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SAFETY REPORTS SERIES No. 126

The BIOMASS Methodology

Biosphere Modelling for Long Term Safety Assessments of Solid Radioactive Waste Disposal Facilities

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THE BIOMASS METHODOLOGY

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THE BIOMASS METHODOLOGY

BIOSPHERE MODELLING FOR LONG TERM SAFETY ASSESSMENTS OF SOLID RADIOACTIVE WASTE DISPOSAL FACILITIES

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FOREWORD

Radioactive waste can be generated by a wide range of activities, ranging from those in hospitals to those in nuclear power plants, mines and mineral processing facilities. Radioactive waste has the potential to present a radiological hazard to people and to the environment, and therefore has to be managed to reduce any associated risks to acceptable levels.

Assessments of post-closure safety form an integral part of the safety case for radioactive waste disposal and support the associated decision making process. A fundamental element of safety assessment is the assessment of the radiological impact on humans and the environment in terms of both potential radiation dose and radiation risks.

The IAEA has been organizing programmes of international model testing since the 1980s, including safety assessment models for evaluating the performance of waste disposal systems and quantifying potential radiological impacts on human health and the environment. This publication describes the work undertaken by Working Group 6, Biosphere Modelling for Long Term Safety Assessments of High Level Waste Disposal Facilities of the IAEA's Modelling and Data for Radiological Impact Assessments (MODARIA II) programme (2016–2019).

A methodology for addressing the biosphere in safety assessments for radioactive waste disposal was developed as part of the IAEA's Biosphere Modelling and Assessment (BIOMASS) coordinated research project and published in 2003. The BIOMASS methodology has subsequently proved useful and been widely referenced in safety assessments for a wide range of contexts, encompassing both near surface and geological disposal of radioactive waste.

This publication builds on the original BIOMASS methodology, providing updated information on addressing the biosphere in safety assessments for radioactive waste disposal, taking into account the experiences in using the BIOMASS methodology, together with other developments in scientific understanding and modelling capabilities. It is intended to provide information for organizations needing to undertake new safety assessments while retaining consistency with the ways that the biosphere is addressed in programmes that are at a more advanced stage of implementation.

The overall framework and workflow provided by the updated methodology are considered to be robust for application in a broad range of contexts. It is anticipated that the detailed information included in this publication will be adapted, as necessary, to suit each situation. While the methodological elements discussed in this publication largely supersede the original BIOMASS methodology, some of the supporting material in the original publication remains relevant, most notably the examples of generic example reference biosphere models; these are referenced herein, as appropriate.

The IAEA wishes to express its gratitude to all those who participated in Working Group 6 of the MODARIA II programme, in particular T. Lindborg (Sweden), G. Proehl (Germany) and R. Walke (United Kingdom). The IAEA officers responsible for this publication were J. Brown and A. Guskov of the Division of Radiation, Transport and Waste Safety.

CONTENTS

1. INTRODUCTION

The IAEA organized a programme from 2016 to 2019, entitled Modelling and Data for Radiological Impact Assessments (MODARIA II), which had the general aim of enhancing the capabilities of Member States to simulate radionuclide transfer in the environment and thereby assess the exposure levels of the public and of the environment to ensure an appropriate level of protection from the effects of ionizing radiation. The scope covered radionuclide releases and existing radionuclides in the environment.

MODARIA II was the latest in a series of IAEA programmes organized since the mid-1980s that have contributed to a general improvement in models, the transfer of data and the capabilities of those developing models in Member States. IAEA publications on this subject over the past three decades demonstrate the comprehensive nature of the programmes and record the associated advances that have been made. There were seven working groups in MODARIA II. This publication describes the work undertaken in Working Group 6 on Biosphere Modelling for Long Term Safety Assessments of High Level Waste Disposal Facilities.

The technical work contributing to this publication was also supported through a concurrent project of the international collaborative BIOPROTA forum. Partway through the MODARIA II programme, BIOPROTA produced an interim report on the development of the Biosphere Modelling and Assessment (BIOMASS) 2020 methodology within Working Group 6, published by the Swedish Nuclear Fuel and Waste Management Company (SKB) in 2018. Permission has been granted by SKB to use text directly from this interim report in this IAEA publication.

