Morecambe Bay and Duddon Estuary Special Protection Area (SPA) (Subsection 2.2.2.3).

2.2.2.1 Drigg Coast SSSI and SAC

The Drigg Coast SSSI and SAC, which is partly adjacent to the LLWR site, covers 1400 hectares and extends 11 km along the West Cumbrian coast, from Seascale to the River Esk [24].

The SSSI citation includes species such as great crested newts, natterjack toads and dragonflies [24]. The sand dunes are a habitat for reptiles, breeding birds and invertebrates. Three important habitats were the primary reason for designation of the SAC:

- estuaries;
- Atlantic decalcified fixed dunes (*Calluno-Ulicetea*);
- dunes with *Salix repens* ssp. argentea (*Salicion arenariae*).

Atlantic decalcified fixed dunes (*Calluno-Ulicetea*) is a habitat type that occurs on mature, stable dunes where the initial calcium carbonate content of the dune sand is low. The surface soil layers rapidly lose their remaining calcium carbonate through leaching and become acidified. There are substantial areas of these dunes at Drigg, showing a wide range of ecological variation. Some areas are dominated by heather *Calluna vulgaris* and bell heather *Erica cinerea*. Within the dry dune heath are wetter areas in which cross-leaved heath *Erica tetralix* is prominent. There are large areas of acidic dune grassland with a prominent lichen component and areas where sand sedge *Carex arenaria* grows in carpets of the moss *Racomitrium canescens*.

Dunes with *Salix repens* ssp. *argentea* (*Salicion arenariae*) is a habitat type that comprises dunes or parts of dunes where creeping willow *Salix repens* ssp. a*rgentea* is dominant, forming prominent, low scrubby growth.

Creeping willow is found on dunes throughout the UK. It grows predominantly in and around dune slacks, though on some sites it may spread up the drier ridges. This type of vegetation marks the mature phase in the life cycle of calcareous dune slacks. When found with other wetland and dry dune vegetation it indicates that successional processes are still active and that the structure and function of the dune system are well-conserved. This type of habitat may migrate inland during long-term erosion processes.

The Drigg Coast SAC has several dunes with *Salix repens* ssp. *argentea*. These span a range of hydrological conditions from very wet to relatively dry. The slacks also grade into more acidic humid dune slacks. The site is representative of dunes with *Salix repens* ssp. *argentea* in north-west England.

The Drigg Coast SAC has been designated as a SAC with marine components because of the presence of the estuary and because of the presence of subtidal sediments including mudbanks and sandbanks.

2.2.2.2 Cumbria Coast MCZ

The Cumbria Coast MCZ protects several habitats, including honeycomb worm reefs and rocky shores, and one species (razorbill, *Alca torda*) [24]. The MCZ protects 27 km of coast, from the cliffs at St Bees Head to the mouth of the Ravenglass estuary. It supports rich communities of fish and burrowing animals and provides a feeding ground to 10,000 pairs of breeding seabirds that nest at St Bees Head.

An important feature of the site is the scars (intertidal reefs of boulders or cobbles). Barn Scar, near the LLWR site, is an important example of a scar that extends below the low-water mark. The boulders provide shelter for organisms including algae, long-clawed porcelain crabs, sea slugs, brittle stars and sponges. These provide food for crabs, fish and young lobsters. These intertidal habitats are protected by the MCZ designation. Examples of all the protected features can be found on the coast adjacent to the site [24].

The habitats in the MCZ are also important for seabirds. There is particular concern for razorbills following unexpected declines in their populations. Therefore, they are also protected by the MCZ.

2.2.2.3 Morecambe Bay and Duddon Estuary SPA

The Morecambe Bay and Duddon Estuary SPA is just south of the site. It protects the largest continuous area of intertidal mudflats and sandflats in the UK. The SPA has important waterbird and seabird species, five of which reproduce in the area and six of which overwinter in the area.

The site qualifies as an SPA through use by 1% or more of the members of the Great Britain populations of nine seabird species and through use by 1% or more of the members of the biogeographical populations of 16 seabird species. It also qualifies through use by over 20,000 individual seabirds and over 20,000 individual waterbirds. These seabird populations are supported, in part, by the features protected by the Cumbria Coast MCZ.

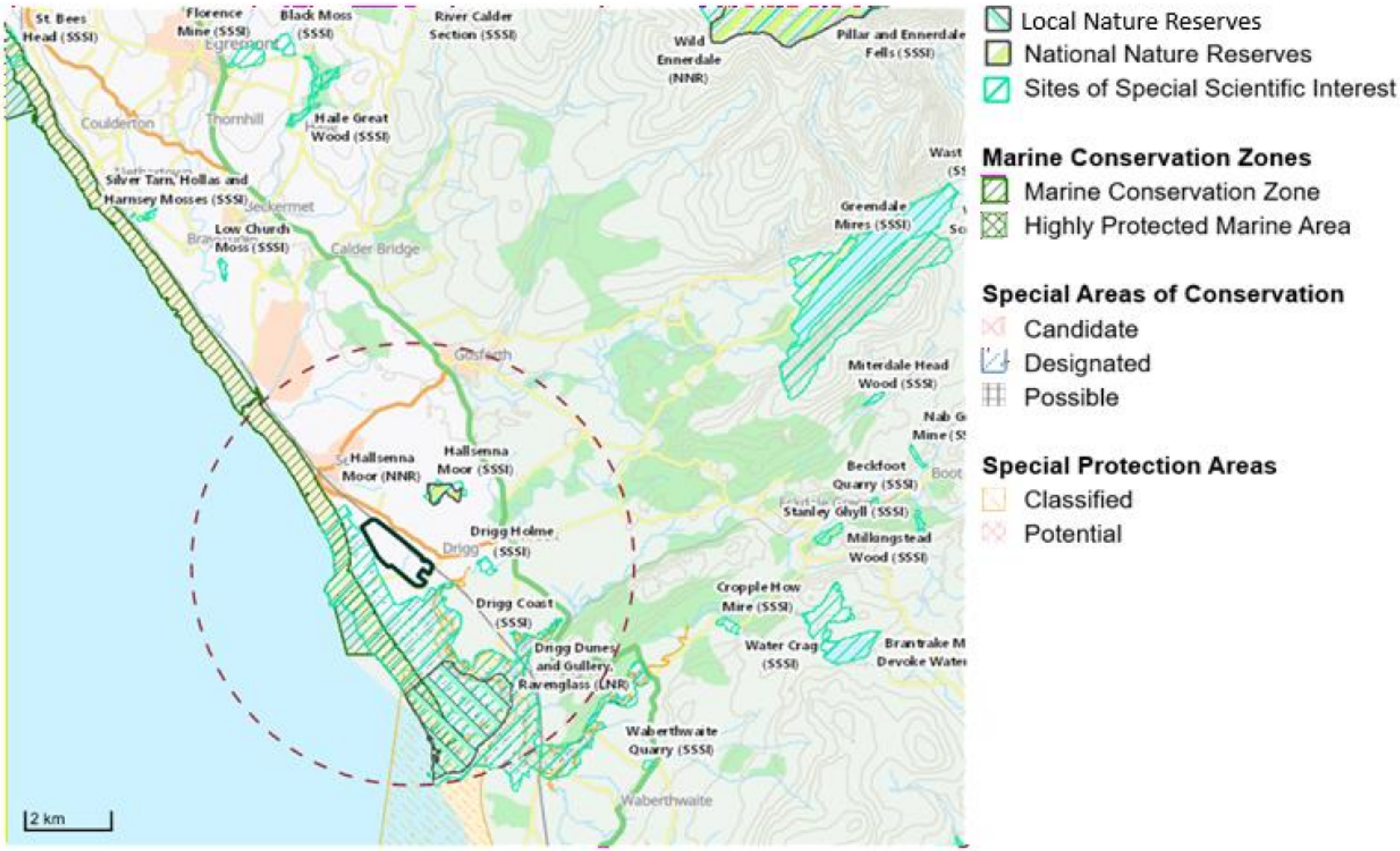


Figure 2.2: LLWR site with conservation areas marked. The red dashed circle indicates a 5 km radius circle centred on the LLWR site (MAGIC used with permission from Natural England).

2.2.3 Habitats Within Site

The LLWR site comprises a variety of habitat types [24], including:

- open dry grassland, comprising areas of unimproved acid and neutral grassland, where the acid grasslands are associated with heath vegetation or are remnant dune communities;
- a small fragment of relict dune heath;
- areas of marshy grassland and swamp;
- several semi-natural and man-made seasonal and permanent ponds;
- the Drigg Stream, which runs the length of the south-east side of the site, has steep banks (about 50° to 60°) that are up to 2 m high, is 2 to 3 m wide and 0.2 to 0.3 m deep, and for much of its length is choked by weeds;
- the East-West Stream, which supports tall, diverse vegetation;
- several small lengths of ditch (most of which are shady and support little vegetation);
- extensive parcels of willow and gorse scrub;
- plantation woodlands composed predominantly of Corsican pine, white poplar, sycamore and alder, but containing numerous other species.

2.3 Future Ecological Contexts

The timeframe over which the NHB assessment will apply is generally the same as that adopted for the assessment of radiological impacts on humans [14], [15]. The time considered in this assessment extends from the first disposals of waste at the LLWR site to erosion of the repository. This is from 1959 AD to 5742 AD.

2.3.1 Terrestrial and Freshwater Environments

The LLWR site will be capped and vegetated at closure. Run-off from the cap will be collected by surface drains which will be directed to a settlement lagoon. The lagoon will discharge to a successor to the existing Drigg Stream which flows into the Irt Estuary. Inland and to the north and south, there is no reason to suppose that land use would be very different from that at the present day, even allowing for a moderate amount of greenhouse-gas induced warming.

2.3.2 Estuarine, Coastal and Marine Environments

Although changes to the terrestrial and freshwater environments are expected to be limited, this is not the case for the estuarine and coastal environments.

Rising sea levels associated with global warming are projected to result in regression of the coastline with erosion of the LLWR on a timescale of several hundred to a few thousand years. Over this period, the area of terrestrial vegetation near to the sea and to the west of the site will be lost to cliff retreat.

The cliff line will eventually retreat sufficiently far that the wastes will be exposed. As the cliff continues to retreat, the wastes will be eroded and distributed throughout the storm beach, foreshore and marine environments. The characteristics of these environments will influence the ecological niches present, the stability of those niches and the types of organisms that might occupy them (Subsection 2.3.3).

The Ravenglass Estuary to the south will exhibit altered water and sediment dynamics as the sea level rises and it is displaced inland. Offshore, changes to the marine environment are projected to be limited, though water depths will increase by a few metres or more over the next few thousand years.

Areas of the terrestrial environment further inshore are not expected to be affected by coastal erosion on the timescales of interest to this assessment.

2.3.3 Changes in Species of Biota that May be Present in Future

Climate and other environmental change may lead to changes in the species present in habitats in and surrounding the LLWR site over the period covered by this assessment. It could also lead to changes in their habits (for example, the amount of time they spend on or near the site or the types of food that they eat). It is difficult to predict what these changes might be. However, even if the specific species present change, we expect that the area will be populated by species of similar types to those present.

We do not assess dose rates to individual species; rather, we assess dose rates to representative organisms (Subsection 3.2). These are organisms defined to represent a range of similar species (considering, for example, size, physiology, food sources and habits). It is unlikely that species present in future would have characteristics outside those covered by our selected representative organisms. Therefore, we do not expect climate change to alter the species present or their habits such that assessed dose rates would be significantly different from those reported here.

This approach is consistent with the recommendation from the Environment Agency (see Recommendation ASS33 [18]) that we assess impacts based on those species that are observed today and assume that similar populations will be present in the future.

3 Radiological Protection of Biota

3.1 Background

3.1.1 Regulatory and International Guidance

Environmental protection is referred to in the International Atomic Energy Agency's (IAEA's) Fundamental Safety Principles [25], the recommendations of the International Commission on Radiological Protection (ICRP) [26] and the IAEA Basic Safety Standards [27]. From these standards, protection of the environment from the harmful effects of ionising radiation has been incorporated in UK regulations and regulatory guidance.

For the ESC, the main governing regulations are the Environmental Permitting Regulations [28], and the corresponding regulatory guidance is the GRA [19] and the GRR [20]. The overall protection objective is to protect the environment from deleterious effects of ionising radiation. Here, protection of the environment is taken to be synonymous with protecting biota. People are not included in this definition because they are protected through other controls on exposure to ionising radiation and on releases of radioactive materials.

The GRA (¶ 4.3.4) [19] states that "*for non-human species the general intent is to protect ecosystems against radiation exposure that would have adverse consequences for a population as a whole, as distinct from protecting individual members of the population*". In practice, protection is often demonstrated by assessing dose rates to exposed individuals. This is based on the argument that if individuals are adequately protected, the higher-order systems in which they have a role will also be adequately protected.

Where exposed individuals receive dose rates that could harm them, the proportion of the population exposed to such dose rates is one consideration that may be used to determine whether the population could be harmed and, thus, whether the protection objective is met. However, defining the population to which the protection objective should be applied can be difficult. Often a population is defined by the geographic area that the species requires to continue to breed successfully. Such a population can be defined as a group of individuals of the same species with factors such as the geographic location and area, density of the individuals, sex ratio, birth and death rates, emigration and immigration determining its extent.

UK legislation does not yet give any statutory criteria against which protection of the environment is to be assessed nor for acceptable dose rates or radiological impacts to NHB. The environment agencies use a screening dose rate of 10 µGy h^{-1} for initial assessments of radiological impacts from a single premises on protected species and sites [20]. Where this is exceeded, they require a more detailed, site-specific assessment to better understand the radiological impacts.

3.1.2 Scientific Background

Protection of the environment from the harmful effects of ionising radiation is an area of ongoing scientific research. Approaches and tools that can be used to estimate the exposure of wildlife to ionising radiation and to assess the impacts of these exposures have been developed. In brief, these approaches involve:

- estimating transfer of radioactivity to non-human biota;
- calculating dose rates to non-human biota;
- assessing the impacts of these dose rates on individual non-human organisms, populations of organisms or habitats.

The transfer of radioactivity in the environment may be estimated by assuming equilibrium transfer based on empirical data (sorption coefficients and concentration ratios) (as in the ERICA Tool [22]). Generic transfer data are available in the literature, for example in the Wildlife Transfer Parameter Database [29].

Dose rates to biota are estimated as absorbed dose rate at a point in time or averaged over a period. Unlike in assessments of radiological impact to people, there is no concept of committed or effective doses for biota. Dose rates to biota are calculated from the internal (biota tissue) and external (environmental media) activity concentrations using dose coefficients. ICRP Publication 136 [30] sets out three approaches to calculating biota dose coefficients; these have different approaches to including or excluding the ingrowth of daughter radionuclides during a specified integration period.

Assessing radiological impacts to wildlife requires that the assessed absorbed dose rate be related to the potential impact on the receptor. This may be done by considering observed effects data for individuals or populations [31]. The relationship between absorbed dose and radiological impact will vary between species and will depend on the impacts being considered. To account for differences between species, ICRP Publication 124 [32] sets out Derived Consideration Reference Levels (DCRLs). These are bands of dose rate within which there is some chance of deleterious effects from ionising radiation occurring to individual organisms from a given category of species.

It is not possible to develop ecological, bioaccumulation or dosimetric models for all types of organisms at all relevant stages of their life cycles. Therefore, usually a limited number of representative organisms are assessed using a simple equilibrium transfer approach. Whole-body dose rates are calculated for these representative organisms and compared with a screening dose rate. Provided that the calculated dose rates to the representative organisms are lower than the screening dose rate, it is accepted that most types of organisms would be adequately protected against deleterious effects, and that the higher-order systems in which they participate would also be adequately protected [20].

In this assessment, we use representative organisms (Subsection 3.2) and a screening dose rate (Subsection 3.3). Where the screening dose rate is exceeded, we consider whether the assessed dose rate is likely to have a negative impact on the species in question by considering the ICRP DCRLs [32] and information about species radiosensitivity available in the academic literature. We also consider whether individuals are likely to receive such dose rates (given their typical ranges and sources of food) and what proportion of a population may receive such dose rates (given the size, range and food source for the population). This is described in Subsection 3.4.

3.2 Representative Organisms

This assessment is based around a set of representative organisms that are characteristic of the types of plant and animal species associated with the LLWR site and nearby protected habitats. These representative organisms do not represent individual species; they are generalised representations of the types of species found locally. However, we map them to observed species to demonstrate that all organism types found near the LLWR are covered by the assessment.

The representative organisms are characterised by:

- their dimensions;
- the concentrations of radionuclides that they exhibit relative to the environmental media with which they are associated, known as their concentration ratios;
- the fraction of time that they are present in different environmental media (water, air or soil) in an ecosystem, known as their occupancy;
- the fraction of time that they are present in an ecosystem, known as their residency.

3.2.1 Occupancy

Table 3.1 gives the locations organisms may occupy in different types of ecosystems.

Table 3.1: Occupancy locations for biota

Aquatic ecosystems	Terrestrial ecosystems
On the water surface	On the soil
In the water body	In the soil
On the sediment surface	In the air
In the sediment	

The occupancies of each representative organism must sum to one (that is, its entire time in the ecosystem must be accounted for and it may only be in one location at a time). The occupancies are used for external dose calculations.

Internal dose calculations use concentration ratios to determine the internal activity concentration from activity concentrations in soil for terrestrial ecosystems[2] and water for aquatic ecosystems (the occupancies within the ecosystem do not influence the internal dose rate).

3.2.2 Residency

Individual organisms or species covered by the reference organisms may reside in different ecosystems. For example, the same species of plant may have members growing both on the cap and on land irrigated with contaminated water. To account for this, we assess most organisms separately in each ecosystem, assuming they spend 100% of their time in that ecosystem. This ensures the assessment considers the most exposed individual or species covered by each representative organism.

Some organisms may spend time in more than one ecosystem. For example, an otter will spend time in a stream or river and on the adjacent land, and a seabird will spend time both on cliffs and at sea. Organisms with large ranges may spend time in the same type of ecosystem in different places; for example, a deer will spend all its time in terrestrial ecosystems, but some of its time may be spent on or near the LLWR and some if its time may be spent farther away. Similarly, migratory organisms spend only some of their time near the LLWR.

We account for these behaviours using residencies. We assess these organisms for a fraction of their time in each relevant ecosystem and add the dose rates from each ecosystem together to give the total dose rate. We scale average dose rates to account for reduced residency. That is, an individual exposed to 10 μGy h^{-1} for half a year has an average dose rate of 5 μGy h^{-1}. We consider this appropriate because the dose rate screening value and DCRLs are based on chronic exposures, rather than acute exposures.

The sum of residencies for any representative organism cannot exceed one (organisms can only be in one place at any time) but may be lower than one (organisms may spend time away from the LLWR). The residency may differ for external and internal dose rates to account for species that may gather food from only one ecosystem but spend some time in a different ecosystem.

For the dose calculations in the ERICA Tool, radioactive decay and ingrowth are integrated over one year. For representative organisms with short lifetimes or lifecycle stages, this calculation will lead to a cautious assessment as the organism, or the organism at the assessed life stage, will not be exposed to a year of ingrowth.

3.2.3 Representative Organisms Used in this Assessment

The representative organisms assessed are listed in Table 3.2. Their characteristics are given in reference [24].

[2] Except for C-14 and tritium, for which the main exposure pathways are release of gas and for which specific activity models are used. These are described in more detail in Subsection 4.4.2.2.

Table 3.2: Representative organisms used in our NHB assessment

Terrestrial	Freshwater	Marine
Amphibian	Amphibian	Bird
Annelid	Benthic fish	Bivalve mollusc
Arthropod - detritivorous	Bird	Burrowing bivalve mollusc
Bird	Bivalve mollusc	Burrowing echinoderm
Flying insect	Crustacean	Cliff-nesting seabird
Gastropod mollusc	Gastropod mollusc	Crab
Grasses and herbs	Insect larvae	Flat fish
Lichens and bryophytes	Mammal	Gastropod mollusc
Mammal - large	Pelagic fish	Macroalgae
Mammal - large burrowing	Phytoplankton	Mammal
Mammal - small burrowing	Vascular plant	Pelagic fish
Mammal - small on-soil	Zooplankton	Phytoplankton
Mammal (semi-aquatic)		Polychaete worm
Reptile		Reptile
Shrub		Sea anemone & true coral
Small flying bird		Small benthic crustacean
Small flying mammal		Vascular plant
Tree		Zooplankton

In Table 3.3, Table 3.4, and Table 3.5, important species present in and around the LLWR site are mapped to the representative organisms used in this assessment. This demonstrates that all the important species, especially those that are protected or that live on protected sites, are covered by the assessment.

Table 3.3: Marine biota - mapping species to representative organisms

Representative Organism	Species: Common name (*Latin name*)	Conservation status
Bird	Oystercatcher (*Haemotopus ostralegus*)	Threatened
	Sandwich tern (*Sterna sandvicensis*)	Threatened
	Common tern (*Sterna hirundo*)	Threatened
	Little tern (*Sternula albifrons*)	Protected
Bivalve mollusc	Blue mussel (*Mytilus edulis*)	Common
	Barnacle (*Semibalanus balanoides*)	Common
	Piddocks (*Pholas dactylis*)	Common
Burrowing bivalve mollusc	Razor shell (*Ensis spp.*)	Common
	Baltic tellin (*Macoma balthica*)	Common
Burrowing echinoderm	Sea potato (*Echinocardium cordatum*)	Common
Cliff-nesting seabird	Razorbill (*Alca torda*)	Threatened
	Black guillemot (*Cepphus grylle*)	Common
	Northern fulmar (*Fulmarus glacialis*)	Threatened
	Atlantic puffin (*Fratercula arctica*)	Threatened
	Shag (*Gulosus aristotelis*)	Threatened
	Kittiwake (*Rissa tridactyla*)	Threatened
Crab	Shore crab (*Carcinus maenas*)	Common
Flat fish	Common sole (*Solea solea*)	Threatened
Gastropod mollusc	Edible periwinkle (*Littorina littorea*)	Common
Macroalgae	Sugar kelp (*Saccharina latissima*)	Common
	Spiral wrack (*Fucus spiralis*)	Common

Representative Organism	Species: Common name (*Latin name*)	Conservation status
	Bladder wrack (*Fucus vesiculosus*)	Common
	Egg wrack (*Ascophyllum nodosum*)	Common
	Horned wrack (*Fucus ceranoides*)	Common
Mammal	Grey seal (*Halichoerus grypus*)	Protected
Pelagic fish	Mackerel (*Scomber scombrus*)	Threatened
	Atlantic salmon (*Salmo salar*)	Protected
Phytoplankton	Phytoplankton	Common
Polychaete worm	Honeycomb worm (*Sabellaria alveolata*)	Protected
	Lugworm (*Arenicola marina*)	Common
	Ragworms (*Hediste diversicolor*)	Common
Reptile	Leatherback turtle (*Dermochelys coriacea*)	Threatened
Sea anemone & true coral	Strawberry anemone (*Actinia fragacea*)	Common
Small benthic crustacean	Burrowing amphipod (*Corophium volutator*)	Common
	Isopod (*Eurydice pulchra*)	Common
Vascular plant	Lax-flowered sea-lavender (*Limonium humile*)	Common
	Saltmarsh grass (*Puccinellia maritima*)	Common
	Jointed rush (*Juncus articulatus*)	Common
	Sea kale (*Crambe maritima*)	Threatened
	Sea-blite (*Suaeda maritima*)	Common
	Glasswort (*Salicornia europea*)	Common

Representative Organism	Species: Common name (*Latin name*)	Conservation status
	Sea rush (*Juncus maritimus*)	Common
Zooplankton	Zooplankton	Common
Reference [25] gives the sources of the threatened and protected designations listed.		

Table 3.4: Freshwater biota - mapping species to representative organisms

Representative Organism	Species: Common name (*Latin name*)	Conservation status
Amphibian	Natterjack toad (*Epidalea calamita*)	Protected
	Great crested newt (*Triturus cristatus*)	Protected
	Smooth newt (*Lissotriton vulgaris*)	Protected
	Palmate newt (*Lissotriton helveticus*)	Protected
	Common frog (*Rana temporaria*)	Protected
	Common toad (*Bufo bufo*)	Protected
Benthic fish	Bullhead (*Cottus gobio*)	Common
Bird	Mallard (*Anas platyrhynchos*)	Protected
Bivalve mollusc	Freshwater pearl mussel (*Margaritifera margaritifera*)	Protected
Crustacean	Amphipod (*Gammarus sp.*)	Common
Insect larvae	Golden-ringed dragonfly (*Cordulegaster boltonii*)	Common
	Emperor dragonfly (*Anax imperator*)	Common
	Moorland hawker (*Aeshna juncea*)	Common
Gastropod mollusc	New Zealand mud snail (*Potamopyrgus antipodarum*)	Common

Representative Organism	Species: Common name (*Latin name*)	Conservation status
Mammal	Otter (*Lutra lutra*)	Protected
Pelagic fish	Atlantic salmon (*Salmo salar*)	Protected
Phytoplankton	Phytoplankton	Common
Vascular plant	Jointed rush (*Juncus articulatus*)	Common
	Pillwort (*Pilularia globulifera*)	Protected
Zooplankton	Zooplankton	Common
Reference [25] gives the sources of the protected designations listed.		