Funding and support for the BIOPROTA project from the following organizations is gratefully acknowledged: Agence Nationale pour la Gestion des Déchets Radioactifs (ANDRA), France; Bundesamt für Strahlenschutz (BfS), Germany; Direktoratet for Strålevern og Atomtryggleik (DSA), Norway; Eidgenössisches Nuklearsicherheitsinspektorat (ENSI), Switzerland; Federaal Agentschap voor Nucleaire Controle (FANC), Belgium; Nationale Genossenschaft für die Lagerung Radioaktiver Abfälle (Nagra), Switzerland; Nuclear Waste Management Organization (NWMO), Canada; Nuclear Waste Management Organization of Japan (NUMO), Japan; Nuclear Waste Services, United Kingdom, Posiva Oy, Finland; Radioactive Waste Management Ltd (RWM), United Kingdom; Strålsäkerhetsmyndigheten (SSM), Sweden; and Svensk Kärnbränslehantering AB (SKB), Sweden.

1.1. BACKGROUND

The safety principles to be applied in all activities for radioactive waste management are set out in the IAEA Safety Standards Series No. SF-1, Fundamental Safety Principles [1]. Safety requirements for the disposal of radioactive waste are provided in IAEA Safety Standards Series No. SSR-5, Disposal of Radioactive Waste [2], which sets out the safety objective and criteria for the protection of people and the environment against radiation risks arising from disposal facilities for radioactive waste, in operation and after closure. Recommendations on the safety case and safety assessment for disposal of radioactive waste are set out in IAEA Safety Standards Series No. SSG-23, The Safety Case and Safety Assessment for the Disposal of Radioactive Waste [3], which provides guidance on how to assess, demonstrate and document the safety of all types of radioactive waste disposal facility.

Assessments of post-closure safety form an integral part of the safety case for solid radioactive waste disposal and support the associated decision making process [3]. Safety assessment entails evaluating the performance of a disposal system and quantifying its potential radiological impact on human health and the environment. The fundamental element of the safety assessment is the assessment of the radiological impact on humans and the environment in terms of both radiation dose and radiation risks.

Depending on the site and the type of disposal facility, safety assessment usually includes models of the near field (including the types of waste, the engineered components of the disposal system and any zone disturbed by facility construction work), the far field (including the geology, hydrogeology, hydrology, geochemistry, tectonic and seismic conditions at the site) and the biosphere (the part of the environment normally inhabited by living organisms and including the atmosphere, topography, soils, water bodies, and the human population and biota in the proximity of the disposal facility). The extent and complexity of safety assessment models will vary with the type of facility and the site and will be related to the hazard potential of the waste.

Consistency between the different components of the post-closure safety assessment is essential, and procedures to achieve such consistency are a key part of the approaches and methodologies that are applied. For this reason, the

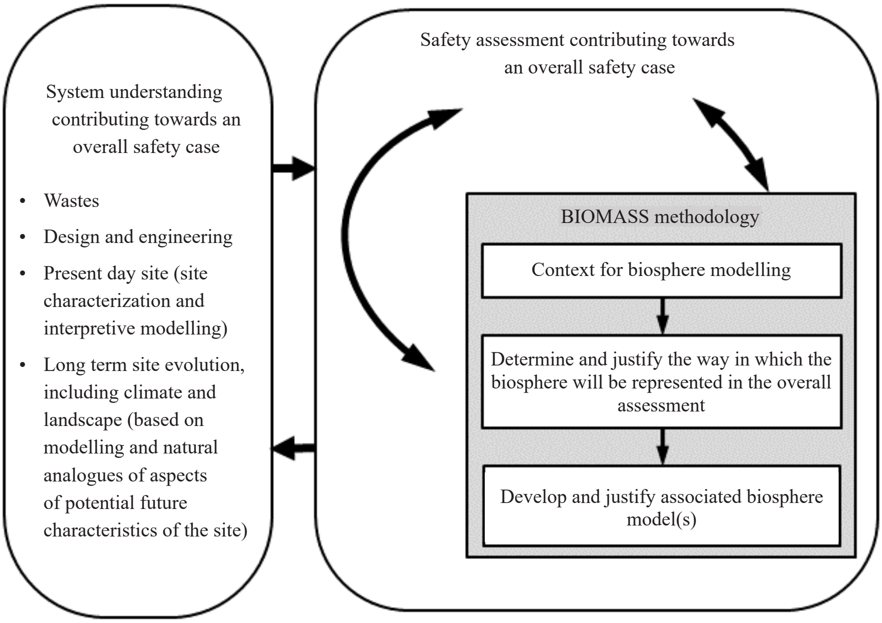

FIG. 1. Context of the BIOMASS methodology within the overall safety assessment and safety case (see Ref. [3]).