Table 3.5: Terrestrial biota - mapping species to representative organisms

Representative Organism	Species: Common name (*Latin name*)	Conservation status
Amphibian	Natterjack toad (*Epidalea calamita*)	Protected
	Great crested newt (*Triturus cristatus*)	Protected
	Smooth newt (*Lissotriton vulgaris*)	Protected
	Palmate newt (*Lissotriton helveticus*)	Protected
	Common frog (*Rana temporaria*)	Protected
	Common toad (*Bufo bufo*)	Protected
Annelid	Common earthworm (*Lumbricus terrestris*)	Common
Arthropod - detritivorous	Black oil beetle (*Meloe proscarabaeus*)	Threatened
	Northern dune tiger beetle (*Cicindela hybrida*)	Threatened
Bird	Mallard (*Anas platyrhynchos*)	Protected
	Oystercatcher (*Haemotopus ostralegus*)	Threatened

Representative Organism	Species: Common name (*Latin name*)	Conservation status
	Buzzard (*Buteo buteo*)	Common
	Barn owl (*Tyto alba*)	Protected
Flying insect	Common carder bee (*Bombus pascuorum*)	Common
	Golden-ringed dragonfly (*Cordulegaster boltonii*)	Common
	Moorland hawker (*Aeshna juncea*)	Common
Grasses & Herbs	Pennyroyal (*Mentha pulegium*)	Protected
	Yellow bartsia (*Parentucellia viscosa*)	Protected
	Cross-leaved heath (*Erica tetralix*)	Common
	Sand sedge (*Carex arenaria*)	Common
	Field gentian (*Gentianella campestris*)	Common
	Cranberry (*Vaccinium oxycoccos*)	Common
	Red fescue (*Festuca rubra*)	Common
	Lax-flowered sea-lavender (*Limonium humile*)	Common
	Jointed rush (*Juncus articulatus*)	Common
	Saltmarsh grass (*Puccinellia maritima*)	Common
Lichen & bryophyte	Oakmoss (*Evernia prunastri*)	Common
	Hoary fringe-moss (*Racomitrium canescens*)	Common
Mammal - large	Roe deer (*Capreolus capreolus*)	Protected
Mammal - large burrowing	Badger (*Meles meles*)	Protected
	Red fox (*Vulpes vulpes*)	Common
Mammal - small burrowing	Mole (*Talpa europaea*)	Common

Representative Organism	Species: Common name (*Latin name*)	Conservation status
Mammal - small on-soil	Brown hare (*Lepus europaeus*)	Protected
Mollusc - gastropod	Garden snail (*Cornu aspersum*)	Common
Reptile	Slow worm (*Anguis fragilis*)	Protected
	Common lizard (*Zootoca vivipara*)	Protected
	Adder (*Vipera berus*)	Protected
Mammal (semi-aquatic)	Otter (*Lutra lutra*)	Protected
Shrub	Common gorse (*Ulex europaea*)	Common
	Broom (*Cytisus scoparius*)	Common
	Sea wormwood (*Artemisia maritima*)	Common
	Heather (*Calluna vulgaris*)	Common
	Bell heather (*Erica cinerea*)	Common
	Dyer's greenweed (*Genista tinctoria*)	Common
	Creeping willow (*Salix repens*)	Common
	Sea purslane (*Atriplex portulacoides*)	Threatened
Small flying bird	Common whitethroat (*Sylvia communis*)	Threatened
	Dunnock (*Prunella modularis*)	Threatened
	Linnet (*Carduelis cannabina*)	Threatened
	Meadow pipit (*Anthus pratensis*)	Threatened
	Reed bunting (*Emberiza schoeniclus*)	Threatened
	Skylark (*Alauda arvensis*)	Threatened

Representative Organism	Species: Common name (*Latin name*)	Conservation status
	Song thrush (*Turdus philomelos*)	Threatened
	Willow warbler (*Phylloscopus trochilus*)	Threatened
Small flying mammal	Noctule bat (*Nyctalus noctula*)	Protected
	Common pipistrelle (*Pipistrellus pipistrellus*)	Protected
	Soprano pipistrelle (*Pipistrellus pygmaeus*)	Threatened
	Brown long-eared bat (*Plecotus auritus*)	Protected
Tree	Corsican pine (*Pinus nigra*)	Common
	Alder (*Alnus glutinosa*)	Common
Reference [25] gives the sources of the threatened and protected designations listed.		

3.3 Dose Rate Screening

There are currently no statutory numeric criteria for radiological protection of NHB in the UK. However, European-funded projects [33], the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) [34], the ICRP [35] and the IAEA [36] have evaluated the academic literature on the effects of ionising radiation on wildlife and proposed methods and numeric criteria for use in dose assessments for wildlife. The FREDERICA database [37] contains effects data of suitable quality and representing types of effects endpoints related to morbidity, reproduction and mortality and has been used as the basis for several of these evaluations.

The environment agencies use a screening value of 10 $\mu Gy\ h^{-1}$ for initial assessments of radiological impacts from a single premises on protected species and sites [20]. Where this is exceeded, they would do a more detailed, site-specific assessment to better understand the radiological impacts. The Environment Agency also use a screening value of 40 $\mu Gy\ h^{-1}$ to screen the radiological impacts on protected sites of all permitted discharges from all sites that may affect the site being assessed [38].[3]

In our 2011 ESC we also used a screening value of 10 $\mu Gy\ h^{-1}$; the basis for this value is given in reference [39]. The Environment Agency accepted its use for our 2011 ESC [39], it

[3] The GRA and GRR do not require operators to assess against this standard or show that it would be met. Because such an assessment would require considering (and potentially forecasting) the discharges of other operators, it is not something that an individual operator could do.

is suggested for protected sites in the GRR [20] and it is applied in the internationally recognised ERICA approach [22]. Therefore, we consider that this screening value remains appropriate, and we continue to use 10 µGy h^{-1} as our screening value in this assessment. Where dose rates exceed this value, we consider the radiological impacts to the organism in more detail (Subsection 3.4).

3.4 Assessment of Representative Organisms with Assessed Dose Rates Above the Screening Criterion

We expect populations of organisms receiving dose rates below the screening dose rate to be protected. However, assessed dose rates above the screening dose rate do not necessarily mean that a population of interest will not be protected. Instead, where this dose rate is exceeded, we evaluate the likely impacts in more detail.

We apply a hierarchical approach, in which we consider each of the following in turn:

- time-dependency of calculated radionuclide concentrations and, thus, dose rates;
- radiosensitivity, e.g. with reference to ICRP Derived Consideration Reference Levels (DCRLs);
- size of the contaminated or affected area relative to the area required to support a population of the affected organisms;
- notable uncertainties and pessimisms associated with the modelling approach.

By completing such an evaluation, we can show that the protection objective (Subsection 1.1) is met.

3.4.1 Derived Consideration Reference Levels (DCRLs)

The DCRLs are dose-rate bands recommended by the ICRP. The ICRP define a DCRL as "*a band of dose rate, spanning one order of magnitude, within which there is some chance of deleterious effects from ionising radiation occurring to individuals of that type of* [reference animal or plant]" [32]. Because different DCRLs are specified for different categories of reference animal and plant, the DCRLs can be used to account for differences in radiosensitivity between different types of organism (Figure 3.1).

The DCRLs are not limits [32] but provide information that can be used with the representative organisms to help us understand the consequences of a dose to that type of organism (Subsection 3.2). Typically, because DCRLs are not limits, when the estimated dose rates to the representative organisms are within or close to the upper bound of the band, the radiological impacts can still be considered acceptable. However, factors such as the size of the area where the dose rates are assessed to occur; the time periods (e.g. for migratory species) predicted for such dose rates; the need to comply with specific legislation; and the degree of precaution needed should be considered [36].

By comparing the calculated dose rates to our representative organisms with the appropriate DCRL, we can evaluate how the radiosensitivity of the organism type may influence the likely effect on individuals. From this, we can determine whether the assessed dose rates would lead to detrimental impacts on the population.

In summary, we use the DCRLs to help understand whether assessed dose rates meet the protection requirements set out in the GRA and GRR, but the DCRLs are not themselves the protection requirement.

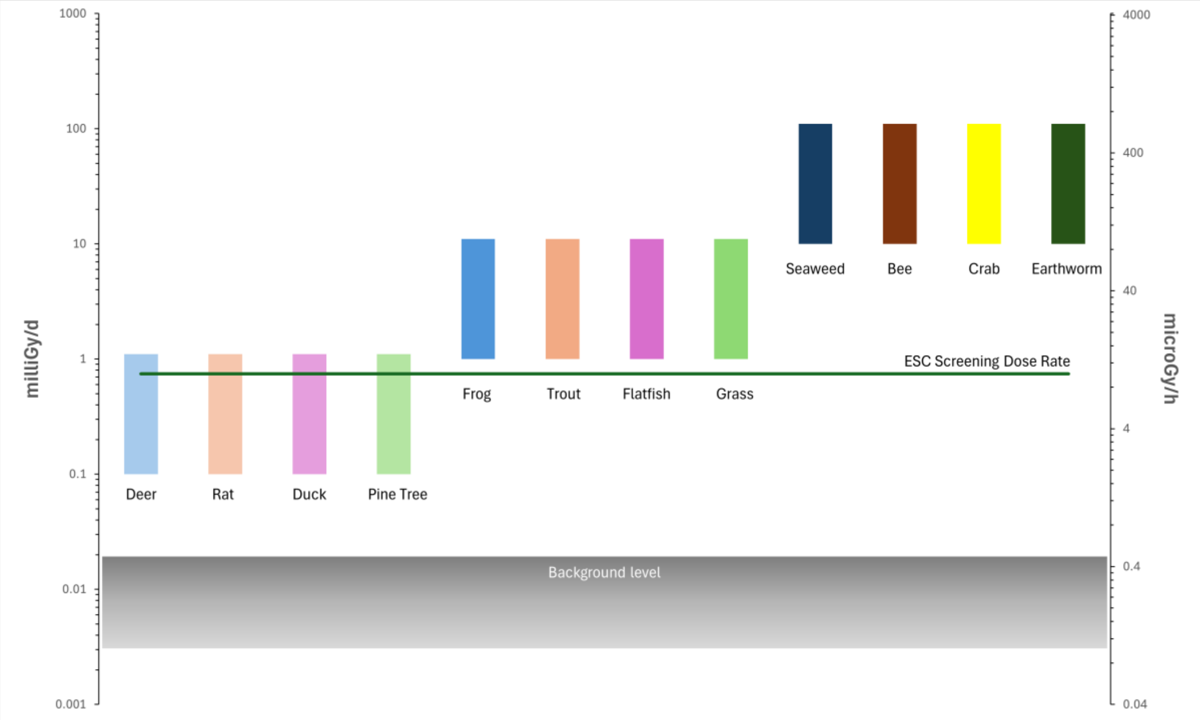


Figure 3.1: Derived Consideration Reference Levels (DCRLs) for the ICRP Reference Animal and Plants for use in environmental assessments modified from references [32], [35]

4 Assessment Approach

4.1 Approach Overview

We assess the dose rates to biota arising from release of radioactivity from the LLWR using the ERICA tool [23]. We describe our approach at a high level in this section, and it is shown in Figure 4.1. In addition to the environmental radionuclide concentrations shown in Figure 4.1, external irradiation from waste stacks during the PoA and potential for contamination to be released in particulate form are given bespoke consideration. The approaches to these assessments are described in Subsections 4.3.1.8 and 4.3.2.6.

We give more details about our approach in the following subsections and in reference [24].

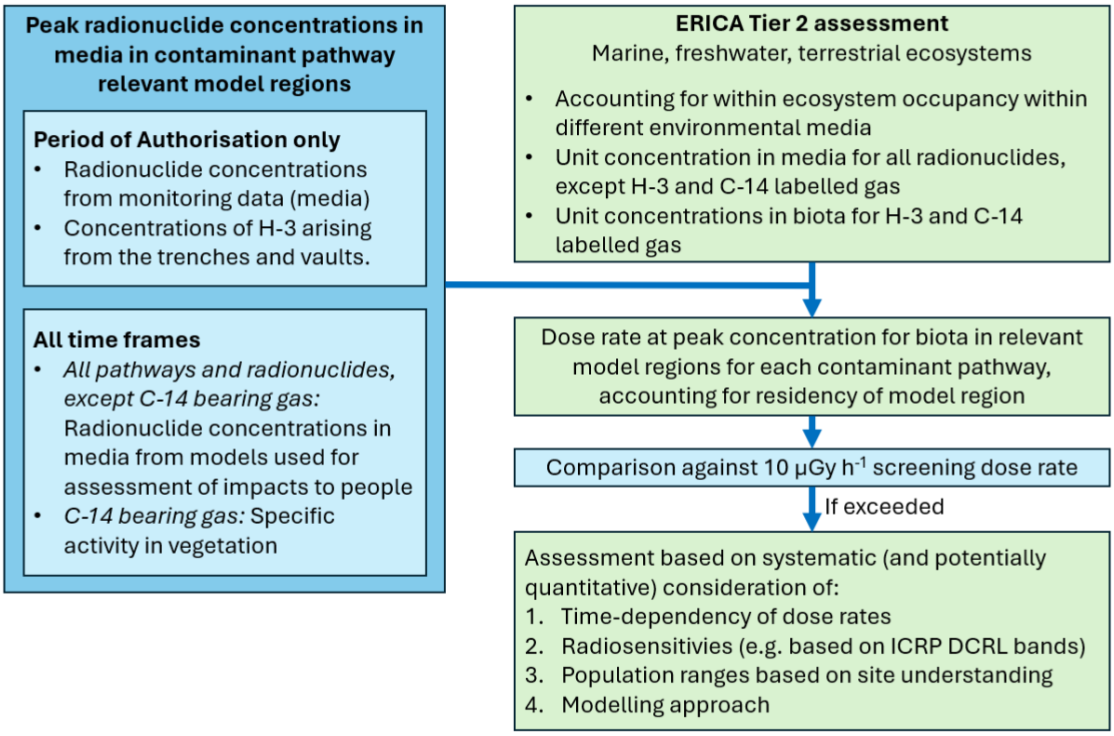


Figure 4.1: High-level schematic of assessment approach

The assessments of radiological impact to people are based on models of the transfer of radioactivity through the environment (and, in the case of the assessment of radiological impacts during the PoA, on discharge and monitoring data).This is detailed for each pathway in Subsection 4.3. As well as doses to people, these models calculate the activity concentrations of radionuclides in different environmental media such as soil, sediment, water and air. To assess dose rates to biota, we use these media activity concentrations as the input to ERICA. This means we ensure that the same understanding of release and transport of radionuclides is used in both the assessments of impacts to people and biota. It also means we assess the impact on biota of every radionuclide for which we assess impacts to people.[4]

The media activity concentrations from the assessments of impacts to people are used as inputs to the latest version of the ERICA tool available at the time of the calculations (version 2.0.228). The ERICA tool is used to assess the dose rates resulting from these concentrations. Dose rates are calculated for a set of representative organisms defined to cover the range of species present on and near the LLWR site, including those on nearby conservation sites. The representative organisms are described in more detail in Subsection 3.2.3.

For each organism, the media activity concentration is multiplied by a concentration ratio (CR) for each to estimate the internal whole body activity concentration radionuclide (except for C-14 and tritium which are assessed as described below). The CR assumes equilibrium has been established between the media and biota of interest and is a measure of the activity concentration in the whole body of the biota of interest relative to the activity concentration of the media. Compilations of CR are available in the literature (e.g. in references [40, 41]). This is then multiplied by an internal dose coefficient [30] to give the internal dose rate.

The external dose is determined by multiplying the media activity concentration by the appropriate dose coefficient [30] specific to the type of organism and its location in the environment relative to the media (Table 3.1) and by the occupancy of the organism in the location. If the organism spends time in more than one location, this is repeated for each location and added together to give the total external dose rate. For example, a seabird may roost in the terrestrial environment but forage exclusively in the marine environment or a deer may spend only a proportion of its time in the contaminated areas.

We assess most organisms separately in each ecosystem, assuming they spend all their time in each ecosystem. This ensures the assessment considers the most exposed individual or species covered by each representative organism. However, where the split between ecosystems is inherent to a species' habits (for example, a seabird that spends time both on land and in water), we assess the organism for a fraction of its time in each relevant ecosystem and add the dose rates from each ecosystem together to give the total dose rate. Similarly, if an organism spends time away from the repository, we scale the dose rates to account for the fraction of time spent elsewhere. That is, an individual exposed to 10 μGy h^{-1} for half a year has an average dose rate of 5 μGy h^{-1}.

For each organism, the dose rates arising from the maximum concentration of each radionuclide to the environment are added together. This gives the dose rate at maximum concentrations for each radionuclide of interest, for initial comparison to the 10 μGy h^{-1} screening value. The maximum concentrations of different radionuclides will occur at different times; the dose rate at maximum concentration is, therefore, a very cautious value.

[4] Except for Am-242m. This radionuclide cannot be assessed using the ERICA tool but would be only a very minor contributor to radiological impact to biota [58].

Where the screening value is exceeded, more detailed evaluations of the representative organisms and exposure pathways are performed, as described in Subsection 3.4.

For some pathways the assessment approach deviates from this because environmental media concentrations are not calculated in the assessments of impacts to people or are not applicable to the pathway. These are the tritium, C-14 and radon gas pathways, external dose from uncovered waste[5] while the repository is operating and dose from exposure to radioactive particles released during coastal erosion.

The assessments of impacts to people use specific activity models for the tritium and C-14 gas pathways [14, 15]. These models also allow biota tissue concentrations to be calculated on the same principles. We then use these concentrations as an input to the ERICA tool to assess dose rates to terrestrial organisms. External dose from gaseous releases of these radionuclides is negligible and not considered. The gas pathway only results in exposure of terrestrial organisms. For aquatic organisms, we do not use the specific activity models; concentrations of radionuclides in water bodies are calculated in the assessments of impacts to people, and we use these concentrations in the same way as for all other radionuclides.

The Rn-222 assessments of impact to people calculate soil-gas and above-ground air concentrations of Rn-222 (Rn-220 is not assessed because its short half-life means it will decay to insignificant activity while migrating through the cap) [14, 15]. We use these concentrations in the ERICA tool to calculate external dose rates arising from immersion in air for all representative organisms except plants and inhalation dose rates for mammals. Currently, there is insufficient understanding to extend the ERICA inhalation model to non-mammals. We, therefore, perform some simple calculations based on the decay characteristics of Rn-222 to understand the likely magnitude of internal and external dose rates to organisms present in the soil, including uptake to tissue by diffusion. Our Rn-222 assessment approach is fully described in Appendix D1 in reference [24].

External dose from uncovered waste during the PoA arises from gamma emissions from containers emplaced in uncovered vaults. The assessment of impacts to people [14] assesses this by modelling photon emissions and scattering and, thus, fluxes and dose rates to human receptors. We use the same photon fluxes to assess dose rates to biota. During the PoA, biota may be in different places to members of the public; therefore, we also use the models to calculate photon fluxes and biota dose rates in places where biota may be, but members of the public will not be.

We do not model direct interactions between biota and waste during the PoA or as a result of human intrusion after the PoA. During the PoA, while biota may be able to access the vault and receive an external dose, we do not expect extensive use of the vaults by biota during the PoA as the LLWR is managed in a way that should limit the potential for biota to interact

[5] We use the term 'waste" in our report. Here waste refers to a mixture of waste and grout, and other packaging or engineering materials.

with the waste. Additionally, the ISO containers and the grouted wasteform will limit opportunities for biota to interact with waste. This does not preclude the possible exposure of individual plants and animals but does limit the potential for chronic exposure to a population of a given species. Post PoA, the coastal erosion assessment effectively considers the interaction of NHB with the exposed waste. This will be bounding for interactions between biota and waste that could occur following human intrusion.

Radioactive particles could be released from the repository following coastal erosion or human intrusion. The habits of some organisms (such as filter feeders) mean there is more potential for significant exposure to radioactive particles for biota than for people. We have cautiously considered the inventory could be released from the repository in radioactive particles. We assessed the impacts of this release using a calculation of the dose rates local to individual particles.

4.2 Radionuclides and Inventory

The environmental media concentrations used in this assessment are based on monitoring data and models that use the reference inventory [14, 4]. The reference inventory includes both low-level and intermediate-level waste that would potentially be suitable for vault disposal. The radionuclides in the inventory have been screened for radiological impact to determine which to assess [42]. We assess only those radionuclides identified by the screening; we do not assess all the radionuclides in the inventory. Thus, we assess the same set of radionuclides considered in the assessments of impacts to people.

Unlike in the assessments of radiological impact to people, we do not derive numerical controls on waste acceptance based on radiological impact to biota. Instead, we show that the reference inventory will not lead to unacceptable radiological impacts on populations of biota. We believe this is enough to show that the protection objective set out in Subsection 1.1 is met.

The capacities and waste acceptance criteria calculated from the assessments of impacts to people are based on assessments of doses and risks to people and dose and risk criteria set out by the Environment Agency. There are no equivalent criteria for biota. Different species can have very different radiosensitivities; using the 10 $\mu Gy\ h^{-1}$ screening dose rate to calculate capacities would lead to overly restrictive capacities for many species. It is not considered to be proportionate to attempt to derive dose rate criteria for all the species we assess, nor would there be sufficient underpinning scientific understanding to do this.

The GRA [19] explains why the assessment of radiological impacts to NHB is required:

> **¶6.3.72**
>
> Although there is no specific evidence that there might be a threat to populations of non-human species from the authorised release of radioactive substances if people are protected, there may be times when there are no people near a disposal facility. Environmental damage might also occur to areas and habitats that are not extensively exploited by people. Furthermore, there is a specific need to be able to demonstrate that non-human species are protected under legislation related to conservation.

We do not know where people or biota will live in the future. Our assessments therefore assume that people and NHB will be exposed to contamination, and our assessments of impacts to people cover pathways and locations that are similar to or the same as our assessment of impacts to NHB (see Subsection 4.3). The contaminant transport assumptions used mean that we consider exposures that are either similar to or worse than would be expected on nearby designated conservation sites.

Based on this, we consider that there are no locations or habitats in which biota may be exposed that are not accounted for in the derivation of capacities and waste acceptance criteria based on impacts to people. We therefore consider that the capacities and waste acceptance criteria derived from our assessments of impacts to people from different inventories will be sufficient to ensure that biota are protected, even if the inventory accepted is different to our reference inventory.

We have a programme of formal updates for our ESC (such as periodic reviews and major reviews). Through these, we incorporate updates in our understanding of the inventory into the ESC. The assessment of radiological impacts to non-human biota is part of these updates. Therefore, we will ensure that the impacts to NHB of changes to the inventory are assessed and within regulatory guidance.

4.3 Pathways

Radioactivity from the LLWR site can enter the environment from releases of radionuclides to surface water or groundwater, releases of radionuclides to air, from coastal erosion or from disturbance of waste during human intrusion. While the repository is operating, biota may be exposed to external irradiation from uncovered vaults. When the repository is eroded by the coast or if it is disturbed by human intrusion, exposures to radioactive particles could occur.

Different pathways will be relevant at different times and in different locations. The assessments of radiological impact to people consider the pathways applicable during [14] and after [15] the PoA. Biota could be present in different locations to people and have different habits. Therefore, we need consider whether the environmental concentrations used in the assessments of radiological impacts to people are appropriate for biota.

Our assessments of radiological impacts to people are based on our best understanding of contaminant transport and potential human habits. In our post-PoA models, radionuclide concentrations are calculated for those locations where we expect radioactivity originating from the LLWR to be present. Because the calculations of radionuclide concentrations do not rely on assumptions about the likely presence of people or biota in specific locations, it is appropriate to use the same activity concentrations for assessments of post-PoA impacts to both people and biota.

During the PoA, there are exceptions to this approach for exposure to radioactive gas and direct and scattered radiation. Early in the PoA, while the site is operating, members of the public will not have access; later in the PoA we expect public access to be allowed, but the site will be managed, and appropriate controls will be applied. The assessment of radiological impacts to people, therefore, focuses on assessing doses to people offsite. However, some types of biota will have access to the site during the PoA (although the site will be managed and, for example, there will be limits on the types, residencies and habits of biota in different areas of the site). We have calculated gas concentrations and external dose rates on site for this assessment [24].

We have an environmental monitoring programme that measures radionuclide concentrations in environmental media on and near the LLWR site [9]. We use the results from this monitoring programme in our assessments of impacts to people [14]. In addition, a previous independent study [43] investigated radionuclide concentrations in the surface environment, which include contributions from the LLWR, Sellafield, other anthropogenic sources, and naturally occurring radionuclides, and found these to be low.

The following sections provide an overview of the environmental radionuclide concentrations that are used to assess the doses to NHB. Details of the derivation of the environmental concentrations for use in the assessment are given in the reports on the human assessments [14] [15].

4.3.1 During the PoA

The exposure pathways applicable during the PoA are:

- migration of radionuclides in groundwater and discharge to the foreshore;
- release of C-14, tritium and Rn-222 gases;
- discharge of radionuclides from the leachate management system in the following situations:
    - routine, managed discharges of leachate to the foreshore via the marine pipeline;
    - occasional discharges of leachate to the Drigg Stream during storms and subsequent flooding of pasture;
- contamination of the sediment in the Drigg Stream because of historical discharges;
- direct and scattered radiation from uncovered vaults.

These pathways are mapped to the principal locations in which the biota could be exposed to them in Table 4.1.