methodology used for assessing the biosphere[1] needs to be an integral part of that used for the overall safety assessment (see Fig. 1).

1.1.1. Collaborative studies and international experience

International guidance on biosphere modelling for radiological assessment was first developed at an international level within Phase II of the collaborative Biospheric Modelling and Validation Study (BIOMOVS II), which ran from 1991 to 1996 [4, 5]. The study included the Reference Biosphere Working Group. The working group concluded that standardized models are difficult to apply in practice, due to the need for biosphere models to reflect the specific context for an assessment. Instead of focusing on reference biosphere models, the working group developed a reference methodology to encourage transparency, consensus

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[1] Where this report uses phrases such as ‘assessing the biosphere’, ‘the biosphere part of safety assessments’, ‘representation of the biosphere’, ‘biosphere models’ or ‘biosphere modelling’, it needs to be understood that the authors are referring to modelling to assess radiological safety involving the evaluation of potential radiological impact on humans and the environment and the quantification of various end points, including potential doses and risks.

and harmonization in biosphere modelling. The BIOMOVS II study concluded that further refinement of the methodology was needed.

Theme 1 of the IAEA Biosphere Modelling and Assessment (BIOMASS) project, which ran from 1996 to 2001, further developed and refined the reference biosphere methodology set out in BIOMOVS II, drawing on experience of its application and developing several illustrative examples [6].

The European Commission's Modelling Sequential Biosphere Systems under Climate Change for Radioactive Waste Disposal (BIOCLIM) project (EURATOM 5th framework programme), which ran from 2000 to 2003, sought to build on BIOMASS and establish a practical methodology for assessing climate and environmental change [7]. BIOCLIM established long term projections for global climate and explored approaches for downscaling the global projections to regional and site scales. The project elaborated on the BIOMASS approach to addressing environmental change, with specific consideration of climate change and its implications for landscape development.

Following the BIOMASS project, the IAEA ran a programme called Environmental Modelling for Radiation Safety (EMRAS). The first phase of EMRAS, which ran from 2003 to 2007, included a working group on updating data for safety assessments [8], but did not review issues relating to the development and application of a biosphere component of a safety assessment methodology. Working Group 3 of the second phase of EMRAS, which ran from 2009 to 2011, considered reference models for waste disposal. That study focused on environmental change, noting that stylized models of sites analogous to future conditions at the site of interest and explicit modelling of the dynamic evolution of the biosphere could each have a role to play in building confidence in long term post-closure safety [9].

Most recently, Working Group 6 (WG6) of the first phase of the MODARIA project, which ran from 2012 to 2015, updated the BIOCLIM recommendations, particularly in terms of potential patterns of long term climate change at local, regional and global scales, and sought to develop a common framework for addressing climate change in post-closure safety assessments [9].

The BIOMASS methodology has been widely referenced in support of safety assessments for geological disposal of radioactive waste [10–14], as well as in support of assessments for near surface disposal, as discussed in Refs [15, 16]. This high level methodology has helped to provide consistency in the way in which the biosphere is characterized and assessed in different disposal programmes and in widely different assessment contexts. The BIOMASS methodology has been an important aspect of establishing an international consensus on how a post-closure safety assessment is conducted.

1.2. OBJECTIVE

The objective of this publication is to provide updated guidance for addressing the biosphere when undertaking post-closure safety assessments for solid radioactive waste disposal, building on the BIOMASS methodology.