Table 4.1: Exposure pathways during the PoA and the principal associated locations where biota could be exposed

Pathway by which biota may be exposed	Biota location and habitat
Migration in groundwater and discharge	Foreshore (marine)
	Local coastal area (marine)
	Lower Irt (freshwater and terrestrial)
Release of C-14 bearing gas	LLWR site (above cap) (terrestrial)
Release of Rn-222 gas	LLWR site (above cap) (terrestrial)
Release of tritium bearing gas	LLWR site (above cap) (terrestrial)
Managed leachate discharge via marine pipeline	Foreshore (marine)
	Local coastal area (marine)
Storm discharge to stream resulting in flooding	Nearby pasture (terrestrial)
Historical contamination of stream	Drigg Stream (freshwater)
Direct and scattered radiation	LLWR site (terrestrial)

The PoA assessment [14] also assesses potential radiological impacts to people arising from abstraction of groundwater and use of the abstracted water as drinking water during part of the PoA. This is not considered likely to occur because the NDA own the lease on the land between the LLWR and the coast, and the PoA assessment considers this a “what-if” assessment. Abstraction of groundwater for use as drinking water for people would not lead to any exposure route for non-human biota. Therefore, we do not assess the radiological impacts of the well pathway on NHB during the PoA.

Data covering the PoA have been taken from:

- Level 3 PoA Assessment [44]: Fluxes of tritium and Rn-222 gases, and modelled environmental concentration data for other reference case releases, including concentrations in marine model regions, historical stream contamination and concentrations in flooded soil.

- Level 3 Gas Assessment [45]: Concentrations of C-14 in vegetation following release as gas during the PoA for the reference case.

- Level 3 Groundwater Assessment [46]: Radionuclide concentrations in the foreshore, marine and estuarine environments for the reference case.

4.3.1.1 Migration of Radionuclides in Groundwater and Discharge to Surface

Radionuclides released in groundwater during the PoA are transported towards the coast. The groundwater passes through the sub-surface lithofacies and discharges to the marine environment via the intertidal zone and foreshore. The groundwater pathway assessment [46] provides the resulting radionuclide concentrations in the foreshore, local coastal waters and Irt estuary. We use these concentrations for the NHB assessment.

4.3.1.2 Release of Gaseous C-14

The degradation of waste in both the trenches and vaults will produce C-14 bearing carbon dioxide and methane. There is potential for C-14 bearing gas be transported through the cap with bulk gases. The model described in reference [45] represents:

- release of C-14 from waste;

- transport of C-14 through the capping layers;

- oxidation of C-14 bearing methane to C-14 bearing carbon dioxide in the soil;

- uptake of C-14 by vegetation via root uptake and by photosynthesis in the subcanopy atmosphere.

The model calculates the specific activity of C-14 in vegetation (Bq C-14 per kg stable carbon) over the trenches and the vaults. These specific activities are used in the NHB assessment to calculate the activity concentrations in the tissues and organs of biota on a fresh mass basis.

The model assumes that C-14 released from the waste as methane is oxidised to carbon dioxide while passing through the cap. If this did not happen, the C-14 bearing methane would be released from the soil to the atmosphere and dispersed. It would not be taken up by plants via photosynthesis. The radiological impacts of exposing plants to C-14 bearing methane would, therefore, be very small (C-14 is a beta emitter with negligible external dose coefficients). As the C-14 would not be taken up by the plant, it would also not be transferred to other organisms (or people) through the food chain. Doses to other organisms would, therefore, also be very small.

Carbon dioxide, however, can be taken up by plants from the root zone in the transpiration stream and from the plant canopy atmosphere in gaseous form. By either route it can be incorporated into the plants via photosynthesis. This would lead to internal dose to the plants and transfer of C-14 through the food chain, leading to internal dose to other organisms in the ecosystem. Therefore, this pathway is more radiologically significant, and we assume complete oxidation in our assessment models. Experimentally, almost complete oxidation is observed in well-aerated soils.

4.3.1.3 Release of Gaseous Tritium

There is potential for tritium to be released as tritiated water vapour. The PoA assessment [44] calculates fluxes of tritium released as water vapour to air above the trenches and vaults. The release is driven by water from precipitation and infiltration reaching the waste; access of this infiltrating water to the waste effectively ceases as the final cap is progressively installed. Once the final cap is in place, gas generation is assumed to stop. We have used these fluxes, together with an understanding of the meteorological conditions and atmospheric dispersion, to calculate the average concentrations of tritium in air above the trenches and vaults during the PoA. We use these concentrations to calculate biota tissue concentrations for the NHB assessment.

The modelled flux to the atmosphere and resulting atmospheric concentrations of tritium during the PoA are shown in Figure 4.2.

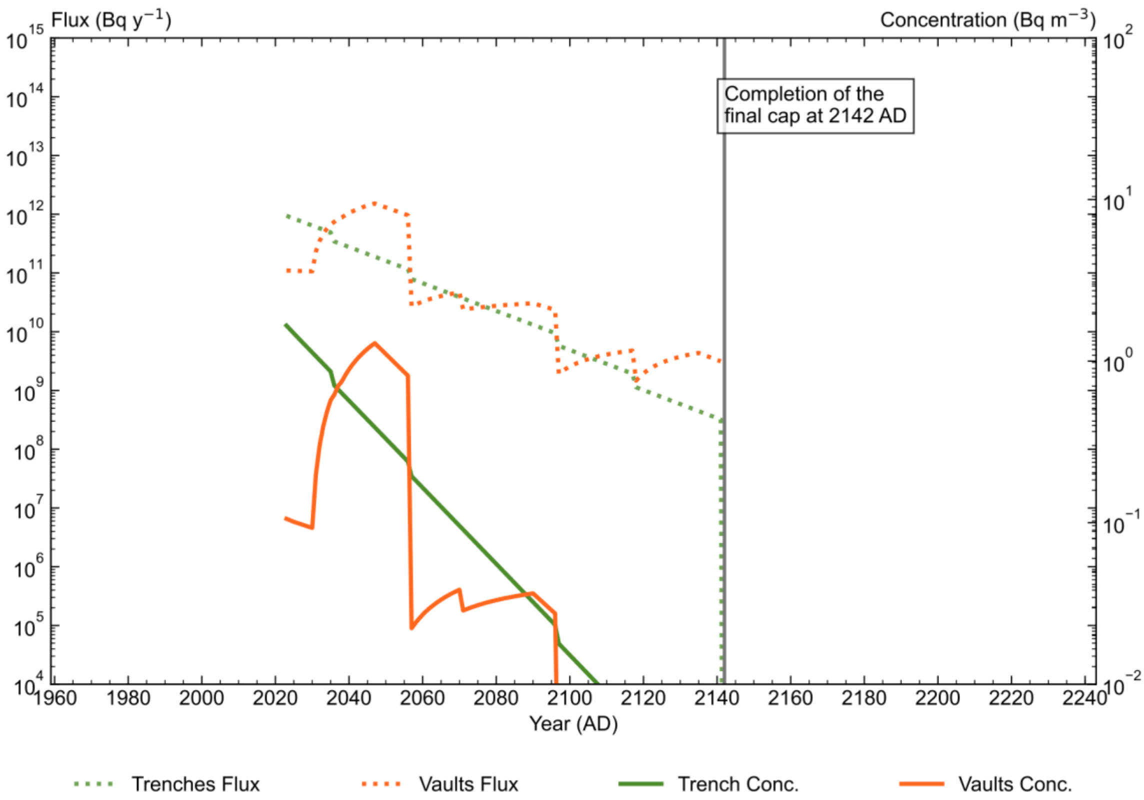


Figure 4.2: Modelled flux of tritium to the surface (Bq y^{-1}) and concentration of tritium in the aboveground atmosphere (Bq m^{-3}) in the PoA assessment (data from reference [44])

4.3.1.4 Release of Gaseous Rn-222

Rn-222 will be released from the waste. The PoA assessment [44] calculates fluxes of Rn-222 gas to air above the trenches and vaults. The PoA model assumes that radon emerges from areas that have not yet been covered by the final cap [44].

The step-wise way that the final cap is emplaced means that releases from the trenches are increasingly confined to a smaller area until the final cap is completed in 2142 AD; this is reflected in the step-wise increases in concentrations associated with the trenches in Figure 4.3.

The pattern of radon releases from the vaults is complicated because of the phased way in which the vaults are constructed, operated, closed and capped, so an atmospheric concentration is not shown in Figure 4.3. The highest radon concentrations associated with the vaults in reference [44] occur in 2071 AD, when Vaults 8, 9 and part of Vault 10 are capped, with the associated fluxes emerging over the remaining uncapped area of Vault 10; this concentration has been used as a basis for NHB dose calculations associated with the vaults.

Once the final cap is completed, radon is taken to emerge through a vent, and the average concentration over the cap is calculated in the PoA model and shown in Figure 4.3.

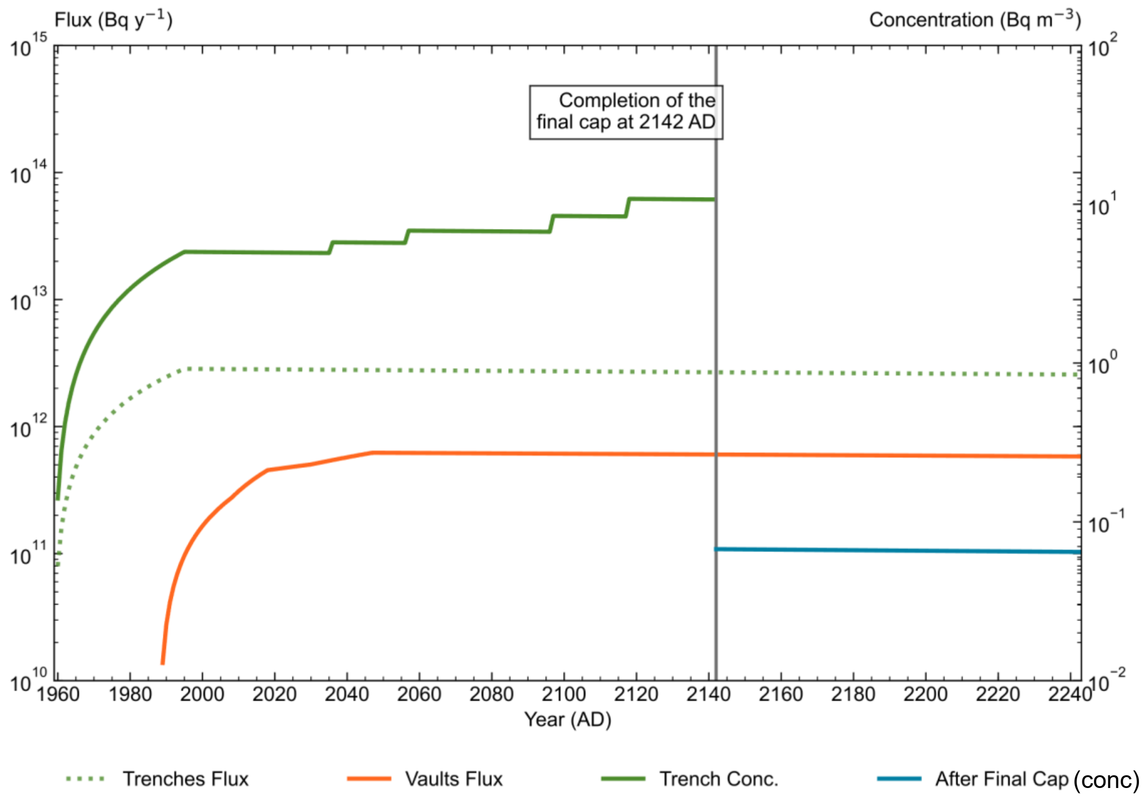


Figure 4.3: Modelled flux of Rn 222 from the trenches and vaults to the surface (Bq y^{-1}) and activity concentration of Rn 222 in the aboveground atmosphere above the trenches (Bq m^{-3}) in the PoA assessment

4.3.1.5 Routine, Managed Discharges through the Marine Pipeline

The PoA assessment [44] models radionuclides released to the environment via the marine pipeline. The resulting radionuclide concentrations in water (Bq m^{-3}) and sediment (beach and bed sediment, Bq kg^{-1}) are shown in Figure 4.4. The highest modelled concentrations are in the local offshore and beach model regions, with the activity reducing further out to sea.

Initially, the radionuclides dominating activity in the waters close to the LLWR site are tritium and C-14. The steep decline in radionuclide concentrations around 2142 AD reflects the end of disposal activities and emplacement of the final cap at the LLWR site. We plan to decommission the marine discharge pipeline by the end of the PoA.

While discharges to the marine environment are expected to decline substantially after 2142 AD, some radionuclides sorb to the sediments and, therefore, gradually accumulate in the marine sediment in the local compartment during the period of radioactive leachate discharges. Radionuclides sorbed to sediments can become resuspended in seawater and will gradually desorb, providing an ongoing source of radioactivity in the water after discharges have ended.

Following final cap emplacement, the greatest contributor to the modelled radionuclide concentrations in both marine sediment and seawater is Pu-239. This is because it is a strongly sorbed radionuclide, which is less mobile in the environment. Therefore, it builds up in the sediment while discharges are occurring and is slowly released from the sediment after discharges end. More mobile radionuclides will disperse more rapidly so are present at lower concentrations after discharges end.

The Pu-239 activity concentration used in this calculation is a surrogate based on measurements of the total alpha activity discharged. This surrogacy was also used for the assessment of impacts of managed marine discharges to people and is justified in reference [44]. The actual behaviour of alpha-emitting radionuclides in the environment will depend on the fingerprint discharged. However, we expect the other significant alpha-emitting radionuclides to be heavy (lanthanide or actinide) elements with similar physicochemical behaviour to plutonium. Therefore, we consider that using Pu-239 is sufficiently representative for this assessment.

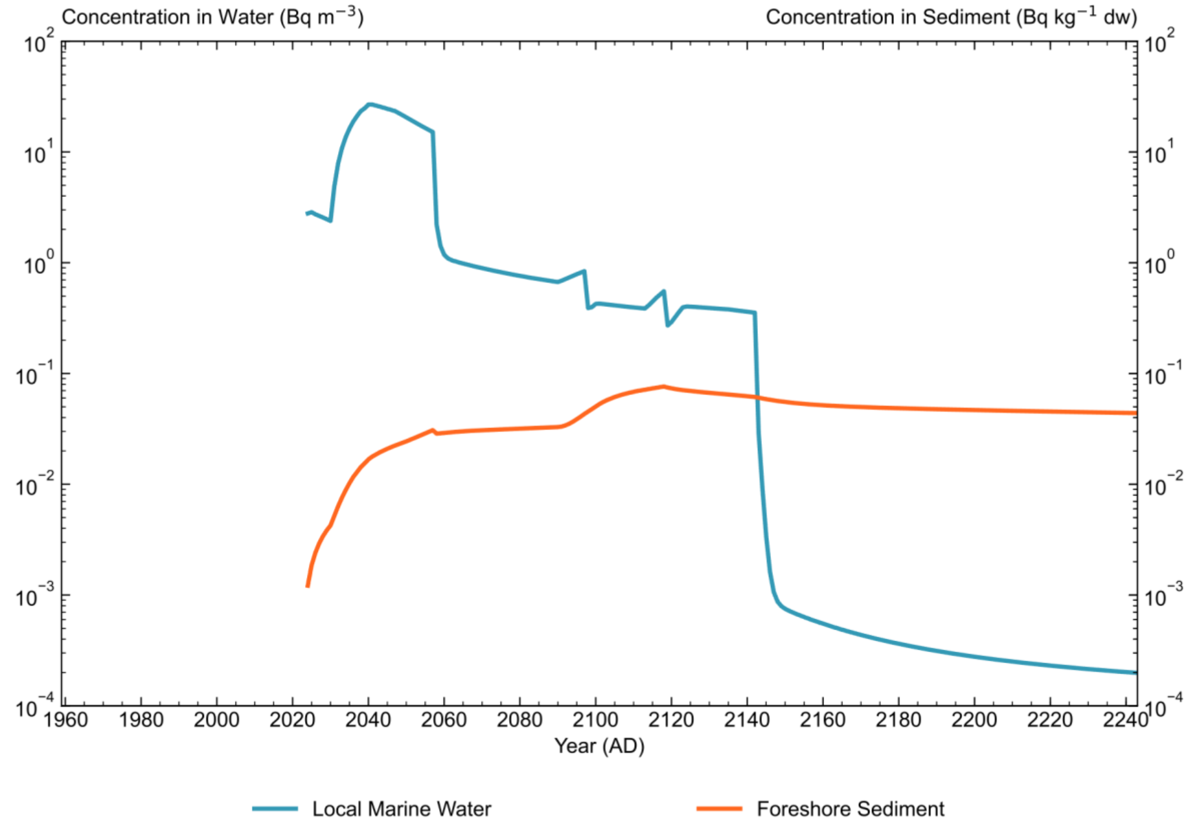


Figure 4.4: Modelled activity concentrations in marine water (B m^{-3}) and marine sediment (Bq kg^{-1} dw) arising from discharges from the marine pipeline (data from reference [44])

4.3.1.6 Historical Contamination of the Sediment in the Drigg Stream

Prior to commissioning the marine discharge pipeline in 1991, some leachate was discharged to the Drigg Stream. Residual contamination from those discharges remains evident, with concentrations in stream sediments being measured in 2009 [44]. We have used the measurements from 2009 to derive radionuclide concentrations in the stream water from 2023 AD onwards, as shown in Figure 4.5 [44]. The model takes radioactive decay into account but does not account for any loss of radionuclides by migration in water from the stream. This means that the modelled maximum concentration during the PoA is either at the start of the assessment 2023 AD for those radionuclides with simple decay, or parent radionuclides of a decay chain, or at the end of the PoA at 2242 AD.

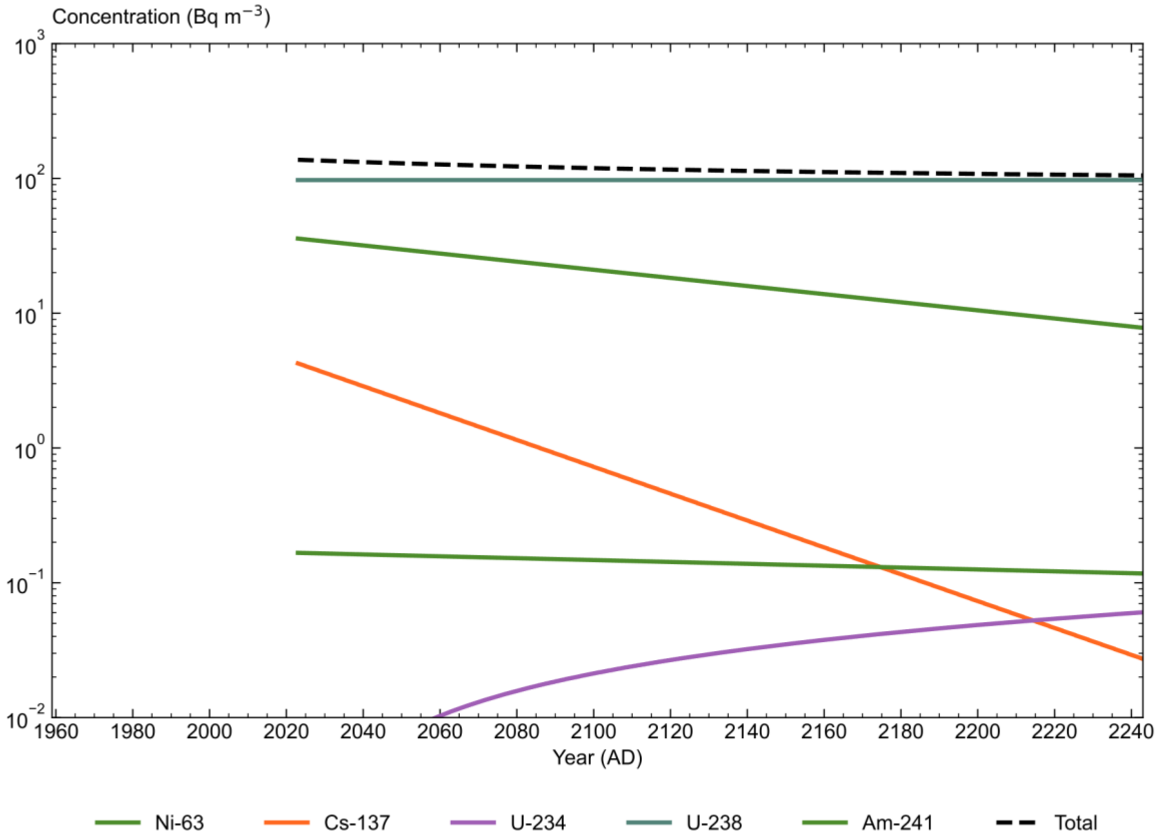


Figure 4.5: Modelled activity concentrations of radionuclides in Drigg Stream water (Bq m^{-3}) arising from historical contamination of the stream (data from reference [44])

4.3.1.7 Occasional Discharges to the Drigg Stream During Storms and Subsequent Flooding of Pasture

Since 1991, there have been no routine discharges to the Drigg Stream. However, discharge to the stream remains a permitted route. Discharges could be made to the stream if the site's effluent management system were overloaded during a storm. During the PoA, the leachate drainage system and pumps are designed to be able to handle a 1 in 50 year storm; however, if this capacity were exceeded excess water could be discharged to the Drigg Stream.

In this case, any activity discharged to the stream would likely be rapidly flushed out the stream by the stormwater. Therefore, we do not expect these discharges to add to contamination in the stream or lead to long-term exposure of organisms living in or near the stream. However, during a storm, the stream could flood nearby land. Radionuclides discharged to the stream during the storm could, therefore, be deposited on nearby land, leading to exposure of terrestrial biota. The PoA assessment [44] therefore assesses discharge to the stream during a storm and associated flooding of adjacent land.

The modelled concentrations of radionuclides in soil flooded following a storm are shown in Figure 4.6. No discharges after 2242 AD are modelled as the whole facility will then be capped and any further leachate arisings will be very low in volume [44].

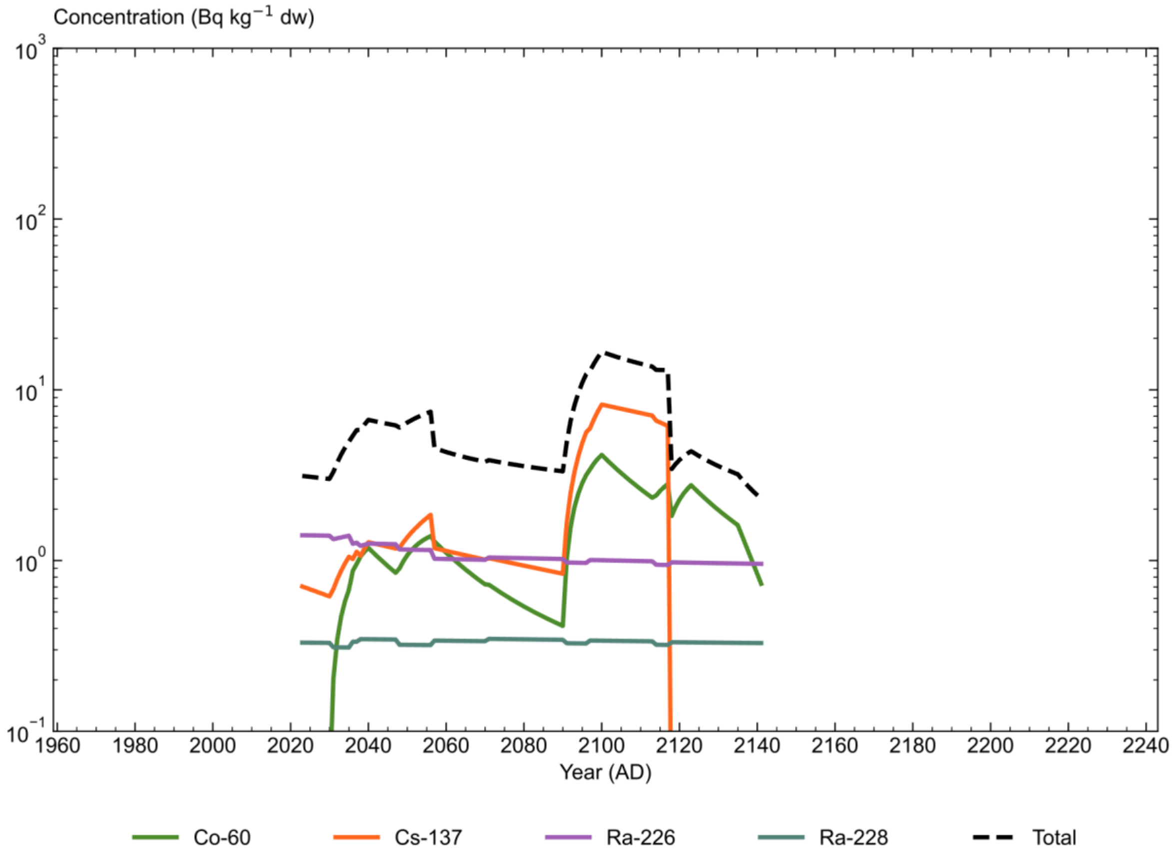


Figure 4.6: Modelled activity concentrations of radionuclides in flooded soil following a single storm in the year it occurs (Bq kg^{-1} dw) (data from reference [44])

4.3.1.8 Direct and Scattered Radiation from Uncovered Vaults

The PoA assessment calculates dose rates from direct irradiation and associated scattered radiation from open vaults (see Section 8 of reference [44]). These are calculated for the reference inventory, and they form the basis for considering potential dose rates to NHB (as described in reference [24]).

4.3.2 After the PoA

The exposure pathways applicable after the PoA are:

- migration of radionuclides in groundwater and:
  - discharge to the foreshore;
  - abstraction from a well placed between the LLWR and the coast and use of that well water for irrigation;

- release of C-14 and Rn-222 gases (the tritium inventory will have decayed to negligible amounts within the PoA);
- release of radioactivity resulting from coastal erosion of the repository in the long term;
- exposure to radioactivity because of inadvertent human intrusion into the repository.

These pathways are mapped to the principal locations in which the biota could be exposed to them in Table 4.2.

Table 4.2: Exposure pathways after the PoA and principal locations where biota could be exposed

Pathway by which biota may be exposed	Biota location and habitat
Migration in groundwater and natural discharge	Foreshore (marine)
	Local coastal area (marine)
	Lower Irt (freshwater and terrestrial)
Migration in groundwater, abstraction from a well and use to irrigate a garden	The irrigated land (terrestrial)
Coastal erosion of repository	Cliffs (terrestrial)
	Storm beach (terrestrial)
	Foreshore (marine)
	Local coastal area (marine)
	Particles on beach (marine and terrestrial)
Release of C-14 bearing gas	LLWR site (terrestrial)
Release of Rn-222 gas	LLWR site (terrestrial)
Material distributed after inadvertent human intrusion	Considered to be adequately bounded by impacts arising from coastal erosion

The following data covering the period after the PoA have been provided from the human dose assessments:

- Level 3 Groundwater Assessment [46]: radionuclide concentrations resulting from releases to groundwater in the foreshore, marine and estuarine environment for the reference case;
- Level 3 Gas Assessment [45]: concentrations of C-14 in vegetation following release as gas and concentrations of Rn-222 in air above trenches and vaults for the reference case;
- Level 3 Coastal Erosion Assessment [47]: radionuclide concentrations on the cliff, storm beach, foreshore and local coastal area for the reference case.

Exposure of NHB to the cliff face during erosion of the repository is representative of exposure to waste with very little or no dilution. it can be taken to bound exposure following inadvertent human intrusion and intrusion of biota to the waste. We discuss human and biota intrusion further in Subsection 4.3.2.7.

4.3.2.1 Migration of Radionuclides in Groundwater and Discharge to the Surface

Radionuclides released to groundwater from the facility will be transported towards the coast. Over time, radionuclides are discharged with groundwater through the sediments of the intertidal zone and enter the marine environment. The groundwater assessment [46] provides the resulting radionuclide concentrations in the foreshore, local coastal waters and Irt estuary. We use these concentrations for the NHB assessment.

4.3.2.2 Migration in Groundwater and Well Abstraction

A well could be sunk between the LLWR site and the coast. This could intercept contaminants from the LLWR that have migrated in groundwater towards the coast. We assess use of abstracted well water for domestic purposes, including irrigation of a garden. There is potential for NHB to be exposed as a result of contamination of garden soils and the soil concentrations resulting from this pathway from reference [46] provide the basis for the NHB assessment.

The modelled radionuclide concentrations in local coastal water, the foreshore (beach) sediment and lower Irt water arising from releases to groundwater after the PoA are shown in Figure 4.7. The results are all extremely low; results are also available for the salt marsh sediment (lower Irt bank sediments), but are even lower, so are not displayed on the graph.

The modelled radionuclide concentrations in irrigated topsoil are shown in Figure 4.8; these are similarly extremely small.

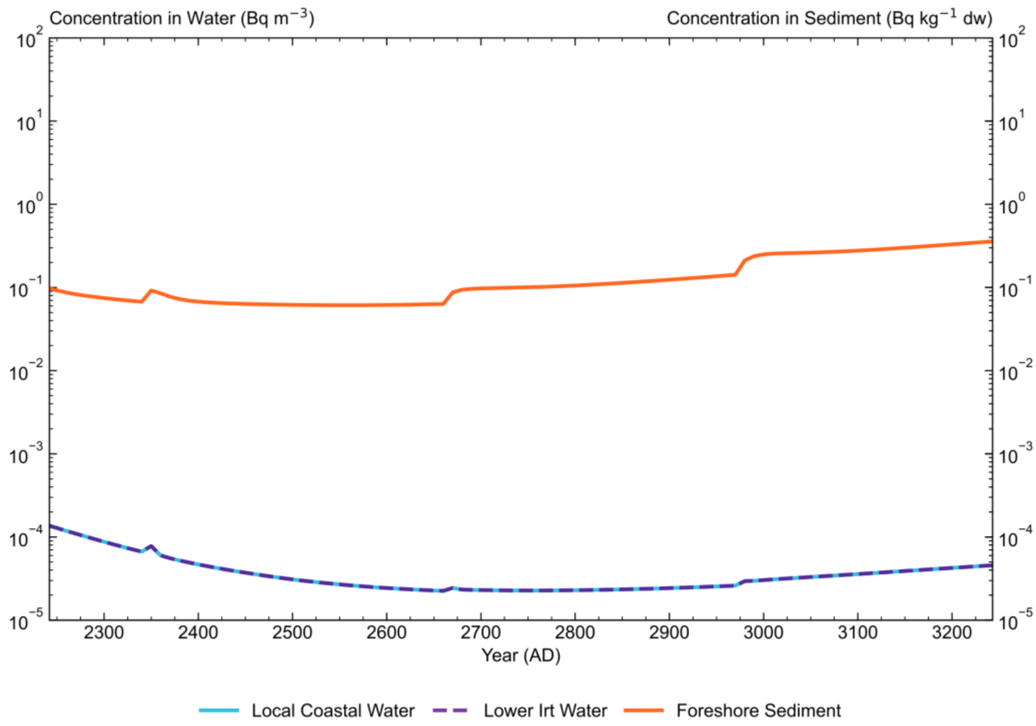


Figure 4.7: Modelled total radionuclide concentrations in local offshore water, lower Irt water and foreshore (beach) sediment from the groundwater pathway assessment; data from reference [46]

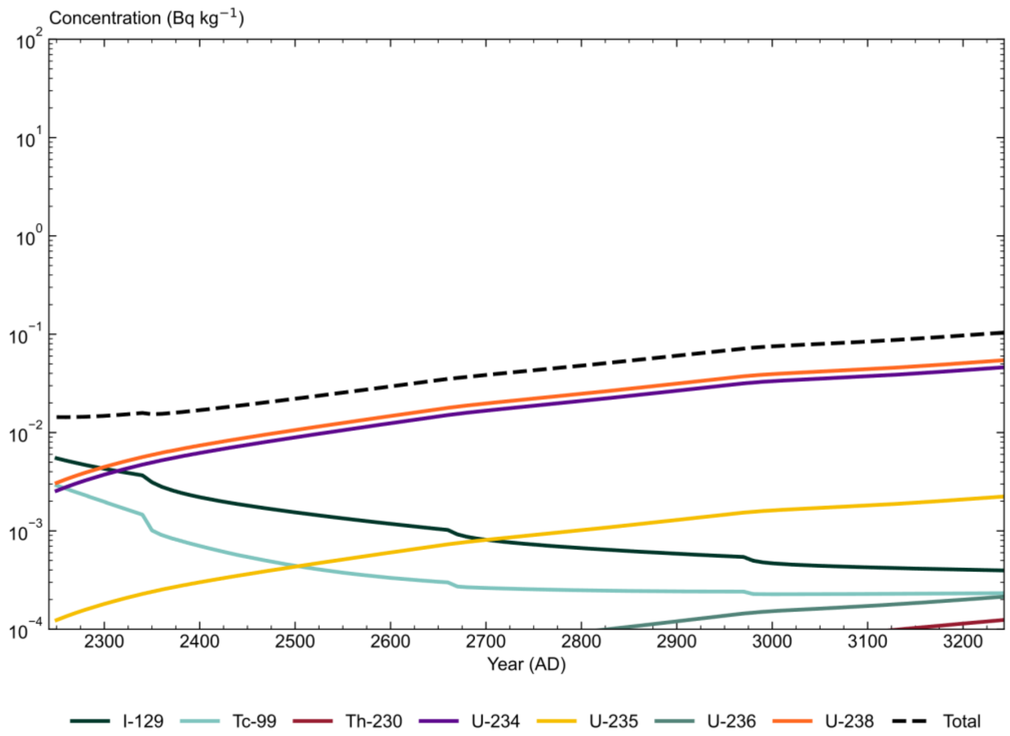


Figure 4.8: Modelled radionuclide concentrations in irrigated topsoil from the groundwater pathway assessment; data from reference [46]

4.3.2.3 Coastal Erosion

In our reference case assessment calculation for coastal erosion, erosion of the repository commences about 1250 years after present, with the vaults being eroded before the trenches. This takes the rate of coastal erosion and erosion of the 'hinterland' between the existing coastline and the repository into account. The model described in reference [47] provides radionuclide concentrations in the cliff, storm beach, foreshore and local coastal environment, which we use for the NHB assessment.

The focus of the coastal erosion assessment is a transect immediately in front of the repository, with some consideration of longshore drift in the foreshore region. The model uses a volume balance approach to evaluate the release and turnover of material from the cliff, through the storm beach to the foreshore, from the foreshore to the local offshore sediments, and then finally to the St Bees to Ravenglass sediment cell.

There are some model regions that are assumed to have no direct contact with the sea throughout the assessment, namely the cliff and the storm beach. Exposure of NHB in these model regions is considered via the terrestrial ecosystem model. Model regions regularly affected by tidal waters, or permanently submerged, are considered via the marine ecosystem model.

Figure 4.9 shows the total radionuclide concentrations (summed over all radionuclides) in environmental media through time, as calculated by the coastal erosion assessment model. In the reference coastal erosion case, the erosion of the site is assumed to begin close to 3270 AD. The vaults are eroded between around 3270 AD and 4130 AD, and the trenches are eroded between around 4130 AD and 5460 AD.

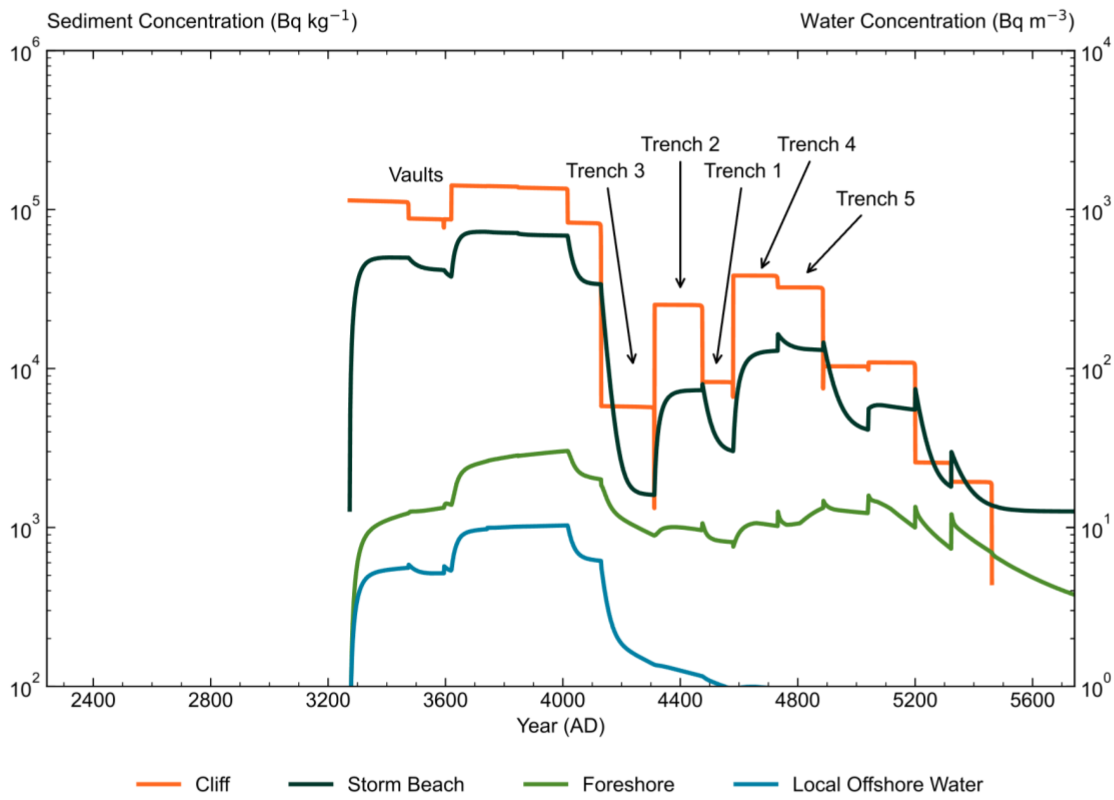


Figure 4.9: Modelled total radionuclide concentrations in solid material and sediments (Bq kg^{-1} dw) for the coastal erosion reference case. Figure reproduced from reference [47].

4.3.2.4 Release of C-14 Bearing Gases

C-14 bearing carbon dioxide and methane will be produced and migrate through the overlying fill and cap materials to enter the soil zone. The C-14 labelled methane can be converted to carbon dioxide and become available for uptake by plants via photosynthesis. As for the PoA (see Subsection 4.3.1.2), we assume that all methane is oxidised to carbon dioxide.

In the gas pathway assessment [45] we calculate the concentration of C-14 in vegetation over the trenches and vaults as a specific activity (Bq C-14 per kg stable carbon). The specific activity in animals cannot be higher than the specific activity in the plants that form the base of the food chain and could be lower, if the animals consume foodstuffs originating from outside the contaminated area. The assessment is made on the basis that all the organisms have the same specific activity as the vegetation, taking any fractional residency of the modelled area into account.

The modelled specific activities of C-14 in vegetation above the trenches and vaults are shown in Figure 4.10 and provide the basis for screening calculations for the radiological impact of gaseous C-14 on plants and animals. In these calculations the specific activity values are converted to a fresh mass basis by multiplying by the ratio of the carbon content to an organism's fresh mass.

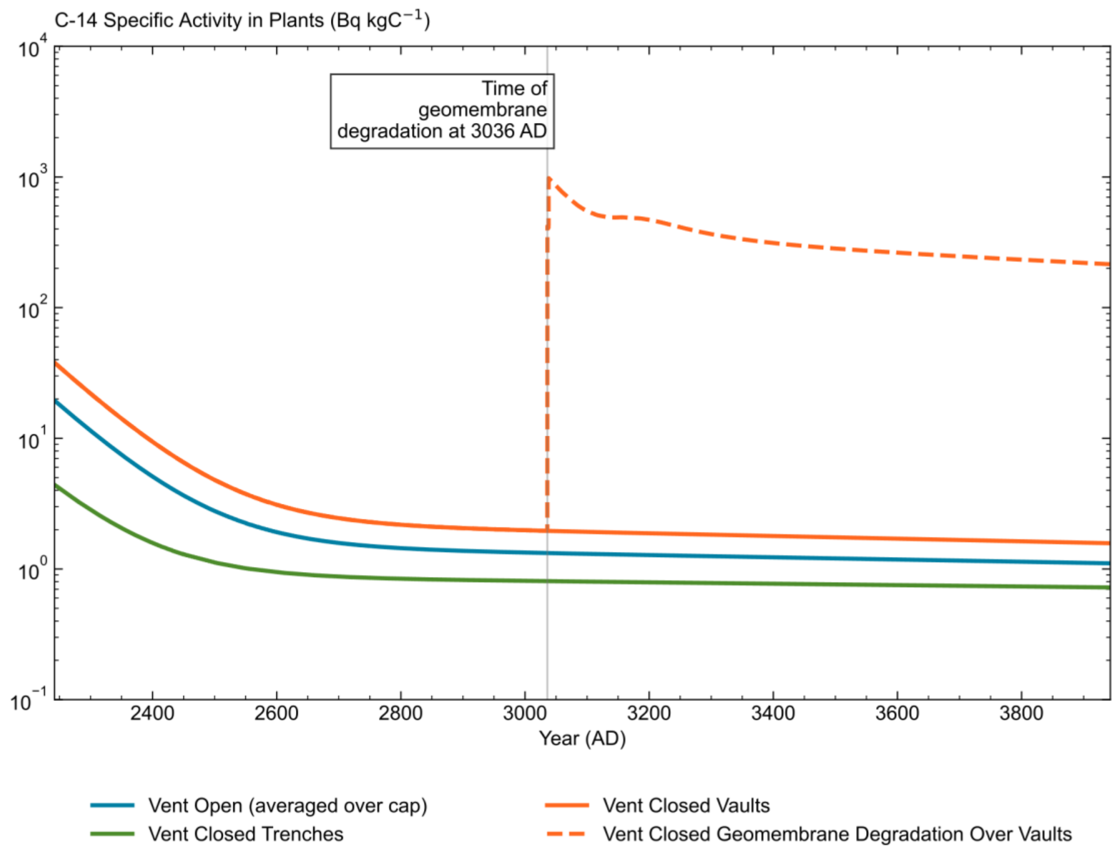


Figure 4.10: Modelled specific activity of C-14 in vegetation (Bq kg^{-1} C) above the trenches and vaults (data from reference [45])

4.3.2.5 Release of Rn-222

Figure 4.11 shows the evolution of the modelled concentrations of Rn-222 in the topsoil pore space and in the aboveground atmosphere after emplacement of the final cap [45]. The figure illustrates concentrations above the trench and vault caps for all times after the PoA. The engineering performance assessment (EPA) concludes that the earliest plausible timeframe for geomembrane degradation is around 500 years after Vault 8 capping [13]; Vault 8 capping is anticipated to be completed in 2136 AD.

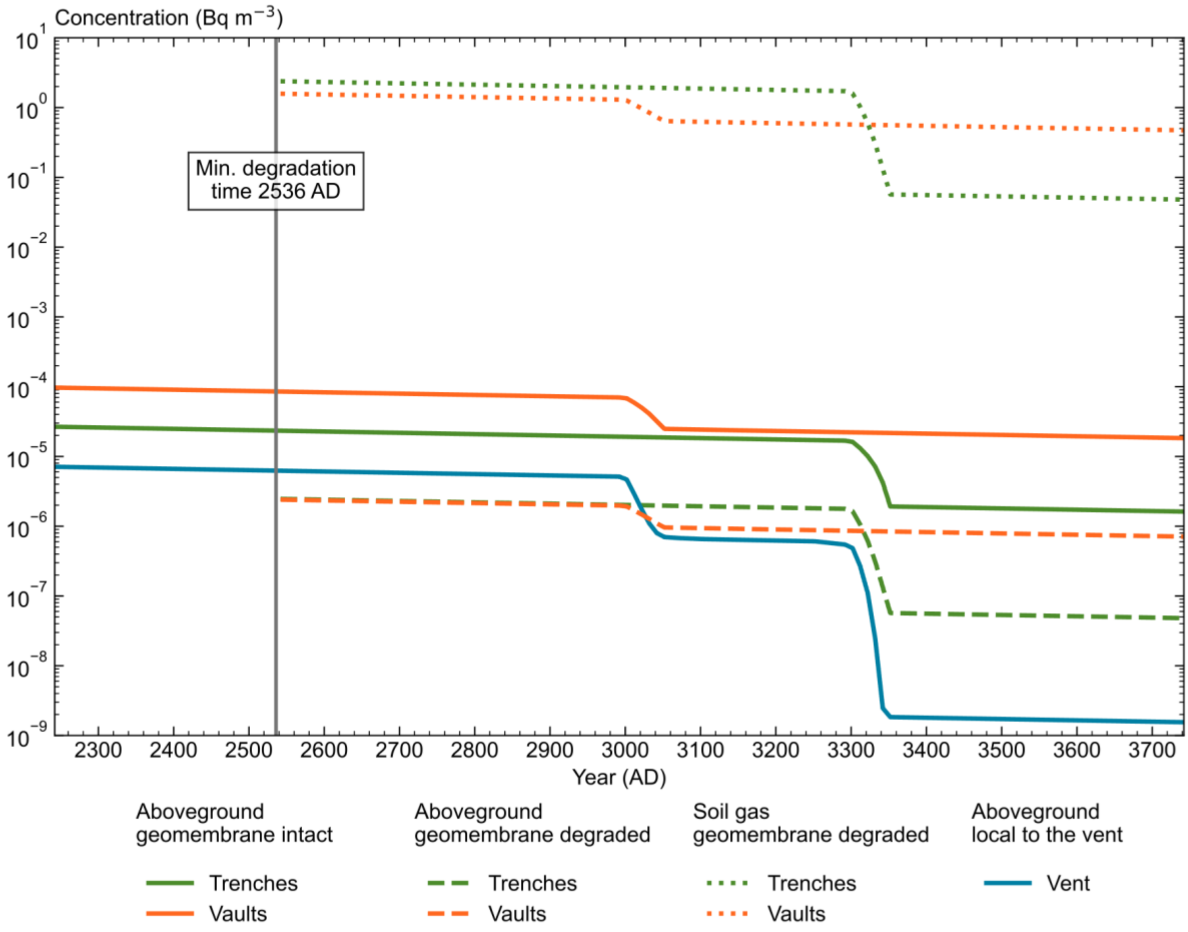


Figure 4.11: Modelled activity concentrations of Rn-222 in soil gas and aboveground atmosphere (Bq m^{-3}) in the gas pathway assessment (data from reference [45])

4.3.2.6 Exposure to Radioactive Particles

After the PoA, there is potential for radioactive particles to be released from the LLWR, because of either coastal erosion or human intrusion. If particles were released, they could vary in terms of the radionuclides present, the total activity of those radionuclides associated with a given particle, and the size and physicochemical composition of the particles.

We do not expect significant releases of particles from the LLWR. However, we have cautiously assessed the impacts of NHB encountering particles. We use information about the types of particles found on beaches at Dounreay, Sellafield and Dalgety Bay, and the types of material that have been disposed of at LLWR (e.g. low activity sources and mineral sands) to inform this assessment. A range of particles is considered, some of which are or could well be present in the LLWR and others that are much less likely to be present [48]. We cautiously assess on the basis that all the inventory of the relevant radionuclides at the LLWR could be released as particles. This assessment is extremely cautious. It does not represent an expected or likely situation, and we have no evidence to suggest that there will be large scale releases of such particles from the repository.

We then assess the impacts of this release using a calculation of the dose rates local to individual particles in individual organisms. We consider deposition of particles at locations within organisms that could affect fertility and fecundity of an individual organism. For particles to persist in the environment, they tend to be robust such that, if ingested, their radionuclide content would only be taken up from the gastrointestinal tract to a very limited degree. Therefore, we focus on the degree to which unabsorbed particles can irradiate radiosensitive cells when they are deposited on the external surface of an organism (e.g. caught within a shell) or pass through the gastrointestinal tract. Particles of higher radionuclide content are typically too large for intakes by inhalation to be significant.

4.3.2.7 Radiological Impacts to Biota Following Inadvertent Human Intrusion and Biota Intrusion

NHB may be exposed to contaminated material because of human intrusion events and because of the biota themselves intruding into the repository.

Reference [15] considers several possible human intrusion events. Each would result in different levels of mixing of extracted wastes with uncontaminated profile fill and cap materials, and in different sizes of area being affected. It is likely that if any human intrusion event did occur it would either affect only a small area (reducing both the possibility of impacting populations of NHB and the likelihood of sustained impacts to individual mobile organisms) or there would be dilution of the exposed material with surrounding uncontaminated materials (reducing dose rates). For most events considered, both these attributes are likely to apply.

This can be compared to exposure of biota during coastal erosion of the repository. Biota inhabiting the cliff face will be exposed to waste with no or very limited dilution for the entire time they spend on the cliffs. Therefore, assessment of exposure of biota arising from the cliff erosion pathway will be representative of worst-case exposure of biota to waste or contaminated material following human intrusion.

It is also possible that biota could intrude into the facility. In this case, it is likely that, the mechanism of intrusion or waste exposure would lead to some mixing of radioactive and inert material, leading to some dilution; that only a few members of a population, rather than a whole population, would be exposed; and that animals would source food from a wider area. Given this, we consider the coastal erosion pathway will also adequately represent exposure following intrusion. Moreover, the cap will include a bio-intrusion layer to reduce the likelihood of biota damaging the low-permeability layers of the cap; this will also reduce the likelihood of biota intruding into the waste [5]. Thus, the biota intrusion pathway is not discussed further in this report.

4.4 ERICA Tool

The ERICA Tool [23] implements the ERICA Integrated Approach [22].The Tool provides three tiers of assessment, described on the next page. We used a Tier 2 assessment.

- Tier 1 screens measured or modelled environmental media concentrations against pre-calculated concentrations that would give rise to a dose rate of 10 μGy h^{-1} to the reference organisms. It is intended as a cautious screening assessment.
- Tier 2 allows site-specific dose rates to be calculated and allows assessors to change the default assessment parameters, including CR, distribution coefficient (K_d) and occupancy factors. It also allows new representative organisms to be defined and more radionuclides to be assessed.
- Tier 3 is a probabilistic risk assessment in which uncertainties in the results may be determined. No screening dose rate is applied at Tier 3; rather, the results should be compared against available effects data to inform on the likely consequences of the calculated dose rates for the organisms of interest.

4.4.1 Use of the ERICA Tool

We used ERICA Tier 2 to calculate internal and external dose rates to each representative organism arising from the maximum concentration of each radionuclide. The approach to calculating these dose rates is given in Subsection 4.4.2; the dose rates account for the occupancy of an organism in different locations within an ecosystem (for example, time spent on the sediment and time spent in the water body. For each combination of organism and exposure pathway, we add the dose rates from each radionuclide. This gives the dose rate at maximum concentrations by each pathway for each organism. The resulting dose rates at maximum concentrations are compared with the 10 μGy h^{-1} screening value.