Much experience has been gained since the original BIOMASS project, particularly as some disposal programmes have moved from the proof of concept stage to detailed analyses at specific sites. This includes a substantial amount of experience of undertaking safety assessments, including site specific stages of assessment and assimilation of understanding from site characterization studies and of integrating biosphere models in safety assessments and safety cases. There have also been further developments in approaches and recommendations for explicitly demonstrating environmental protection through assessments of radiological impacts on biota. Over the same period, the concept of the safety case has been established in international recommendations [3], emphasizing the need for integration of safety assessments in this broader framework. The MODARIA II project provided a timely opportunity to draw on experience gained since 2001, together with other developments in scientific understanding and modelling capabilities, to review and enhance the original BIOMASS methodology in collaboration with and with technical support from the BIOPROTA forum [17].

Guidance and recommendations provided here in relation to identified good practices represent expert opinion but are not made on the basis of a consensus of all Member States.

1.3. SCOPE

This publication enhances previous methodological guidance on modelling the biosphere when undertaking post-closure safety assessments and considering the long timescales of relevance to solid radioactive waste disposal [6]. It represents the output of WG6 from the MODARIA II project, addressing biosphere modelling for long term safety assessments of solid radioactive waste disposal facilities, working in collaboration with the international BIOPROTA forum, as discussed in Section 1.1. Some examples are presented that show how specific issues related to the BIOMASS methodology have been treated in national generic and site specific safety assessments. The examples are included to show ways to address issues in specific cases and not to be used as methodological templates. Rather, the examples can be seen as a stimulus to thinking when developing a programme of work for addressing the aspects of a safety assessment related to the biosphere and context specific plans. Also, in Annex I, published examples of biosphere modelling in support of safety assessments are provided for various specific contexts. These examples and the lessons learned were considered when this version of the BIOMASS methodology was developed.

This publication is intended to be suitable for use in various countries at different stages in the development of disposal options for a wide variety of different types of radioactive waste. It is recognized that individual countries have different regulatory regimes. The methodology has the flexibility to accommodate different types of assessment (from initial scoping studies through to inputs to safety cases submitted in support of licence applications) of various types of facility in diverse regulatory contexts. Therefore, this publication highlights the need for a broad approach with a workflow in which the methodology and steps included in the BIOMASS methodology interact with the wider safety case within the waste disposal programme.

1.3.1. Terminology

There is broad recognition of the usefulness of the original BIOMASS methodology, as evidenced by the range of assessments that reference it, as noted above. In this publication, the enhanced methodology is also referred to as the BIOMASS methodology, to emphasize that it is similar in structure to, and has been developed from, the original BIOMASS methodology.

The biosphere models adopted for safety assessment do not represent the actual conditions that will be present when a future release to the biosphere occurs. Rather, they are intended to be adequately representative of possible situations that need to be addressed for assessment purposes, ideally spanning the range of possible conditions and potential impacts on human health and the environment.

Although the BIOMASS methodology is primarily focused on radiological impacts on human health and the environment, it is also relevant to the possible impacts of non-radioactive materials released from a disposal facility. Therefore, in this publication the term ‘contaminants’ is used where either radionuclides or non-radioactive materials are of potential relevance.

To provide a consistent basis for calculations of potential radiological and other impacts arising from the presence of contaminants migrating from disposal facilities on long timescales, it is necessary to make assumptions about the biosphere system(s) of relevance. Biosphere models are calculational tools that represent these assumptions. Biosphere modelling is the process of calculating the potential impacts in the assumed biospheres by quantitatively evaluating those impacts using biosphere models.

Herein, where the term ‘biosphere’ is used, it means the biosphere adopted for assessment purposes, as described above.

As appropriate, protection of the environment is evaluated by assessing potential radiation dose rates to and/or chemical exposure of non-human biota. For simplicity, non-human biota dose assessment is referred to as ‘biota dose assessment’ in this publication, encompassing both types of impact.

In the broader context of post-closure safety assessments, reliance has often been placed on generic or project specific sets of features, events and processes (FEPs) that require consideration in the assessment [3, 18]. Sets of FEPs are often reported in the form of structured lists. The amount of detail provided on the individual entries depends upon the context in which the FEP list was created or elaborated.