4.4.2 Internal and External Dose Rate Calculations

To calculate internal and external dose rates, each representative organism is represented as an ellipsoid or sphere with a given mass. The ellipsoid dimensions and mass are input into the ERICA Tool to calculate the internal and external DCs (see Table 3.2 to Table 3.5). The approach is described in detail in reference [24]. Each representative organism may include many different species or individuals from a range of sizes. Studies have shown that organism size is not a significant influence on dose assessments (e.g. references [49] and [50]). The representative organism approach is used by the ERICA Tool and the ICRP approach [23], [35].

There were four situations where the dose calculations differ from the standard ERICA approach. These were for C-14 and tritium which used a specific activity model, active particles and radon dose calculations. We describe our approaches to these in Subsections 4.4.2.2 to 4.4.2.4.

4.4.2.1 Handling of Decay Chains

ERICA uses dose coefficients from ICRP Publication 136 [30]. These include one year's worth of ingrowth of all daughter radionuclides (both short- and long-lived). The ingrowth and decay calculations consider radiological properties only; they do not account for physicochemical or biokinetic behaviours. For internal dose calculations, this is intended as a simple means of accounting for the dose contribution from daughters that ingrow in the organism tissue (the same phenomenon that is represented by the concept of committed internal dose to people). For external dose calculations, this is different to the assessments of impacts to people, where concentrations of and doses from non-secular daughters are calculated entirely separately from their parents.

The ERICA Tool provides a simplified approach for dealing with short-lived progeny in specific decay chains (these are listed in the ERICA help and include, for example, the decay chains of Ra-226, Ra-228, Th-228). ERICA treats these short-lived daughters as being in radiological secular equilibrium with their parents. This is like the assumption made in our assessments of radiological impacts to people, but there are some differences. The most important differences are:

- In ERICA, secular daughter transfer is governed by the properties of the daughter (daughter sorption coefficients and concentration ratios are used), whereas in the assessments of impacts to people, secular daughter transfer is governed by the properties of the parents.
- ERICA only considers radionuclides with a half-life of less than 10 days to be in secular equilibrium with their parents. This does not include all the radionuclides assumed to be in secular equilibrium in the assessments of impacts to people [51]. In this assessment, we have explicitly included these extra daughters in our assessment.

4.4.2.2 C-14 and Tritium Approach

A specific activity approach [14, 15] is used for the dose rate calculations for exposure to tritium and C-14 bearing gas in this assessment. This ensures that the NHB assessment for these two radionuclides, released in gas, is based on biosphere transport and behaviour assumptions consistent with those made in the human assessments. For these gaseous releases, consideration must be taken of the stable element concentration in environmental media and the representative organisms. These are used to derive a concentration in the representative organisms on a fresh weight (fw) basis that can be scaled by DRPUCs (for concentration in the organism, not in media) calculated using the ERICA Tool.

External exposure to tritium and C-14 is not included in the calculated dose rates within the ERICA Tool. As both radionuclides are low energy beta emitters, any external dose would be low in comparison to the internal dose rates because the beta emission cannot penetrate the outer (dead) layers of skin, fur etc.

In aquatic environments (including during coastal erosion), tritium and C-14 are assumed not to be in gaseous form and are modelled in the same way as other radionuclides. Exposure of terrestrial organisms to C-14 on the cliffs and storm beach is also not modelled, as this would not be in a chemical form that is readily taken up by biota. Because of the short half-life of tritium (around 12 years), we do not assess gaseous release of tritium after the end of the PoA.

4.4.2.3 Radon Approach

We use radon concentrations taken from the soil-gas and above-ground air concentrations calculated in the human assessments [14, 15] as an input into the ERICA tool to calculate external dose rates arising from immersion in air for all representative organisms except plants and the inhalation dose rates for mammals. Currently, there is insufficient understanding to extend the ERICA inhalation model to non-mammals. We perform some simple calculations based on the decay characteristics of Rn-222 to understand the likely magnitude of internal and external dose rates to organisms present in the soil, including uptake to tissue by diffusion.

4.4.2.4 Active Particles Approach

We have made cautious assumptions about the inventory that could be released from the LLWR as radioactive particles. Based on these assumptions, we assess the radiological impacts of exposure to UO_2 particles of various sizes, activities and prevalences [24]. The assessment is made on cautious assumptions; we do not expect this prevalence of particulate activity to be released from the repository.

We use these values as input into the NHB assessment for radioactive particles but consider for biota that there is:

- limited potential for inhalation of particles by NHB.
- potential for uptake into the gastrointestinal tract, although if any effect occurred, it would likely be a few microscopic lesions around trapped particles with no significant impact on the health of the individual organism.
- an effective barrier to entry of particles by the outermost surface of organisms which we also consider to be of limited or no radiosensitivity, as sensitive cells are found at depths of several tens of μm or more below the surface. i.e. beyond the range of alpha emissions from uranium isotopes incorporated in particles located on the surface.

We use the information on particle size to calculate absorbed dose rates for. These are then used to evaluate the implications of exposure of representative organisms to radioactive particles. We use the information on particle prevalence to consider the likelihood of an organism encountering a particle.

4.5 Treatment of Climate Change

We have selected three greenhouse-gas emissions scenarios for assessing the impacts of climate change in the context of the 2026 ESC [8]. These comprise a reference scenario in which carbon dioxide emissions remain around current levels until the middle of the century before decreasing, a low emissions scenario in which carbon dioxide emissions decline to net zero after 2050 followed by net negative emissions, and a high emissions scenario involving carbon dioxide emissions that roughly double from current levels by 2050 and peak at three times current levels around 2090.

Changes in mean annual temperature at the LLWR site peak at about 1 °C above their 1981-2010 average [8] in the low emissions scenario, about 2.4 °C above that average in the reference scenario and about 7.4 °C above in the high emissions scenario. The corresponding increases in mean annual precipitation are about 2%, 5% and 16%, respectively. Together, these changes in temperature and precipitation will result in peak increases in potential evapotranspiration (which influences the exposure of plants to environmental radioactivity) of 7%, 17% and 52% in the low, reference and high scenarios, respectively. The peak changes will occur at around 100 to 300 years After Present (AP).

Increased warming will tend to result in increasingly dry summers and wetter winters. Based on these projected changes, it is concluded that, for both the low and reference scenarios, the LLWR site is expected to retain its temperate oceanic humid climate. However, for the high-emissions scenario, the site is expected to experience subtropical humid conditions between 300 and beyond 3,000 years AP, but to be temperate oceanic humid at earlier and later times. The subtropical conditions are projected to be like those of present-day Portugal. Importantly, arid conditions leading to desertification are not expected to occur.

There is no reason to expect that climate change will lead to a radically different vegetational regime from that at the present day. It is likely, therefore, that the overall characteristics of the ecosystems present and the organisms in those ecosystems will be like the present day. We assess representative organisms; each covers a range of the species present on or near the site. The representative organisms have a simple, stylised geometry and key habits specified. We do not expect new species that fall outside the range of characteristics covered by our representative organisms to reside near the site in future. Minor changes in species and habits (for example, a change in migration patterns or different species occupying the same ecological niche) will still be covered by our selected representative organisms. On this basis, we adopt the same organism types for assessing impacts to non-human biota as are adopted for present-day conditions.

A far more radical change to the local landscape and associated habitats would be due to changes in sea level and associated effects on coastal erosion, with rapid retreat on the existing cliff line. In the next few hundred years, sea level is projected to rise by about 1, 3 and 10 m in the low, reference and high scenarios, respectively. By 3,000 years AP, the corresponding increases in sea level are projected to be about 1, 8 and 23 m, respectively. The effects of these increases in sea level have been addressed in the coastal erosion modelling. However, although the relevant habitats will be spatially displaced inland of the current coastline, there is no reason to suppose that they will be substantially altered in their overall characteristics. Therefore, again, we consider it appropriate to adopt the same organism types for assessing impacts to non-human biota as are adopted for present-day conditions.

4.6 Treatment of Uncertainty

Our '*Assessments Manual*' [52] sets out how we address uncertainty in ESC assessments. It requires that the approach to treating uncertainty in each assessment be planned and documented. This subsection sets out how we address uncertainty in this assessment, including how we address the requirements of the '*Assessments Manual*'.

The assessment manual gives several options for addressing uncertainty. We use a qualitative evaluation. The other approaches would not be appropriate or proportionate for this assessment. They would require significantly more underpinning data (for example, a probabilistic assessment would require probability density functions for the parameters being investigated) and they may not be appropriate for some of the important uncertainties in this assessment (such as the uncertainty arising from the use of generic transfer data and the use analogues where data are not available).

We set out the uncertainties in this assessment in reference [24] and we have audited the biases in the assessment to understand how the uncertainties and assumptions affect the results [24]. As part of this bias audit, we decided which uncertainties and biases needed to be accounted for in the assessment to reduce their effect.

This assessment uses concentrations of radionuclides in the environment calculated in the assessments of radiological impacts to people (and, for some of the assessment of impacts during the PoA, concentrations of radionuclides measured in the environment). Each of those assessments has taken account of uncertainty and has its own bias audit in line with the process set out in our '*Assessments Manual*' [52]. The details of how uncertainties have been treated in each assessment and their effects on the assessment results are given in the relevant Level 3 report (e.g. references [44], [45], [46], [47]). Any uncertainties that affect the calculation of radionuclide concentrations in those assessments will apply to this assessment because it uses the concentrations. Uncertainties in those assessments that do not affect the calculation of radionuclide concentrations (for example, uncertainties in the habits of receptors) will not apply to this assessment.

The ERICA approach to modelling contaminant uptake by biota, which we use in this assessment, has been validated using measurements taken near the LLWR site. Activity concentrations in media and biota near the site were measured. The media concentrations were then used as inputs to ERICA and the concentrations in biota calculated compared to those measured [43].

4.6.1 Approach to Treating Uncertainty

We use simple, stylised models, both for the ecosystems being assessed and for the organisms in those ecosystems. The models used represent a broad range of potential ecosystems and organisms and, thus, account for potential future variations in the ecosystem and organisms. Transfer of radionuclides from the environment to the organisms is modelled using concentration ratios. These are based on empirical data. The assessment does not include detailed representations of the physiology of individual organisms nor their habits (for example, food chains, which are instead accounted for in the empirical concentration ratios). This means that the assessment and its conclusions are applicable to the range of ecosystems that may be expected on and near the LLWR site for the duration of the assessment.

Where data used in this assessment are also used elsewhere in the ESC, we take the data from our data management system [53]. The data in our data management system have been chosen to be appropriate for the LLWR and its environs and the uncertainties in the data are recorded. We have specified organism properties based on those species present on and near the LLWR site. We take other data from literature sources (including the database incorporated in the ERICA tool) and, where there are data gaps, we use the gap-filling method recommended in the ERICA approach.

We assess the expected evolution of the LLWR site and its environment, using the reference calculation cases from the assessments of radiological impacts to people. We do not consider the results of the alternative calculation cases (Subsection 4.6.2).

A key uncertainty in site and environmental evolution that will influence dose rates to non-human biota is uncertainty in climate change (Subsections 2.3 and 4.5). This may affect the species that are present and their habits. We address this uncertainty by assessing dose rates to representative organisms. We do not expect new species that become present or habits that change because of climate change to fall outside the range of characteristics covered by our representative organisms.

Uncertainty in climate change also contributes to uncertainty in the rate and mode of the erosion of the LLWR site by the sea. The main influence this would have on dose rates to non-human biota is by changing the amount of decay and ingrowth of radionuclides that has occurred in the repository at the time at which biota are exposed by the coastal erosion pathway. The reference case reflects our best understanding of the likely timescale of erosion of the facility. Therefore, and for the reasons given in Subsection 4.6.2, we only assess impacts to biota following coastal erosion for the reference case.

4.6.2 Alternative Calculation Cases

Our assessments of radiological impacts to people use alternative calculation cases to investigate some uncertainties. These cases allow us to understand how the behaviour of the system and the results of the assessments are affected by the uncertainties considered. We can then determine whether the uncertainty being considered would challenge the arguments made in the ESC.

The alternative calculation cases considered in the assessments of radiological impacts to people relate to uncertainties about phenomena that are represented in those assessments or in data that are used in those assessments. They do not relate to phenomena directly represented in nor data directly used in this assessment of NHB. Applying these alternative calculation cases to this assessment would not, therefore, add significantly to our understanding of or confidence in the results of the NHB assessment. The consideration of these uncertainties in the assessments of radiological impacts to people is sufficient to show that they do not challenge our arguments made in the ESC for NHB. Therefore, this NHB assessment uses only concentrations of radionuclides in the environment from the reference case calculations of the assessments of radiological impacts to people.

4.6.3 Important Uncertainties

In the following list we describe the most important uncertainties in our NHB assessment and explain the effects they have on the assessment results and how we account for them.

1) **Uncertainty in the inventory:** In this assessment, we consider a reference inventory, but the inventory that is actually disposed of will be different. We ensure that radiological impacts to people will be within regulatory criteria through the waste acceptance controls that we derive from our assessments of impacts. However, we do not do this for NHB. We explain why we do not do this and how we ensure that NHB will be protected in Subsection 4.2.
2) **Timing of maximum radionuclide concentrations:** The initial screening of biota dose rates is done based on the sum of the maximum dose rate of each radionuclide assessed. These maximum concentrations occur at different times; in reality, no organism will be exposed to the maximum dose rate for every radionuclide. Therefore, using the sum of maximum dose rates will overestimate the dose rate. Where dose rates exceed the screening value, the timing of the maximum concentrations and the time-dependence of the sum of the dose rates is one of the aspects of the assessment we consider in more detail, reducing this uncertainty in those cases.
3) **Uncertainties in the species present and their habits:** The species present on and near the LLWR site may change over time or their habits may change (especially because of climate change). We assess representative organisms. These are organisms defined for this assessment (Subsection 3.2), and each covers a range of the species present on or near the site. The representative organisms have a simple, stylised geometry and key habits (such as occupancies and residencies, see Subsection 3.2) specified. We do not expect new species that fall outside the range of characteristics covered by our representative organisms to reside near the site in future. Minor changes in species and habits (for example, a change in migration patterns or different species occupying the same ecological niche) will still be covered by our selected representative organisms. Therefore, our assessment accounts for uncertainty in the species present and those that may be present in the future and their habits.
4) **Uncertainty in whether organisms will be present in the locations with the greatest concentrations of contaminants:** The contaminant transport models used in our assessments are based on our understanding of how contaminants may enter and migrate in the environment. We assess dose rates to organisms assuming that they are present in locations affected by contaminant release. If organisms did not spend time in these locations, the assessed dose rates would be lower.
5) **Uncertainty in the input data used for the assessment:** This assessment uses input data from a range of sources, including the assessments of radiological impacts to people, our data management system, publicly available databases and the

scientific literature. These data have different levels of uncertainty and different means by and degrees to which the uncertainty has been investigated in the original source. There are some data gaps, which we fill using an analogue approach. Where we take data from our data management system, we use best-estimate values. Where we take data from the assessments of radiological impacts to people, we use our reference calculation case data (Subsection 4.6.2). Where we take data from publicly available databases or literature, we seek to use realistic data that are applicable to the conditions at or near the LLWR site.

6) **Uncertainty in the modelling approach in ERICA**: The ERICA Tool uses a simple but pragmatic approach including equilibrium CRs, K_ds and dose coefficients derived for simple organism geometries (e.g. ellipsoids). These simplifications introduce some uncertainty into the assessment. This approach is consistent with the level of scientific understanding of radiation impacts to NHB. It is consistent with recommendations and guidance from the Environment Agency and guidance from international organisations such as the IAEA and the ICRP. Some of these simplifications have been studied. For example, organism geometry has been shown to have a small influence on assessed dose rates [49, 50]. The transfer modelling in the ERICA Tool has been compared with measured data from the Drigg Dunes [43]. Where the screening dose rate is exceeded, we consider whether any of these simplifications have implications for the assessed radiological impacts.

7) **Uncertainty in the radiological impact of the assessed dose rates:** To assess radiological impacts to people, effective dose is used. Effective dose relates exposure and the potential for induction of cancer and is underpinned by scientific understanding of human physiology and radiobiology. Because there is less such understanding for non-human organisms and because the protection goals are different, doses to non-human organisms are assessed using absorbed dose rates. However, the same absorbed dose rate may have different magnitudes of impact for different species. In this assessment, the absorbed dose rates calculated use the ERICA-recommended weighting values to account for the relative biological effectiveness of different types of ionising radiation. We then compare these assessed absorbed dose rates to a screening value. The screening value was calculated from observed effects data and is low enough to ensure protection for all species. Where the screening value is exceeded, we use the ICRP DCRLs and literature data about species radiosensitivity to understand the impacts of the assessed dose rate on relevant species.

8) **Uncertainties in assessing the exposure of populations:** The protection endpoint specified for non-human biota in the GRA and GRR is to protect populations of these species. There are uncertainties in how a population is defined and in the uniformity of exposure of individual organisms from a population. We calculate dose rates to an individual exposed to the highest concentrations of radionuclides expected in the environment. The dose-rate screening value used and the ICRP DCRLs are specified

assuming that the whole population is exposed to the assessed dose rate. Using these with our assessed dose rates ensures that this assessment will not underestimate impacts to populations.

4.6.4 Further Work to Reduce Uncertainties

We have been able to accommodate all of the major uncertainties identified, as briefly described in Subsection 4.6.3 (for example, by use of the representative organism approach or by the use of detailed considerations, such as organism radiosensitivity, when needed). As such, no additional work is required to further reduce the uncertainties in this assessment.

Our ESC is subject to ongoing review and development, and we have a programme of formal updates (such as periodic reviews and major reviews). Through these, we incorporate developments in topics such as inventory, site understanding and underpinning understanding into the ESC, especially where there are significant changes. This ensures our ESC remains technically accurate and up-to-date and meets the requirements of the GRA and our permit to keep the ESC up to date. The assessment of radiological impacts to non-human biota is part of these updates.

There is ongoing development of international guidance on NHB assessment and underpinning scientific understanding. For example, ICRP Task Group 99 is revising guidance on the DCRLs and the IAEA are updating Safety Report Series No. 19. When such updated guidance is issued, the implications it has for this assessment will be considered as part of our ongoing review and development programme.

4.7 Developments Since 2011

Several important developments have taken place since the 2011 ESC NHB assessment. These include changes in response to recommendations made by the Environment Agency as part of their review of the 2011 ESC; our '*Addressing Regulatory Requirements and Feedback*' [18] report explains how we have addressed these recommendations.

Developments in our NHB assessment include the following.

- This assessment uses environmental concentrations from the latest contaminant transport models and monitoring results that are used for assessments of radiological impacts to people.
- We have included of all radionuclides considered for the human dose assessments for NHB (except for Am-242m that ERICA does not model).
- This assessment has more detailed consideration of the time of maximum concentrations in the environment.
- This assessment used the latest version of the ERICA tool (v2.0). Since the 2011 ESC, the ERICA Tool has been updated. The most significant updates are to the underpinning databases for transfer (concentration ratios) and to the dosimetry; the

dosimetry is now based on the time-integral activities ratio approach from ICRP Publication 136 [30].

- This assessment has a more detailed mapping of the species present in the environs of the LLWR to the representative organisms used in the assessment and systematic consideration of occupancy and residence times of these representative organisms.
- We have evaluated the potential for harm to NHB through exposure to high-activity particles.
- This assessment has a more explicit evaluation of the uncertainties in the assessment outputs (described in reference [24], which includes a bias audit, and Subsection 4.6). This evaluation follows the approach set out in our '*Assessments Manual'* [52].

The assessment of radiological impacts to NHB in the 2011 ESC assessed three periods comprising the present day, the remainder of the PoA and the post-PoA period [39]. The assessment at the present day used monitoring data and results published in the literature. It focused on the Drigg Coast SSSI and on the Drigg Stream, the latter being taken as indicative of concentrations of radionuclides local to the LLWR. The Drigg Coast SSSI was covered in studies of the Drigg Dunes [39]. The remainder of the PoA was not specifically studied, it being argued that controls of environmental radioactivity due to releases of leachates from the site will be at least as rigorous as those applied to date. Thus, environmental impacts due to release of leachates were projected to decrease over this period.

For this ESC, the PoA assessment models discharges and external dose rates for the assessments of impacts to people. We have used the results of this modelling for this assessment of impacts to NHB and consider the PoA as a single period, rather than distinguishing between the present day and the remainder of the PoA. We also assess the impacts on NHB of gaseous releases of tritium and Rn-222 during the PoA; these were not assessed in the 2011 ESC.

5 Assessment of Radiological Impacts During the PoA

This section gives the assessed dose rates to non-human biota during the PoA for each pathway. It then discusses the potential impacts of these dose rates.

5.1 Aqueous Discharges

5.1.1 Marine Discharges

The modelled total dose rates at maximum concentrations for the representative organisms exposed to contamination following discharges via the marine pipeline are shown in Figure 5.1. In all model regions, total dose rates at maximum concentrations are below the screening value. The highest assessed total dose rate at maximum concentrations is for phytoplankton (0.038 μGy h^{-1}). For all other representative organisms, total dose rates at maximum concentrations are typically at least three orders of magnitude lower than the screening value.

As the cliff model region is only of relevance for the coastal erosion pathway, there is no cliff considered during the PoA. This means that the only route of exposure for the cliff-nesting seabirds during the PoA is their time spent in contaminated marine ecosystems.

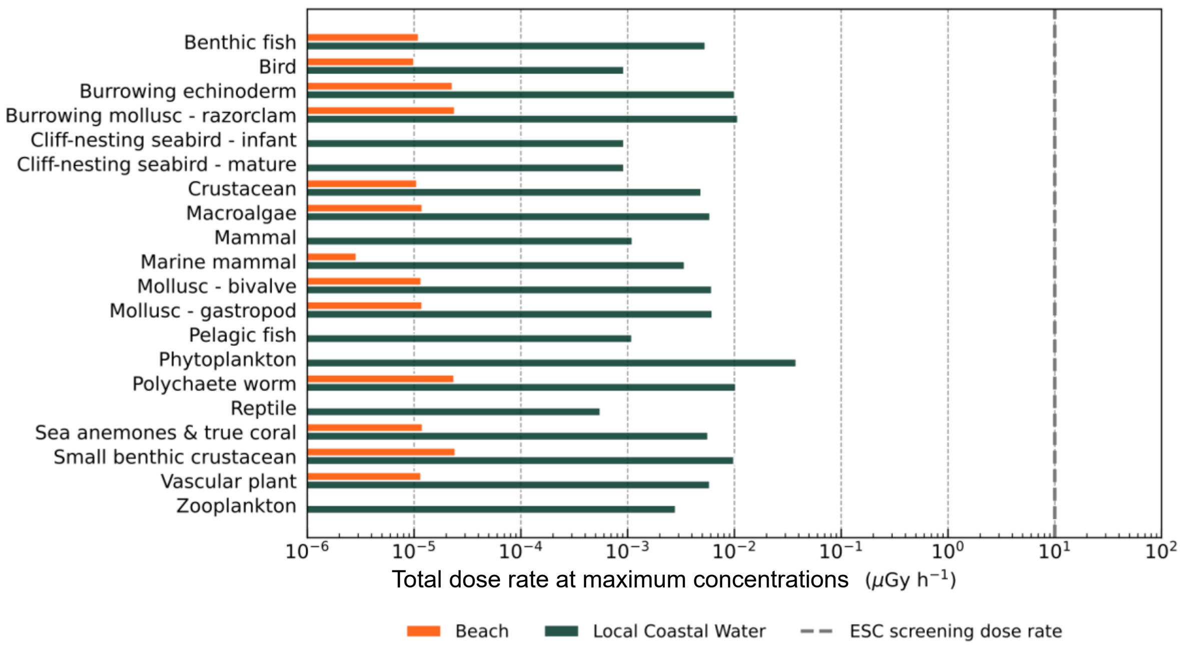


Figure 5.1: Modelled total dose rates at maximum concentrations for marine representative organisms resulting from discharges from the marine pipeline during the PoA (μGy h^{-1})

After the calculations underpinning the ESC were done, an error was identified in the modelling of leachate discharges during the PoA (see Subsection 4.2.2 of reference [14]).

Correcting this error would increase leachate concentrations by up to a factor of about two. Dose rates to organisms in the marine environment are more than two orders of magnitude below the screening dose rate. Therefore, they would still be below the screening dose rate if the error were corrected.

5.1.2 Flooded Soil Following a Storm

The modelled total dose rates at maximum concentrations associated with terrestrial representative organisms occupying a field affected by flooding with contaminated water following a storm are shown in Figure 5.2. The total dose rates at maximum concentrations are all more than an order of magnitude below the screening value, with lichen and bryophytes being the most exposed with an assessed total dose rate at maximum concentrations of 0.18 μGy h^{-1}.

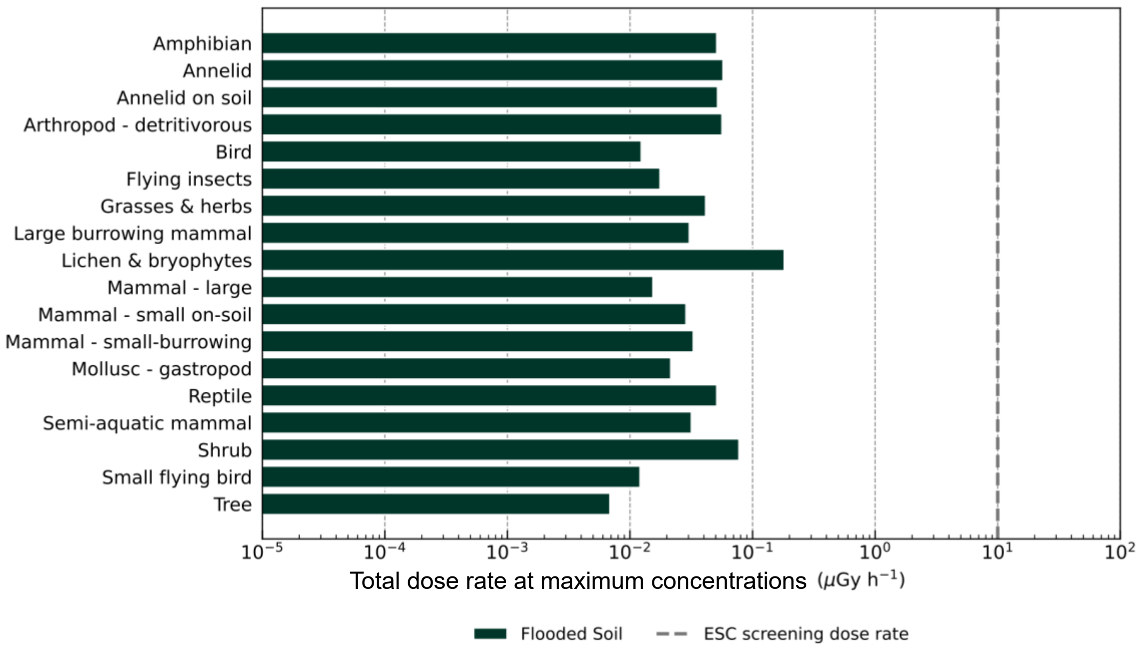


Figure 5.2: Modelled total dose rates at maximum concentrations to terrestrial representative organisms from exposure following a storm that results in flooding during the PoA (μGy h^{-1})

After the calculations underpinning the ESC were done, an error was identified in the modelling of leachate discharges during the PoA (see Subsection 4.2.2 of reference [14]). Correcting this error would increase leachate concentrations by up to a factor of about two. Dose rates to organisms after discharge to the stream are over an order of magnitude below the screening dose rate. Therefore, they would still be below the screening dose rate if the error were corrected.

5.2 Historical Contamination of Stream

The modelled total dose rates at maximum concentrations for the representative organisms exposed to historical contamination in the stream are shown in Figure 5.3. All dose rates are below the screening value with insect larvae being the most exposed with a total dose rate at maximum concentrations of 4.9 μGy h^{-1}.

For all representative organisms, the modelled total dose rates at maximum concentrations are dominated by exposure to U-238 (Figure 5.3), which was used to represent the other alpha activity measured in the stream sediment in the PoA assessment [44]. Other significant contributors to the total dose rates at maximum concentrations are Am-241 and Cs-137, with the three radionuclides contributing at least 99.8% of the total dose rate at maximum concentrations for all freshwater biota. The maximum concentrations of U-238, Am-241 and Cs-137 occur at the start of the assessment (2023 AD).

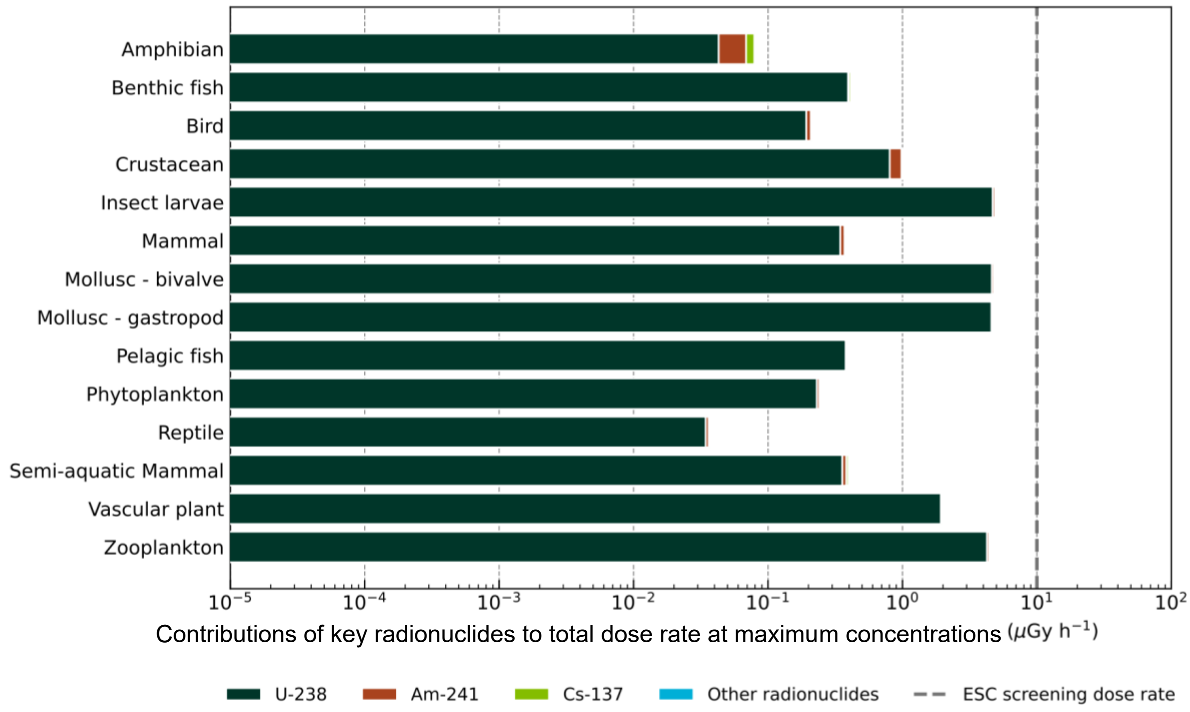


Figure 5.3: Contributions to total dose rates at maximum concentrations for freshwater representative organisms from exposure to historical contamination in Drigg Stream (μGy h^{-1})

5.3 Organisms Residing in More than One Ecosystem

The semi-aquatic mammal is assumed to spend 50% of its time residing on land, and 50% in the freshwater ecosystem. It will receive external dose from its time in both ecosystems but will receive internal dose only from the freshwater ecosystem where it forages for food. Therefore, when calculating internal dose for the semi-aquatic mammal it is assumed to have 100% residency in the Drigg Stream.

Table 5.1 shows the modelled total dose rate at maximum concentrations associated with the semi-aquatic mammal exposed to aqueous discharges following flooding (terrestrial environment) and historical contamination in the Drigg Stream (freshwater environment) during the PoA. The total dose rate at maximum concentration (0.39 µGy h^{-1}) is more than an order of magnitude below the screening value.

Table 5.1: Modelled total dose rates at maximum concentrations for semi-aquatic mammal exposed to aquatic discharges during the PoA (µGy h^{-1})

Exposure type	Source of exposure	Maximum total dose rate (µGy h^{-1})
External	Flooded soil (50% residency)	3.0 10^{-3}
External	Drigg Stream (50% residency)	1.1 10^{-2}
Internal	Drigg Stream (100% residency)	3.7 10^{-1}
Total		3.9 10^{-1}

5.4 Dose Rates from Gaseous Releases

The modelled total dose rates at maximum concentration to terrestrial representative organisms during the PoA following exposure to tritium are shown in Figure 5.4. The semi-aquatic mammal is included in this assessment. This is cautious because the specific-activity model assesses ingestion of tritium containing foodstuffs in the terrestrial ecosystem, whereas this representative organism would obtain its food from the freshwater ecosystem (which would not have much less or no tritium contamination).

The modelled total dose rates at maximum concentration to most of the terrestrial representative organisms during the PoA following exposure to Rn-222 are shown in Figure 5.5. These are external dose rates for all except the mammalian species. The mammalian dose rates also account for internal exposure. Due to uncertainties in the application of radon dosimetry to plants, immersion dose rates for grasses and herbs, shrubs and trees have not been calculated [54].

The dose rates at maximum concentrations associated with tritium and Rn-222 exposure are more than three and two orders of magnitude below the screening value respectively. With respect to Rn-222, the small flying mammal is the most exposed with an assessed total dose rate at maximum concentrations of 5.4 10^{-2} µGy h^{-1}.

The modelled total dose rates to terrestrial representative organisms resulting from maximum gaseous C-14 releases from the trenches or vaults are shown in Figure 5.6 for the PoA. All the dose rates associated with maximum C-14 exposure in gaseous form are more than an order of magnitude below the screening value.

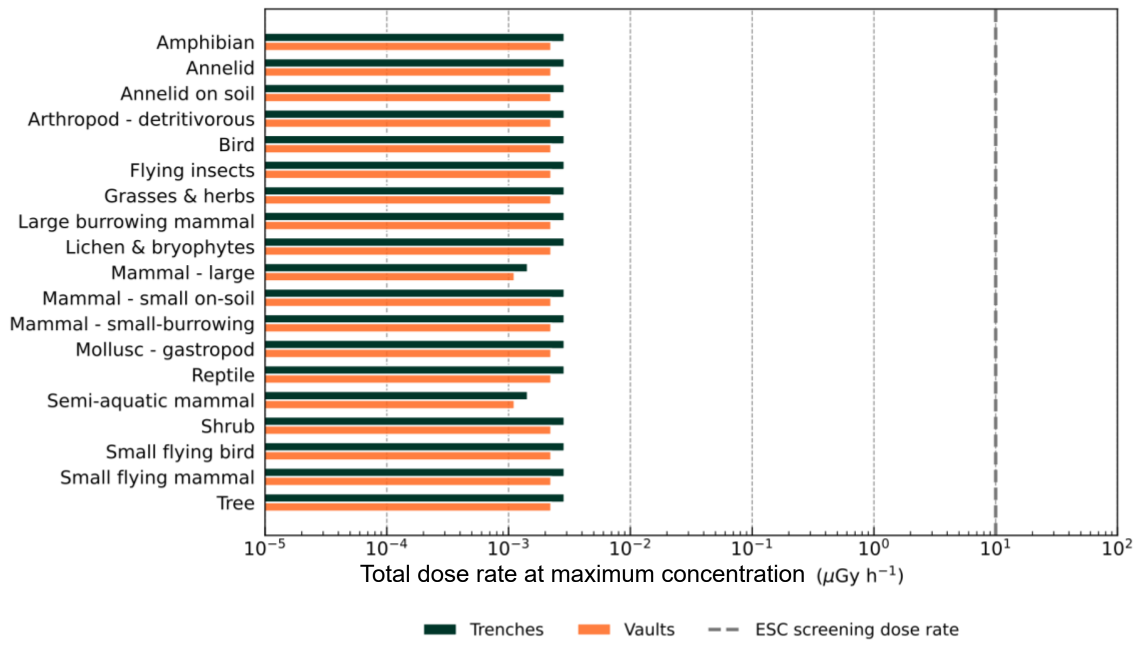


Figure 5.4; Modelled total dose rates at maximum concentration to terrestrial representative organisms from atmospheric releases of tritium during the PoA (µGy h^{-1})

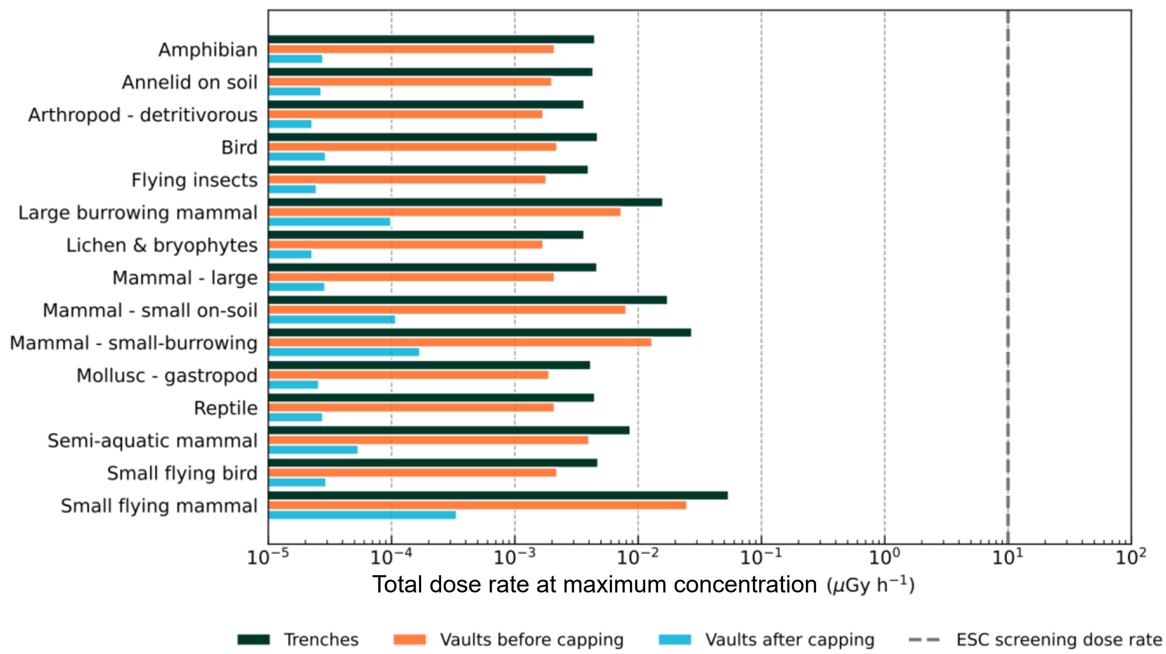


Figure 5.5; Modelled total dose rates at maximum concentration to terrestrial representative organisms from atmospheric releases of Rn-222 during the PoA (µGy h^{-1})

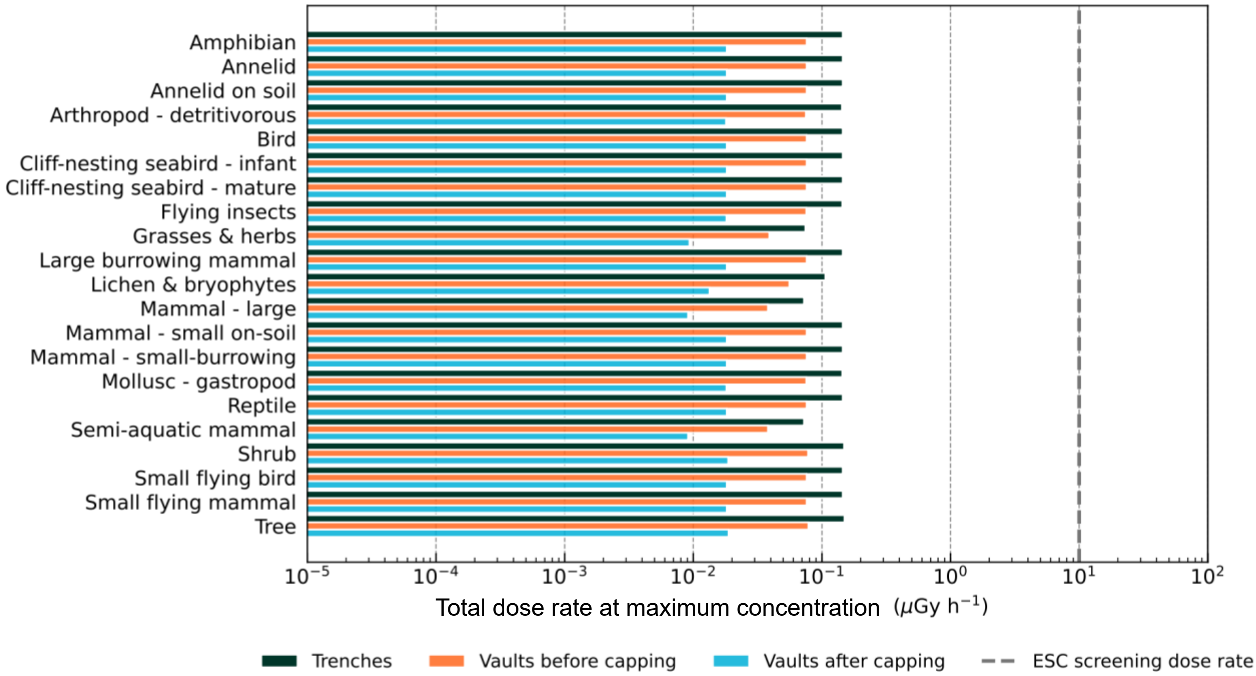


Figure 5.6: Modelled total dose rates at maximum concentration associated with C-14 bearing gas exposure during the PoA (μGy h^{-1})

After the calculations underpinning the ESC were done, an error was identified in the modelling of gaseous tritium releases (see Subsection 4.1.2 of reference [14]). Correcting this error would increase tritium releases from the vaults by a factor of between 1.2 and 3.8. Dose rates to organisms above the vaults from gaseous tritium releases are all several orders of magnitude below the screening dose rate. Therefore, they would still be below the screening dose rate if the error were corrected.

5.5 Dose Rates from Direct and Scattered Radiation from Uncovered Vaults

The PoA assessment evaluates potential dose rates for people due to direct irradiation from uncovered waste stacks and also from associated scattered radiation [44]. In assessing potential dose rates to the public, distances from uncovered vault disposals range from about 90 m to 300 m. Shorter distances would be relevant to NHB, as non-human organisms could get closer to the waste stacks than members of the public.

We use the PoA models to calculate direct and scattered radiation dose rates at 30 m from the vaults. The direct irradiation model tends to overestimate dose rates at distances shorter than 30 m. As the site is likely to be substantially disturbed by operations at distances shorter than 30 m, we only evaluate potential doses to NHB at distances of 30 m or more.

Outputs from the PoA model are expressed as effective dose rates to humans (Sv h^{-1}). Because these dose rates are for low-LET photon irradiation, we can apply a radiation weighting factor of 1.0 meaning the effective dose rates can be taken to be numerically equal to the absorbed dose rates (Gy h^{-1}) received. As absorption of photons by human tissue and biota tissue will be similar, these absorbed rates will also apply to biota under the same irradiation conditions.

The resulting dose rates at 30 m from each of the vaults are provided in Table 5.2; these are calculated when each of the vaults is full and the results are broken-down by contributing radionuclide [55]. Each of the vaults will be capped when full, shielding the photon emissions, so the contributions from the different vaults need not be combined.

Table 5.2: Calculated dose rates (µGy h^{-1}) for exposure by direct and scattered irradiation from uncapped vaults at 30 m, when each vault is full

Source (year)	Direct	Scattered	Total
Vault 8 when full (2018 AD)	0.37	0.12	0.49
Vault 9* – Existing & Committed (2030 AD)	0.23	0.16	0.39
Vault 9* – Future (2047 AD)	4.7	0.91	5.6
Vault 10 (2090 AD)	3.5	0.81	4.3
Vault 11 (2113 AD)	**24**	4.6	**28**
Vault 12 (2135 AD)	2.7	3.1	5.8

Notes: The results for Vault 11 exceed the 10 µGy per hour screening value and are highlighted in bold. * The Vault 9 result distinguishes existing and committed disposals to 2030 AD, from anticipated subsequent disposals ('future' in the table above), see reference [44].

Table 5.2 shows that only the assessed dose rates for Vault 11 at 2113 AD exceed the NHB screening value of 10 µGy h^{-1} at 30 m. The dose rates at this time are dominated by Co-60 and Cs-137 (the latter due to inclusion of its short-lived daughter Ba-137m). The results for Vault 11 when full at 2113 AD at distances between 30 m and 120 m are given in Table 5.3. These show that the calculated dose rates at 2113 AD fall below the NHB screening value within 90 m of Vault 11. Those that exceed the screening dose rate are highlighted in bold.

Table 5.3: Calculated dose rates (µGy h^{-1}) for exposure by direct and scattered radiation from Vault 11 when full at 2113 AD, prior to capping

Distance	Direct	Scattered	Total
30 m	**24**	4.6	**28**
60 m	**12**	2.5	**15**
90 m	6.3	1.5	7.8
120 m	3.3	0.94	4.2

Given that the screening dose rate is exceeded at 2113 AD adjacent to Vault 11, further consideration is required, following the steps set out in Subsection 3.4.

Step 1: Time-dependency – for this pathway, time-dependency is already explicitly taken into account in the results presented above.

Step 2: Radiosensitivity – the exceedance of the screening dose rate only occurs close to Vault 11, so the spatial extent is limited (see, also Step 3). Those representative organisms that have the smallest population ranges such as annelids and detritivorous arthropods will be the most exposed. These representative organisms are associated with the higher 400 – 4000 µGy per hour DCRL band, which is significantly higher than the assessed dose rate from this pathway.

Step 3: Spatial scale – the screening dose rate is only exceeded within 90 m of Vault 11. This is a small spatial extent, relative to the adjacent habitat. Furthermore, the immediate vicinity of Vault 11 will comprise engineering, notably concrete, and be subject to disturbance through operations and preparation for emplacement of the final cap. It is, therefore, considered very unlikely that a sufficient proportion of an NHB population would be present in the small region within which the calculated dose rates exceed the screening dose rate.

Step 4: Additional considerations – the calculated dose rate from direct and scattered radiation from the waste stack is pessimistic in that (i) it only applies perpendicular to the waste stacks whereas off-axis dose rates, as would be received over most of the area potentially affected, would be lower, and (ii) it excludes some components that will contribute to shielding, including clean grout between the waste and container walls.

Given these considerations, we conclude that the potential dose rates to NHB from direct and scattered radiation from waste stacks would not have adverse impacts on NHB populations.

5.6 Impacts Due to Releases from the LLWR Site at Present and During the PoA

We used the reference inventory to assess dose rates to NHB at maximum environmental concentrations during the PoA for several pathways. We found that total dose rates were below the screening level of 10 µGy h^{-1} for all representative organisms for all pathways except direct and scattered radiation from the open vaults. The direct and scattered radiation will affect only a small area and we expect that only a few individual organisms would be exposed to it. Therefore, we conclude that during the PoA ionising radiation from the LLWR will not have detrimental radiological impacts on populations of NHB.

For discharges to the marine environment via the marine pipeline, the highest dose rate was for phytoplankton (0.038 µGy h^{-1}) with all other representative organisms being at least three orders of magnitude below the screening value.

Doses to NHB arising from historical contamination of the Drigg Stream were below the screening value, with insect larvae being the most exposed at 4.9 µGy h^{-1}.

For discharge to the Drigg Stream followed by flooding during a storm, the lichen and bryophyte representative organism was the most exposed, at 0.18 µGy h^{-1}. This is well below the screening value.

The semi-aquatic mammal that might spend time in both the terrestrial ecosystem (contaminated following flooding) and freshwater ecosystem (historical contamination in the Drigg Stream) had a total dose rate of 0.39 µGy h^{-1}.

The modelled total dose rates at maximum concentration to terrestrial representative organisms during the PoA following exposure to tritium are several orders of magnitude below the screening value, being 0.003 and 0.002 µGy h^{-1} for the trenches and vaults respectively.

For the Rn-222 fluxes from the trenches and vaults, the dose rates to terrestrial representative organisms are also low. The highest dose rates of 0.054, 0.0072 and 0.00034 µGy h^{-1} for the trenches, vaults before capping and vaults after capping, respectively, are for the small flying mammal representative organism. All other terrestrial representative organisms had doses between 2 and 15 times lower than the small flying mammal.

Fluxes of C-14 from the trenches and vaults during the operational period are considered in our assessment. All the dose rates associated with maximum C-14 exposure in gaseous form are at least an order of magnitude below the screening value. Typically, the dose rates are around 0.15 µGy h^{-1} and 0.076 µGy h^{-1} for the trenches and vaults respectively.

We assess dose rates to people from direct radiation and associated scattered radiation from uncovered waste stacks at distances from 90 m to 300 m from the stacks. We use these data to evaluate dose rates to NHB. Our findings show that up to 90 m from Vault 11 in 2113 AD, the dose rates from direct and scattered radiation exceed the screening dose rate.

The dose rates are 28, 15 and 7.8 µGy h^{-1} for distances of 30, 60 and 90 m respectively. However, the spatial extent of the high assessed dose rates is limited to the area within 90 m of the waste stacks. When considering radiosensitivity, it is likely to be those species that have the smallest home ranges (for example, annelids and detritivorous arthropods) that may be most affected. These representative organisms are associated with a 400 to 4000 µGy h^{-1} DCRL band, which is significantly higher than the assessed dose rates. Additionally, much of this area will comprise engineered facilities and be subject to normal operations at the time of these exceedances. There are also notable pessimisms in the calculations as the dose rates only apply perpendicular to the waste stacks, meaning the dose rates will be lower off this axis. Therefore, we can be confident then that while individual organisms may be exposed to these dose rates from direct and scattered radiation, NHB populations will be protected.

In summary, the releases of radionuclides from the LLWR throughout the PoA are assessed as giving dose rates to NHB that are below the screening value of 10 µGy h^{-1} based on the maximum environmental concentrations that will occur at different times throughout the PoA. Only the direct and scattered radiation exposure gives rise to dose rates above the screening value of 10 µGy h^{-1}. These dose rates only cover a small area which is likely to have limited suitable habitat for NHB, and it is unlikely that populations of organisms will be exposed to these dose rates. Therefore, we conclude that these dose rates would not lead to adverse impacts to populations of NHB.

6 Assessment of Radiological Impacts following the PoA

This section gives the assessed dose rates to non-human biota after the PoA for each pathway. It then discusses the potential impacts of these dose rates.