Such FEP lists can help to demonstrate the comprehensiveness of the assessment process as well as keeping track of how possibly relevant issues have been addressed within an assessment. A generic FEP list can be used either to identify what may need to be included in a post-closure safety assessment or to audit an existing assessment to ensure that all relevant FEPs have been appropriately considered. The recording of how individual FEPs have been included or why they were excluded in a specific assessment is a useful part of knowledge management, for future reference generally and future phases of safety assessment in particular.

FEP lists or sections of FEP lists have been developed specifically relating to the biosphere and these can be used to inform biosphere identification, description and characterization, which are steps in both the original and this version of the BIOMASS methodology (discussed further below). In addition, it can be helpful to distinguish FEPs that are internal to the disposal system (including the engineered facility and the local geosphere and biosphere in which it is located[2]) from external features, events and processes (EFEPs) that operate outside the disposal system but influence it. Climate change, global sea level changes, Earth processes, such as volcanism and isostatic shoreline migration, as well as extraterrestrial processes such as meteorite impact, would typically be classed as EFEPs.

As discussed later in this publication, the main components of the biosphere (e.g. soils, water bodies, biota), together with their relevant characteristics, are typically thought of as features of the disposal system in which the disposal facility is located. Processes in the biosphere generally operate to modify the

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[2] The term ‘local’ includes both the zone in which the engineered facility is emplaced and zones through which and to which contaminants released from the engineered facility are projected to be mainly transported. Thus, a surface environment located some distance from the ‘footprint’ of the engineered facility would be considered local if it was determined to be a potential recipient of a significant amount or fraction of a release of contaminants from the engineered facility.

characteristics of those features, or to transport contaminants between them, for example by groundwater flow or fluvial erosion. Events are more difficult to characterize, but they can be viewed as processes operating on a relatively short timescale that significantly change the characteristics or arrangement of features of the disposal system, or substantially alter the processes that act upon it. Thus, a volcanic eruption, the advance of an ice sheet over a site, or an earthquake would be considered as an event.

1.4. STRUCTURE

Although the structure of the BIOMASS methodology is essentially iterative and is most appropriately shown in a cyclic form (Fig. 2), for the purpose of describing its components, it is convenient to adopt a linear representation (Fig. 3). Inputs to the methodology from closely associated sources of information, models and data are shown at the sides of Fig. 3.

The publication structure follows this linear version of the BIOMASS methodology:

- Definition of the assessment context is described in Section 3;
- Identification and description of the biosphere system(s) to be assessed are presented in Section 4;
- Development of biosphere models is described in Section 5;
- Application of the biosphere models and evaluation of results are discussed in Section 6.

Conclusions are presented in Section 7. References are provided at the end of the main text, as well as a glossary and a list of abbreviations.

A list of biosphere studies for a wide range of different contexts is included in Annex I, as a knowledge base for those tasked with undertaking safety assessments. Annex II summarizes experience gained in developing the guidance concerning identification and characterization of potentially exposed human and biota dose end points.

Further details on specific considerations and supporting information for the methodology are provided in the form of examples from national programmes in Annex III.

During the production of this publication, additional scientific papers were published in a special issue of the Journal of Environmental Radioactivity linked to the MODARIA II programme. The papers discuss specific aspects of the BIOMASS methodology as well as the use of supporting information in safety assessments. The lessons learned and experiences described in these papers formed part of the basis for the methodological enhancements described in this publication (e.g. Refs [19–21]).