6.1 Groundwater

6.1.1 Groundwater Releases to the Coast and Estuary after the PoA

The modelled total dose rates at maximum concentrations to representative organisms exposed to contamination following releases via groundwater after the PoA are typically more than five orders of magnitude below the screening dose rate throughout the local offshore, foreshore, lower Irt estuary and salt marsh model regions. The highest calculated total dose rate for any representative organism is 3.8 10^{-5} µGy h^{-1} for small benthic crustaceans in the foreshore model region.

There are two cases relating to groundwater releases after the PoA where representative organisms may reside in multiple model regions:

- large marine mammals residing in the local offshore water and on the foreshore;
- semi-aquatic mammals spending time in both the lower Irt estuary water and on the saltmarsh.

The modelled total dose rates at maximum concentrations are 1.6 10^{-6} and 1.2 10^{-7} µGy h^{-1} for the large marine mammal and the semi-aquatic mammal respectively.

6.2 Coastal Erosion

6.2.1 Terrestrial Ecosystem Exposure

The modelled total dose rates at maximum concentrations to terrestrial biota on the cliff and storm beach are given in Table 6.1. Where the total dose rate at maximum concentrations exceeds the screening dose rate of 10 µGy h^{-1}, the values are given in italics. Where that dose rate is exceeded by exposure to the modelled maximum concentration of an individual radionuclide, the value is given in bold. These dose rates exclude exposure to C-14, as it will not be in a suitable form for uptake by biota.

Table 6.1: Modelled total dose rates at maximum concentrations for terrestrial representative organisms occupying the cliff or storm beach model regions following coastal erosion (µGy h^{-1})

Representative organism	Cliff		Storm beach	
	Total dose rate (µGy h^{-1})	Key radionuclides	Total dose rate (µGy h^{-1})	Key radionuclides
Amphibian	**68**	**Th-228, U-234, U-238, Ra-228**	-	-
Annelid	**130**	**Pu-239, Ra-228, Ra-226, U-234, U-238,** Pu-240, Am-241, Th-228	-	-
Arthropod - detritivorous	**160**	**U-234, U-238, Pu-239, Ra-226**	**70**	**U-234, U-238, Pu-239,** Pa-228, Ra-226, Pu-240
Bird	*16*	Ra-228, Ra-226, Cl-36, Am-241, Pu-239, Th-228	6.8	Ra-228, Cl-36, Pu-239, Am-241, Ra-226, Th-228
Flying insects	**44**	**Pu-239,** Am-241, Ra-228m, Ra-226, Pu-240	*19*	Pu-239, Am-241, Ra-228, Pu-240, Ra-226
Grasses and herbs	**240**	**Pu-239, Ra-228, Ra-226,** Am-241, Pu-240	**100**	**Th-228, U-234, U-238,** Ra-228
Lichens and bryophytes	**1300**	**U-234, U-238, Th-228, Pu-239, Am-241**	**-**	**-**
Mammal - large-burrowing	*19*	Pu-239, Ra-228, Ra-226	-	-

Representative organism	Cliff		Storm beach	
	Total dose rate ($\mu Gy\ h^{-1}$)	**Key radionuclides**	**Total dose rate ($\mu Gy\ h^{-1}$)**	**Key radionuclides**
Mammal - small on-soil	**38**	Pu-239, Ra-228, Ra-226, Pu-240, Cl-36	-	-
Mammal - small-burrowing	**40**	Pu-239, Ra-228, Ra-226, Pu-240, Th-228, Cl-36	-	-
Mollusc - gastropod	**160**	**Pu-239, Pu-240, U-234, U-238, Am-241**	-	-
Reptile	**56**	**Ra-228, Ra-226,** Pu-239, Am-241	*23*	Ra-228, Pu-239, Ra-226, Am-241
Shrub	**170**	**Th-228, Ra-228, U-234, Pu-239, U-238, Ra-226**	**-**	**-**
Small flying bird	*17*	Ra-228, Ra-226, Cl-36, Am-241, Pu-239, Th-228	6.8	Ra-228, Cl-36, Pu-239, Am-241, Ra-226, Th-228
Small flying mammal	*37*	Pu-239, Ra-228, Ra-226, Pu-240, Cl-36	*16*	Pu-239, Ra-228, Ra-226, Pu-240, Cl-36
Tree	*12*	U-234, U-238, Ra-228, Ra-226, Th-228, Pb-210, Pu-239	-	-
Values in italics indicate that the total dose rate at maximum concentrations exceeds the screening dose rate. Values in bold indicate that the dose rate arising from the maximum concentration of a single radionuclide exceeds the screening dose rate.				

6.2.1.1 Time-dependency of Modelled Dose Rates

For the cliff model region, Table 6.1 shows that the total dose rates at maximum concentrations for all representative organisms exceed the screening dose rate. In the storm beach model region, the total dose rates at maximum concentrations for five representative organisms exceed the screening dose rate. Since it is unlikely that the maximum concentrations for each radionuclide would occur simultaneously, the highest time-dependent dose rates were considered. Figure 6.1 shows that for all representative organisms in the cliff model region, other than birds, small flying birds and trees, the screening dose rate is exceeded at some point during the assessment period.

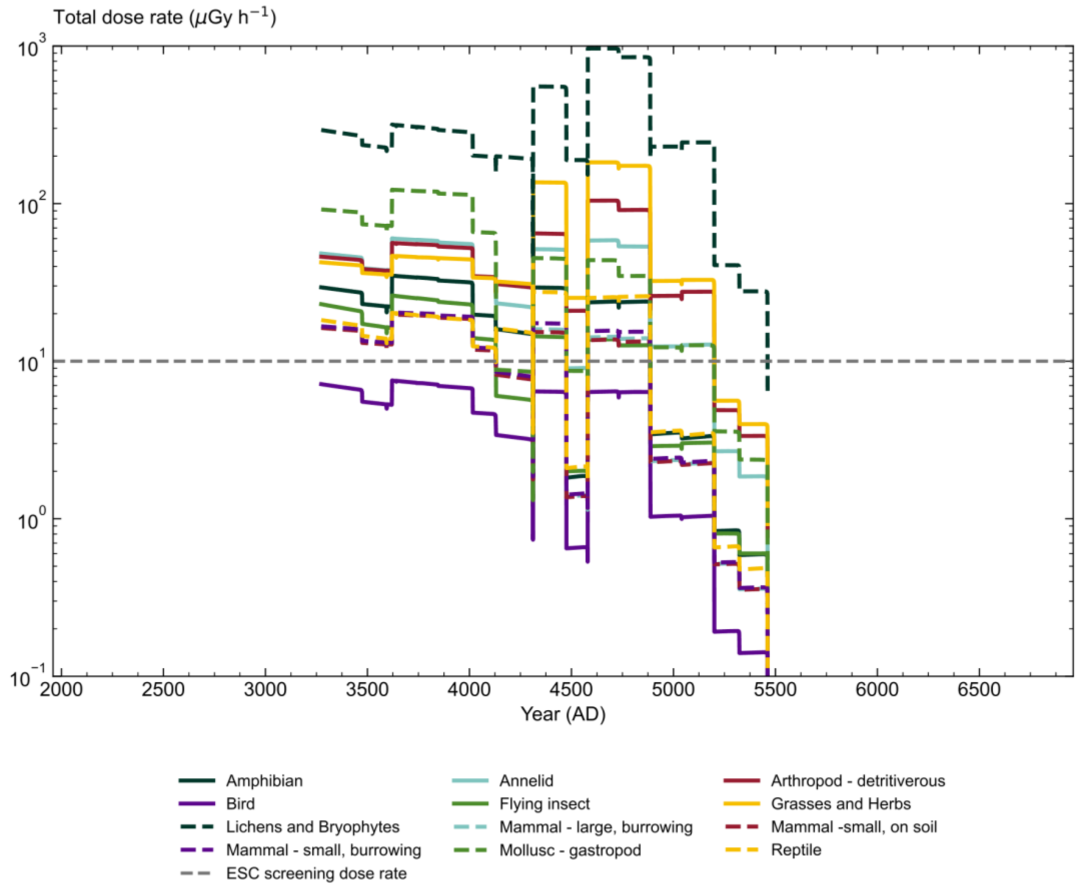


Figure 6.1: Time-dependent dose rates to all representative organisms in the cliff model region.

In the storm beach model region five representative organisms have modelled dose rates that equal or exceed the screening dose rate at some point during the assessment period (Figure 6.2).

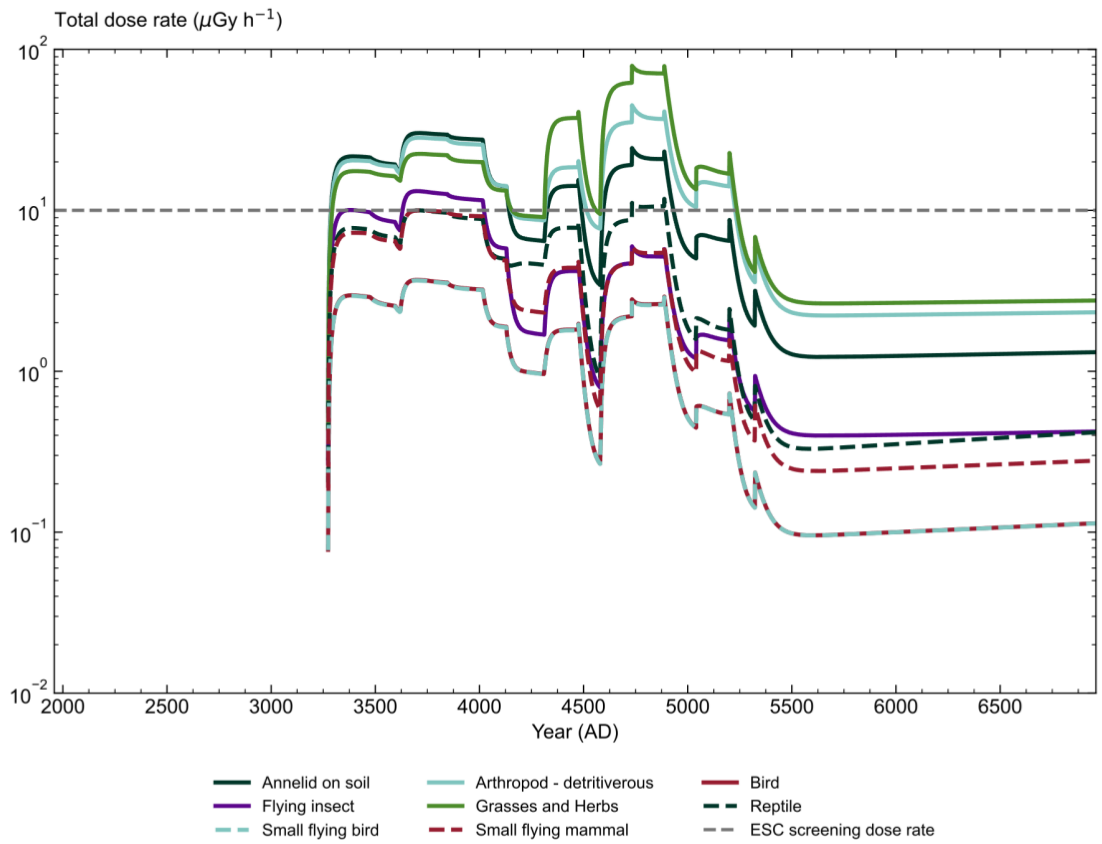


Figure 6.2: Time-dependent dose rates to all representative organisms in the storm beach model region

6.2.1.2 Consideration of Radiosensitivity

Since both model regions have representative organisms which exceed the screening dose rate at some point during the assessment period, consideration was given to the radiosensitivity of the organisms. Representative organisms are compared against their associated DCRL range (Table 6.2). No organisms have modelled dose rates which exceed the upper bound of the DCRL range, those that lie within the DCRL range are highlighted in **bold**. Birds, small flying birds, and trees exceed their respective lower DCRL value, but do not exceed the screening dose rate. Lichens and bryophytes do not have an appropriate ICRP RAP and therefore merit further consideration.

Table 6.2: Comparison of peak assessed (time-dependent dose rate) on the cliff or storm beach following coastal erosion with relevant DCRLs

Representative Organism	Cliff (μGy h^{-1})	Storm beach (μGy h^{-1})	ICRP RAP	DCRL Range (μGy h^{-1})
Amphibian	35	-	Frog	40 – 400
Annelid	60	-	Earthworm	400 - 4000
Arthropod - detritivorous	110	45	Bee	400 – 4000
Bird	**8.0**	3.7	Duck	4 - 40
Flying insects	26	13	Bee	400 – 4000
Grasses and herbs	**180**	**79**	Wild grass	40 – 400
Lichens and bryophytes*	970	-	-	-
Mammal - large-burrowing	**20**	-	Deer	4 – 40
Mammal - small on-soil	**20**	-	Rat	4 – 40
Mammal - small-burrowing	**20**	-	Rat	4 - 40
Mollusc - gastropod	120	-	Earthworm	400 - 4000
Reptile	**28**	**12**	Duck	4 – 40
Shrub	**110**	-	Wild grass	40 – 400
Small flying bird	**7.0**	3.7	Duck	4 - 40
Small flying mammal	**20**	**10**	Rat	4 – 40
Tree	**9.0**	-	Pine tree	4 – 40

6.2.1.3 Area of Coast Affected by Coastal Erosion of the LLWR

The spatial scale of the modelled cliff and storm beach impacted by coastal erosion of the LLWR merits consideration.

The modelled concentration in the cliff applies to the width of coast immediately adjacent to the LLWR during erosion and cautiously represents concentrations close to those in the waste itself. The length of cliff represented varies depending on the length of the eroding face of the repository from about 160 m to about 700 m. The modelled storm beach represents a strip of the beach above the foreshore immediately adjacent to the section of cliff that is modelled, extending 40 to 50 m away from the cliff line. The area of the storm beach modelled varies depending on the length of the eroding face of the repository from about 6.5 10^3 m^2 to about 3.1 10^4 m^2 [47].

The affected areas of cliff and storm beach are only a small portion of the extent of these habitats along the west Cumbrian coast. They are transient environments with continual turnover, migration and movement of organisms between the potentially contaminated area and other areas of beach and cliff further away. Many of the assessed organisms are unlikely to spend all their time in the cliff and storm beach habitats, with many likely to be only temporary visitors. Therefore, it is unlikely that whole populations of organisms would be continuously exposed to these dose rates.

6.2.1.4 Lichens and Bryophytes on the Cliff

Lichens and bryophytes exceed the screening dose rate, do not have a DCRL against which to compare their radiosensitivity and do not have an applicable area for the maintenance of a population. Lichens and bryophytes are therefore considered further in the following subsection.

Lichens and bryophytes are the most exposed terrestrial representative organism, with a peak time-dependent dose rate of 9.7 10^2 μGy h^{-1} in the cliff model region. The radionuclides contributing most to the assessed dose rates are U-234, U-238, Th-228, Pu-239, and Am-241, each of which leads to a dose rate at maximum concentration over 80 μGy h^{-1} in the cliff model region (Figure 6.3). Additional radionuclides leading to dose rates at maximum concentrations exceeding the screening value of 10 μGy h^{-1} are Ra-228, Pb-210, Ra-226, Th-232, Pu-240, and U-235. Figure 6.3 shows that the highest calculated dose rates occur after about 4500 AD; these are associated with erosion of Trench 4.

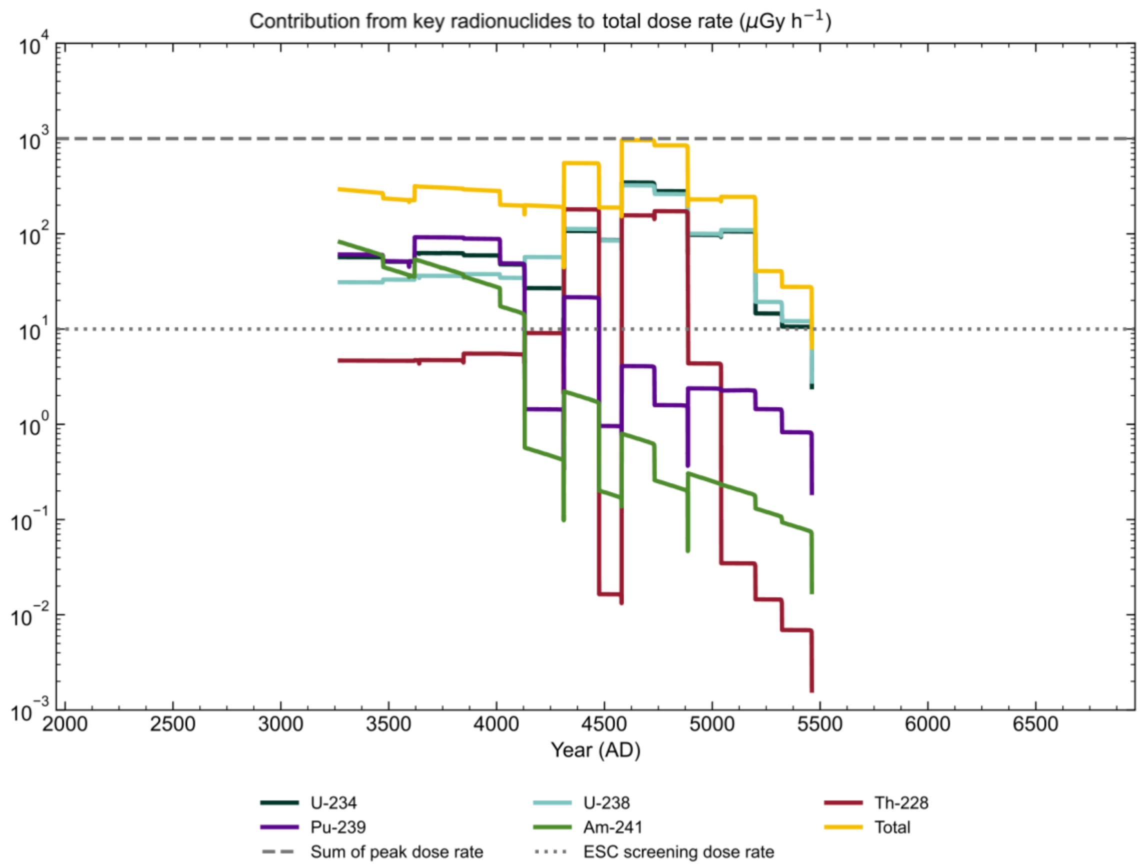


Figure 6.3: Modelled dose rate over time to lichens and bryophytes in the cliff model region as impacted by coastal erosion (μGy h^{-1})

Most of the radionuclides that result in dose rates above the screening value for lichens and bryophytes are alpha-emitting radionuclides. Those that are not themselves alpha-emitting radionuclides (Pb-210 and Ra-228) are beta emitters that have substantial alpha emissions associated with decay chain members (Po-210, Th-228 and Ra-224) [24]. The main contribution of these radionuclides to the total assessed dose rates is from internal exposure.

Lichens and bryophytes are radioresistant and are the least radiosensitive of the plant phyla [34]. Dose rates that would give rise to a 10% reduction in the density of lichen in the exposed group in comparison to a control group (EDR_{10}, μGy h^{-1})[6] have been reported. Appendix 2 of reference [56] gives an EDR_{10} value of over 100,000 μGy h^{-1} for lichens, more than three orders of magnitude greater than the highest time-dependent dose rate calculated in this assessment.

There is uncertainty as to whether viable populations of lichens and especially bryophytes would colonise and exist on the cliff. Both types of organism require time to establish, while

[6] The 10 μGy h^{-1} screening value used is defined as the dose rate that is below the EDR_{10} for 95% of species (considering the most restrictive EDR_{10} reported for each species), multiplied by an uncertainty factor of 2 [24]. Therefore, we consider the EDR_{10} for the individual organism to be an appropriate indicator of whether the assessed dose rates could lead to a population-level effect on that specific organism.

the cliff is an inherently dynamic system, with ongoing erosion. The modelled exposures would only occur along a relatively short section of cliff line, as the LLWR is eroded. Lichen and bryophytes on the cliff either side of this section would be unaffected.

The assessment of radiological impacts to NHB during coastal erosion uses soil-to-organism concentration ratios to model uptake of radionuclides. This assumes that the radionuclides are transferred to the organisms from soil. Lichens and bryophytes are non-vascular; they do not have roots but rather obtain their nutrients from the air [57]. The applicability of the assessment results for lichens and bryophytes is, therefore, questionable. The radionuclides released during coastal erosion will not be present as gases or vapours, so there is very limited potential for them to be taken up by lichens and bryophytes. Consequently, we expect radionuclide concentrations in lichens and bryophytes to be very low, and significantly below those modelled in the assessment calculations.

It has also been suggested that apparent uptake of elements such as uranium by lichens and bryophytes may result from unremoved surface contamination prior to analysis [24]. As such, measured activity concentrations in lichens and bryophytes may overestimate uptake, especially for relatively unavailable elements including uranium, thorium, and plutonium.

Given these points around their low radiosensitivity, ability to colonise the cliff area, and the route of radionuclide uptake, it is unlikely that lichen and bryophyte populations would suffer deleterious radiological impacts from erosion of the LLWR site.

6.2.1.5 Summary of Assessment of Terrestrial Organism Exposure Following Coastal Erosion

The assessed total dose rates at maximum concentrations for most representative organisms at the cliff and storm beach exceed the screening dose rates. However, in considering the time-dependency of the dose rates and the radiosensitivity of the organisms we find that the organisms are not exposed to dose rates that are likely to have deleterious effects for populations. Given the transient nature of the environments, their setting as part of a larger continuous area of similar habitats and the likely habits of terrestrial organisms, we also think it unlikely that populations of organisms would be continuously exposed to the assessed dose rates. Therefore, we conclude that populations of organisms present at the cliffs and storm beach will not suffer negative radiological impacts.

We considered lichens and bryophytes in more detail, as these had the highest assessed dose rates, but do not have a DCRL. We do not expect populations of lichens and bryophytes to suffer negative radiological impacts for the following reasons. Literature information shows that these organisms have very low radiosensitivity. The assessment of dose rates to lichens is based on soil-organism concentration ratios; however, lichens and bryophytes would take radionuclides up from the air, rather than the soil. We do not expect the radionuclides released during coastal erosion to be in a form likely to be taken up by lichens and bryophytes. The assessed organism radionuclide concentrations and, thus, internal dose rates for these organisms are likely to be significant overestimates. Populations of lichens and bryophytes are unlikely to establish on the storm beach.

6.2.2 Marine Ecosystem Exposure

The modelled NHB dose rates at maximum concentrations to marine biota occupying the foreshore and local offshore model regions during coastal erosion are shown in Figure 6.4. There are no cases where the total dose rate at maximum concentration exceeds the screening dose rate of 10 μGy h^{-1}, although phytoplankton approach that value in the local coastal water with a maximum dose rate of 9.7 μGy h^{-1}. Results for all other representative organisms remain more than an order of magnitude below the screening value of 10 μGy h^{-1}.

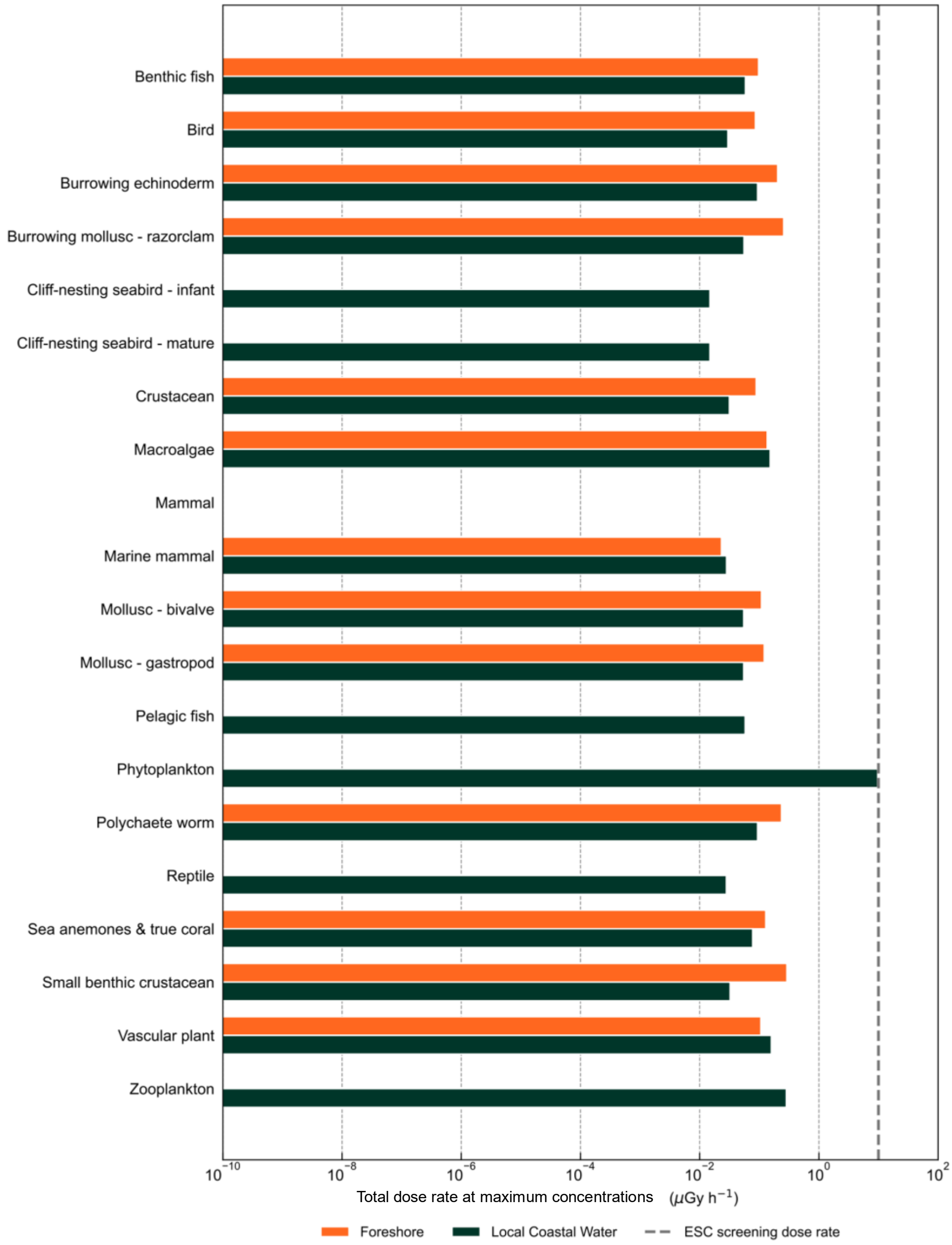


Figure 6.4: Total dose rates (μGy h^{-1}) at maximum concentrations in the foreshore and local coastal water model regions after coastal erosion of the repository [24]

6.2.3 Exposure of Organisms that Reside in both Terrestrial and Marine Ecosystems

We consider the potential exposure of cliff-nesting seabirds and marine mammals that may spend time on the foreshore sediment when the tide is out. There are two cases where representative organisms may reside in multiple model regions contaminated following coastal erosion:

- cliff-nesting birds (infants and adults) residing on the cliff and obtaining their food from the local offshore water;
- large marine mammals residing in the local offshore water and on the foreshore (beach).

Table 6.3 gives the calculated dose rates to these representative organisms across the multiple model regions. The total calculated dose rates in all cases remain below the 10 µGy h^{-1} screening dose rate. Consistent with the location of the highest calculated concentrations arising from coastal erosion of the LLWR, external irradiation from the cliff dominates calculated exposures for the cliff-nesting birds and external irradiation from the foreshore dominates for the large marine mammal.

Table 6.3: Total dose rate at maximum concentrations to cliff-nesting seabirds accounting for multiple ecosystem residencies (µGy h^{-1})

Representative Organism and Source	Location and pathway	Dose rate (µGy h^{-1})	
		By pathway	Total
Cliff nesting bird (infant)	Internal local offshore water (100%)	$3.0\ 10^{-2}$	$8.3\ 10^{-1}$
	External local offshore water (60%)	$4.2\ 10^{-8}$	
	External cliff (40%)	$8.0\ 10^{-1}$	
Cliff nesting bird (adult)	Internal local offshore water (100%)	$3.0\ 10^{-2}$	$4.3\ 10^{-1}$
	External local offshore water (60%)	$5.6\ 10^{-8}$	
	External cliff (40%)	$4.0\ 10^{-1}$	
Large marine mammal	Internal local offshore water (100%)	$8.5\ 10^{-2}$	2.6
	External local offshore water (80%)	$4.5\ 10^{-7}$	
	External cliff (20%)	2.5	

6.3 Radon

The total dose rates at maximum concentration associated with Rn-222 exposure post-final cap emplacement are shown in Figure 6.6(a) and, with an extended y-axis used to show the dose rates associated with aboveground Rn-222, in Figure 6.6(b). Here we assume that the concentrations of Rn-222 in the soil gas following complete degradation of the geomembrane would be encountered by all organisms, except for bats and birds for which occupancy in soil is assumed to be zero.

Dose rates are at least two orders of magnitude lower than the screening dose rate, with the small burrowing mammal being the most exposed with a maximum dose rate of 6.0 10^{-3} µGy h^{-1} in topsoil above the LLWR trenches.

6.4 Release of Gaseous C-14

The modelled total dose rates to terrestrial representative organisms resulting from maximum gaseous C-14 releases from the trenches and vaults are shown in Figure 6.5. This figure shows the dose rates after the PoA. All the dose rates associated with maximum C-14 exposure in gaseous form are more than an order of magnitude below the screening value. For locations, other than over the vaults after geomembrane degradation the dose rates are three or more orders of magnitude lower than the screening value.

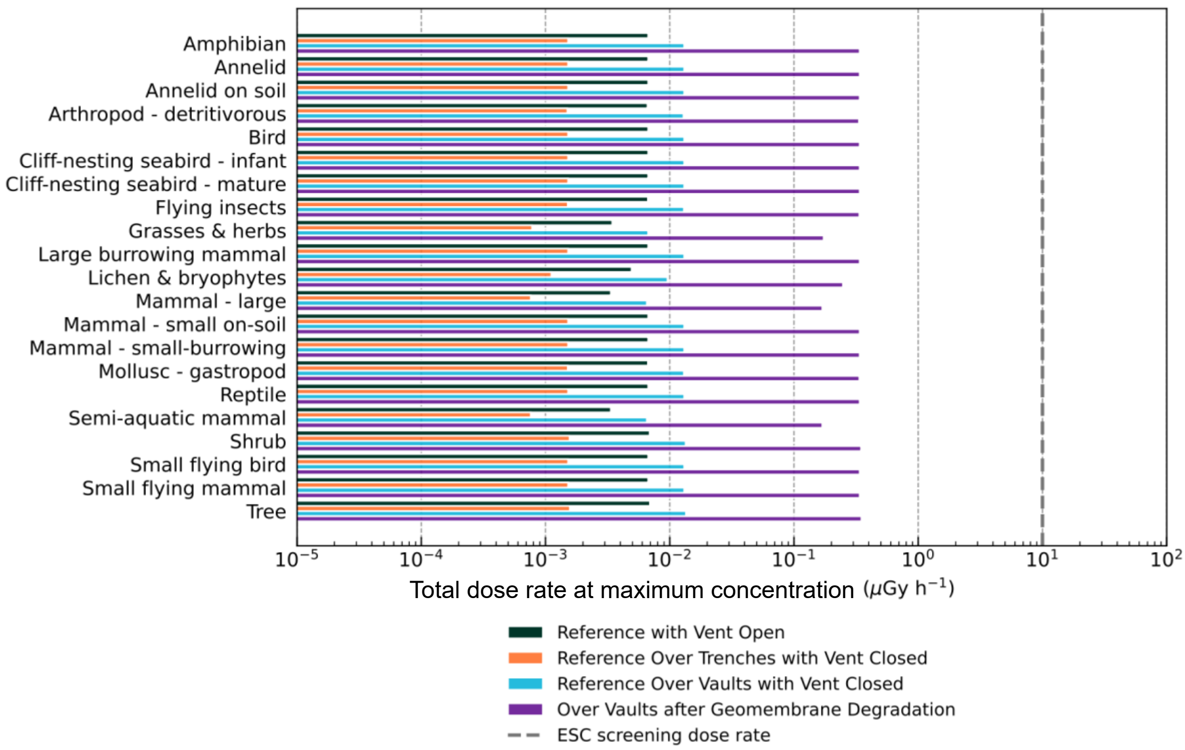


Figure 6.5: Modelled total dose rates at maximum concentration associated with C-14 bearing gas exposure after the PoA (µGy h^{-1})

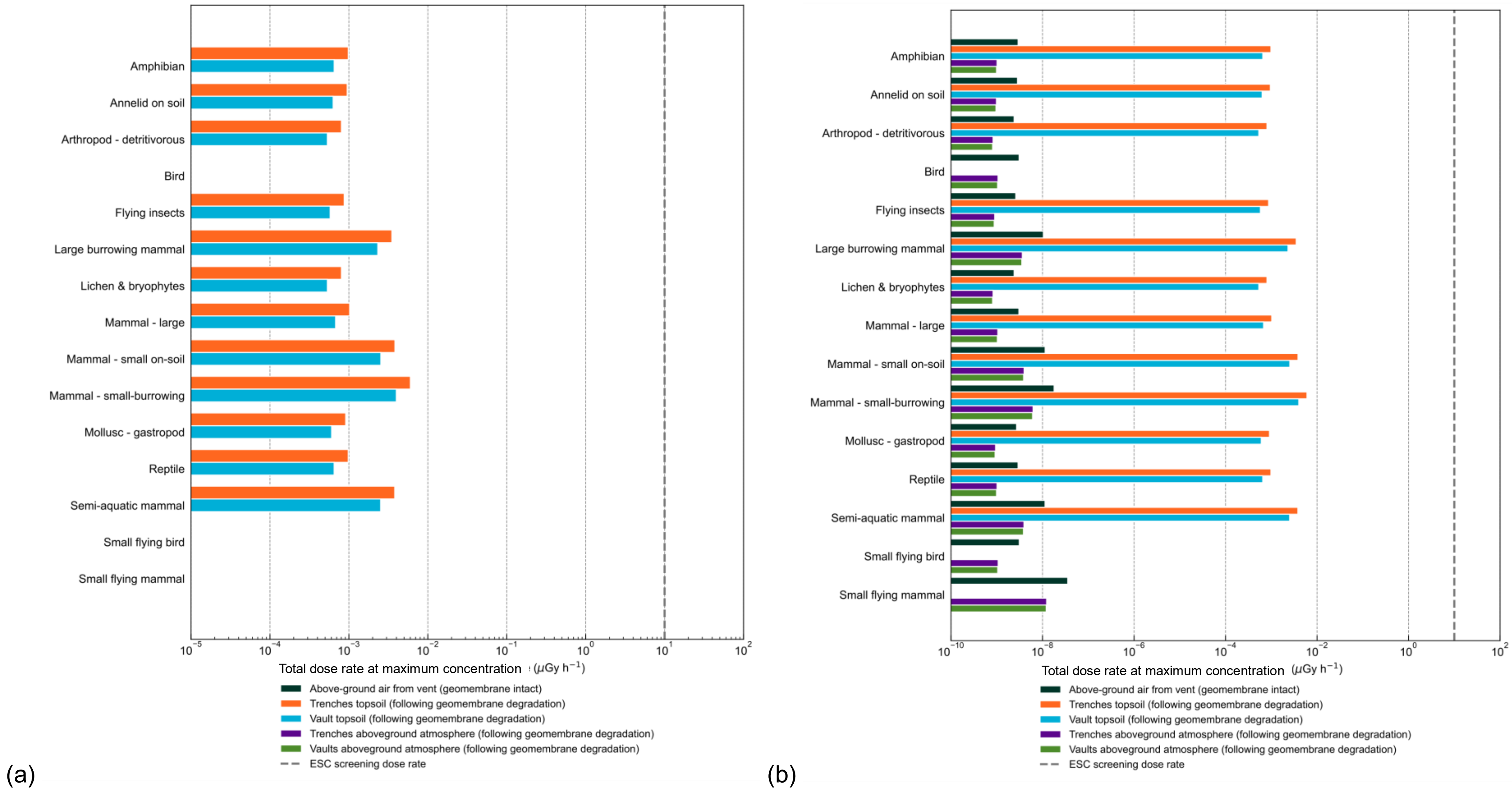


Figure 6.6: Modelled total dose rates at maximum concentration associated with Rn-222 exposure, post-final cap emplacement (μGy h^{-1}). (a) Dose rates in topsoil following geomembrane degradation. (b) Dose rates in all locations.

6.5 Impact of High-Activity particles

Some of the contamination within the LLWR may be present in the form of small particles with relatively concentrated activity. Particles could be released during coastal erosion or human intrusion. Any particles released could vary in terms of the radionuclides present, the concentrations and total amounts of those radionuclides, and the size and physicochemical composition of the particles.

We do not expect significant releases of radioactive particles from the LLWR. However, we have cautiously assessed the impacts of NHB encountering particles. We use information about the types of particles found on beaches at Dounreay, Sellafield and Dalgety Bay, and about the types of material that have been disposed of at LLWR (e.g. low activity sources and mineral sands) to inform this assessment. A range of particles is considered, some of which are or could well be present in the LLWR and others that are much less likely to be present. We cautiously assess on the basis that all the inventory of the relevant radionuclides at the LLWR could be released as particles [48].

Of the particles considered, we find that uranium particles have most potential to persist in the environment (because they are highly refractory). Based on observed particle size and uranium inventory in the LLWR, they also have most potential to lead to exposures that could affect many organisms but that may not be well covered by the heterogeneous contamination assumption in the assessment of impacts during coastal erosion (Subsection 6.2). Therefore, we quantitatively assess the impact of exposure of biota to these particles. We reiterate that this assessment is based on the extremely cautious assumption that the entire uranium inventory can be released as particles; it does not represent an expected or likely situation, and we have no evidence to suggest that there will be large scale releases of such particles from the repository.

If particles were released, organisms could be exposed by respiratory intake, ingestion and external irradiation. Some organism types, for example filter feeders, may be particularly exposed to particles. Only small particles of up to about 5 μm radius are likely to be taken into the respiratory system, either via inhalation or via the gills.

Particles that persist in the environment are likely to be highly refractory such that, if ingested, the particle will remain intact and radionuclide uptake across the gut wall will be very limited. Therefore, we focus on the likely impact on tissues close to the particle. We assess the impacts of exposure to radioactive particles using a calculation of the dose rates local to individual particles in individual organisms.

Table 6.4 shows how absorbed dose rate to nearby tissue varies with particle size. For small particles, the dose rate increases rapidly with size. However, with larger particles, the increase in size is largely compensated by the increase in mass of tissue irradiated and the decreasing fraction of alpha energy that escapes from the particle (self-absorption).

Table 6.4: Local absorbed dose rates for UO_2 particles of different sizes[7]

Radius of particle (μm)	Dose rate (mGy h^{-1})
1	0.44
10	17
100	32
1000	32

We focus on particles with a radius of around 10 μm. For a given activity released as particles, this size would lead to a relatively high prevalence of particles in the environment, making it likely that organisms could encounter multiple particles, while not being well represented by the uniform contamination models used in the ERICA assessments.

The uranium particles considered would be predominantly alpha emitting. External contamination would not be radiologically significant, because of the short range of alpha particles in tissues and because the outermost surface of organisms is both an effective barrier to entry of particles and is not likely to be radiosensitive, with the sensitive cells lying at depths of several tens of μm or more.

Absorbed dose rates around such particles are likely to be 20 mGy h^{-1} (Table 6.4).[7] Although alpha irradiation is highly effective at sterilising and killing cells, tissue function may not be impaired if the cells are not required to reproduce, e.g. fibroblasts in muscle. Sterilisation is also likely to be achieved at absorbed doses of less than 1 Gy. However, cell death may not be significant until absorbed doses of several Gy have been received. Therefore, the effects of exposure to particles with a local dose rate of around 10 mGy h^{-1} would likely be a few microscopic lesions around trapped particles with no significant impact on the health of the individual organism. Thus, overall, there is little reason to anticipate biological impacts of such particles over and above a limited number of microscopic lesions in some individuals within a population or community. Those lesions are unlikely to have adverse impacts on the health, fertility or fecundity of organisms. Therefore, effects at the population or community level are not expected to arise.

6.6 Impacts Due to Releases from the LLWR Site following the PoA

Table 6.5 summarises the most exposed representative organisms after the PoA. Where the modelled total dose rate at maximum concentrations exceeds the screening value of 10 μGy h^{-1} it is given in bold. For the two coastal erosion results (terrestrial and marine) in Table 6.5 those radionuclides that dominate the assessed dose rates are given. Where an

[7] This is dose rate in the local tissue volume around the particle (10 μm tissue shell). It is not a whole-body dose rate and, thus, can't be directly compared with the screening dose rate

individual radionuclide also exceeds the 10 µGy h^{-1} screening value on its own, it is given in bold.

Table 6.5: Summary of most exposed representative organism and associated total dose rates at maximum concentrations (µGy h^{-1}), from the assessment pathways considered post-PoA

Contaminant Pathway	Most exposed representative organism	Model Region	Total dose rate at maximum concentrations (µGy h^{-1})	Dominant radionuclide
Gas: Rn-222	Small burrowing mammals	Trenches (geomembrane degradation)	$6.0\ 10^{-3}$	-
Gas: C-14 (post-PoA)	Tree	Vaults (geomembrane degradation)	$3.5\ 10^{-1}$	-
Coastal erosion: terrestrial[8]	Lichen and bryophytes	Cliff	**$1.3\ 10^{3}$**	**U-234, U-238, Th-228, Pu-239, Am-241**
Coastal erosion: marine	Phytoplankton	Foreshore	9.7	-
Coastal erosion: resident in both terrestrial and marine ecosystem	Cliff-nesting bird (infant)	Cliff	$8.3\ 10^{-1}$	-

Post-PoA, the majority of the modelled maximum dose rates to NHB are lower than 10 µGy h^{-1}. The exceptions arise for the assessment of exposure of representative organisms to wastes on the cliffs and storm beach because of coastal erosion, where, in addition to the most exposed organisms noted in Table 6.5, other representative organisms

[8] Table 6.5 give the most exposed representative organism. Additional representative organisms also have total dose rates at peak concentrations above 10 µGy h^{-1}.

had total dose rates at maximum concentrations that exceed the 10 μGy h^{-1} screening dose rate (Table 6.1).

When we consider the time-dependency of the concentrations on the cliff and storm beach, the results exceed the screening dose rate at some point in the assessment period for most representative organisms; only for birds, small flying birds and trees does the assessed dose rate not exceed the screening dose rate. The remaining 13 terrestrial representative organisms were considered further, using our four-step process of considering time-dependency, radiosensitivity of the organisms, spatial extent and dose assessment aspects.

For the representative organisms that exceed the screening dose rate in the cliff model region, the results show that:

- amphibians, annelids, detritivorous arthropods, flying insects and gastropods do not exceed the associated lower DCRL;
- grasses and herbs, mammals, reptiles and shrubs fall within the associated DCRL bands.

Lichens and bryophytes do not have an associated DCRL band. This representative organism has a time-dependent dose rate of 970 μGy h^{-1} in the cliff model region (Subsection 6.2.1). However, these organisms have been shown to have low radiosensitivities, the CRs used are likely to lead to overestimated internal dose rates for lichens and bryophytes and it is unlikely that populations of these organisms will establish themselves on the cliffs or storm beach. Thus, lichens and bryophytes are unlikely to be impacted at the level of the population.

We have considered the potential for harm to NHB through contact with high-activity particles (Subsection 6.5). There is limited potential for inhalation of particles by NHB. There is potential for ingestion and uptake into the gastrointestinal tract, although if any effect occurred, it would likely be a few microscopic lesions around trapped particles with no significant impact on the health of the individual organism. There is potential for a particle to be trapped against the skin of an organism. However, the outermost surface of organisms is both an effective barrier to entry of particles and is not likely to be radiosensitive, with the sensitive cells lying at depths of several tens of μm or more (below the penetration depth of alpha particles).

Overall, there is little reason to expect biological impacts of such particles other than a small number of microscopic lesions in some individuals within a population or community. Those lesions are unlikely to have adverse impacts on the health, fertility or fecundity of organisms. Therefore, effects at the population or community level are not expected to arise.

7 Findings and Conclusions

As well as ensuring that humans are adequately protected from exposure to radiation and radioactive materials originating from the LLWR, there is a need to ensure that the environment is adequately protected. To show that the environment around the LLWR is protected, we have assessed dose rates to non-human biota resulting from disposal of the LLWR Reference Inventory.

We use transport modelling from the assessments of radiological impacts to people to model the transfer of radioactivity through the environment and the maximum concentrations of radionuclides in environmental media. We also use environmental monitoring data where it is available. We identify representative organisms around the LLWR based on species that may be at or near the repository, including on nearby protected areas of land. We use version 2 of the ERICA Tool to assess the dose rates to the representative organisms. We compare the assessed dose rates to a screening dose rate of 10 µGy h^{-1}. The maximum concentrations of different radionuclides will occur at different times. Therefore, this screening is cautious.

Where the screening value is exceeded, we perform more detailed evaluation of the representative organisms and exposure pathways. Where the screening dose rate is exceeded, we consider the:

- time-dependency of calculated radionuclide concentrations and, thus, dose rates;
- radiosensitivity, e.g. with reference to DCRLs;
- size of the contaminated or affected area relative to the area required to support a population of the affected organisms;
- notable uncertainties and pessimisms associated with the modelling approach.

The findings of this assessment are summarised in Subsection 7.1. These show that exposure of biota by most pathways results in dose rates below the 10 µGy h^{-1} screening value.

Where assessed dose rates exceed these screening thresholds, detailed evaluations of the representative organisms and exposure pathways have shown that the assessed dose rates are not likely to harm populations of organisms. For example, lichens and bryophytes on the cliffs and beaches receive assessed high dose rates but have low sensitivity to radiation and are unlikely to establish themselves in the affected locations.

We conclude that even if there is potential for radiological detriment to some individuals of NHB, it is highly unlikely that there would be harm to the local populations of these species. This conclusion is based on an assessment of the potential impacts of the reference inventory. However, we have derived waste acceptance controls based on impacts to people; we conclude that these controls will also be sufficient to ensure that NHB are protected. Therefore, we conclude that the LLWR meets the requirement "*to protect* *ecosystems against radiation exposure that would have adverse consequences for a population* [of NHB]".

7.1 Summary of Assessment Findings

Summaries of the most exposed representative organisms from this assessment are given in Table 7.1 and Table 7.2 for the PoA and post-PoA periods, respectively. Where the modelled total dose rate at maximum concentrations exceeds the screening value of 10 μGy h^{-1}, it is given in bold.

During the PoA, all assessed dose rates are lower than our screening dose rate except for direct and scattered radiation for one vault (Vault 11) and time (2113 AD, when the site is still operational). Although the direct dose rate from Vault 11 (to all organisms) exceeds the screening dose rate, the affected area would be very small and dose rates in most of the affected area would be lower than calculated in our assessment. Therefore, we conclude that these dose rates will not result in detriment to populations of non-human organisms.

All other dose rates to NHB during the PoA are below the screening dose rate. The highest dose rates are from the historical contamination in the Drigg Stream for insect larvae and is 4.9 μGy h^{-1}, about half the screening value. Dose rates from the other pathways are all at least one order of magnitude lower than the screening dose rate. We find therefore that, during the PoA, the environment is protected from the radiological impacts of the repository.

Post-PoA, all assessed dose rates are below the screening dose rate except for dose rates resulting from the coastal erosion pathway.

For many of the organisms on the cliff and storm beach during erosion of the repository, considering the time-dependency of the dose rates and the radiosensitivity of the organisms shows that detrimental effects to populations of organisms are unlikely. Moreover, the areas of the cliff and storm beach affected are limited to the areas in front of the eroding repository. These will be relatively small areas, constantly disrupted by weather and wave and tidal action. They will not be large enough to sustain populations of most organisms, and the organisms present will typically be transient, spending only a small proportion of time in the areas. Therefore, we conclude that these dose rates will not result in detriment to populations of non-human organisms.

We also consider the potential for harm to non-human biota from contact with high-activity particles during coastal erosion or human intrusion. We find that exposure to particles would not have significant impact on the overall health of individual organisms (and, therefore, would not have impacts on the health of populations).

Table 7.1: Summary of most exposed representative organism and associated total dose rates at maximum concentrations (µGy h^{-1}), from the assessment pathways considered during the PoA

Contaminant Pathway	Most exposed representative organism	Model Region	Total dose rate at maximum concentrations (µGy h^{-1})	Dominant radionuclide
Marine discharge	Phytoplankton	Local offshore	$3.8\ 10^{-2}$	Pu-239
Releases to groundwater	Bivalve moluscs	Lower Irt	$1.9\ 10^{-5}$	Sr-90
Historical contamination of Drigg Stream	Insect larvae	Drigg Stream	4.9	U-238*
Flooded soil	Lichen and bryophytes	Flooded soil	$1.8\ 10^{-1}$	Ra-226
Resident in Drigg Stream and flooded soil	Semi-aquatic mammal	Drigg Stream	$3.9\ 10^{-1}$	U-238
Gas: tritium	Representative organisms above the trenches	Trenches	$2.9\ 10^{-3}$	-
Gas: Rn-222	Small flying mammal	Trenches	$5.4\ 10^{-2}$	-
Gas: C-14	Representative organisms above trenches	Trenches	$1.5\ 10^{-1}$	-
Exposure to uncovered waste stacks	Representative organisms close to Vault 11	Vault 11 (when full) at 30 m distance	28	Co-60, Cs-137

* For the assessment of historical contamination in the stream, U-238 has been used as an analogue for total alpha activity [44].

Table 7.2: Summary of most exposed representative organism and associated total dose rates at maximum concentrations (µGy h^{-1}), from the assessment pathways considered post-PoA

Contaminant Pathway	Most exposed representative organism	Model Region	Total dose rate at maximum concentrations (µGy h^{-1})	Dominant radionuclide
Groundwater releases to coast/estuary	Small benthic crustaceans	Foreshore (beach)	$3.8\ 10^{-5}$	U-238
Groundwater to irrigated soil	Lichen and bryophytes	Irrigated soil	$2.6\ 10^{-3}$	U-238, U-234
Gas: Rn-222	Small burrowing mammals	Trenches	$6.0\ 10^{-3}$	-
Gas: C-14 (post-PoA)	Trees and shrubs	Vaults (geomembrane degradation)	$3.5\ 10^{-1}$	-
Coastal erosion: terrestrial[9]	Lichen and bryophytes	Cliff	**$1.3\ 10^{3}$**	**U-234, U-238**
Coastal erosion: marine	Phytoplankton	Foreshore	9.7	Th-228
Coastal erosion: resident in both terrestrial and marine ecosystem	Large marine mammals	Foreshore	2.6	U-238

[9] Additional representative organisms have total dose rates at peak concentrations more than 10 µGy h^{-1}.

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