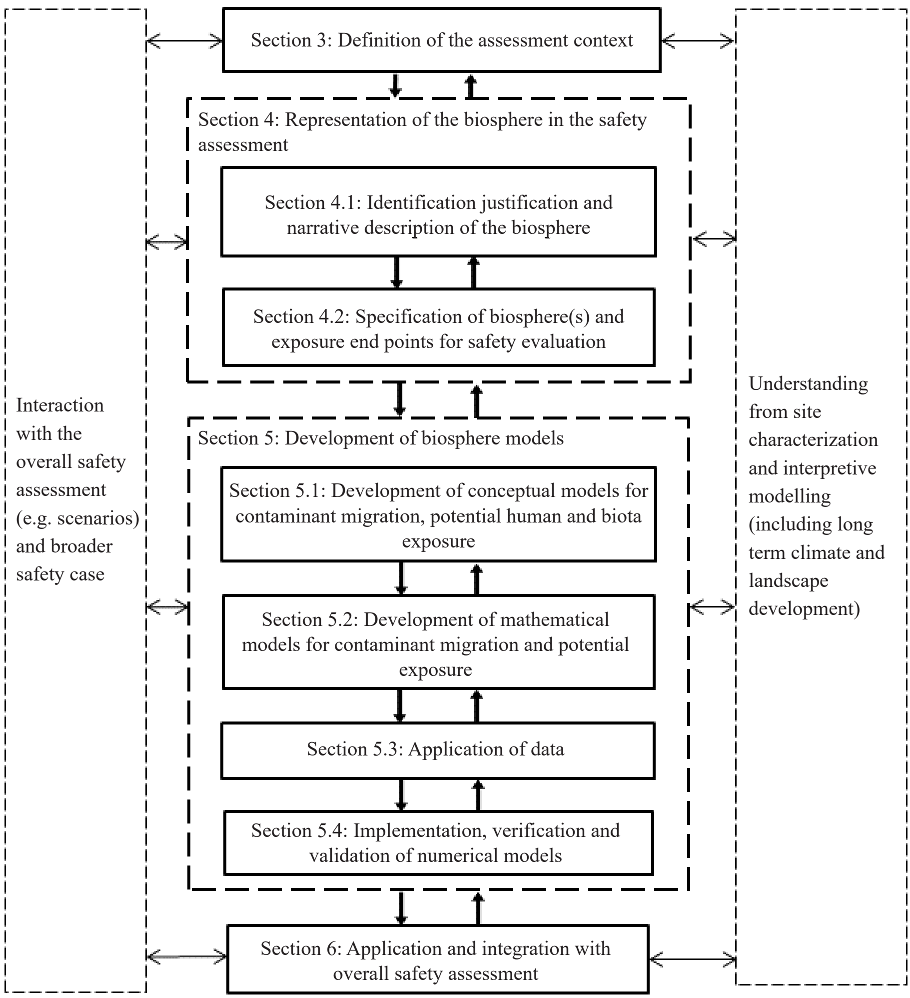

FIG. 2. Structure of the BIOMASS methodology presented herein (central column) with supporting information and interactions (shown at the sides).

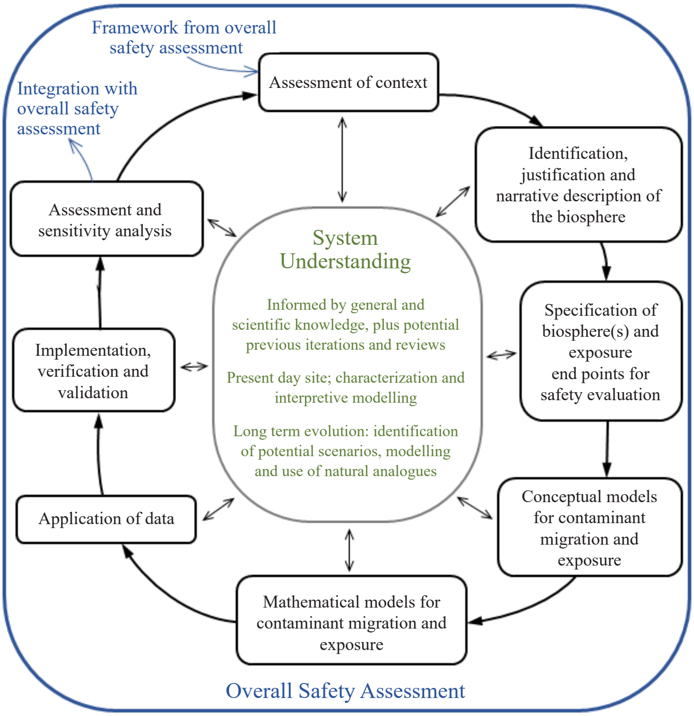

FIG. 3. Schematic illustration of the BIOMASS methodology (in black) showing integration with the overall safety assessment (in blue) and the central role that is played by understanding of the disposal system (in green).

2. ENHANCEMENT OF THE BIOMASS METHODOLOGY

There is a substantial consensus that the overall structure of the original BIOMASS methodology has proved sound [17]. The enhancements made to the BIOMASS methodology, shown in cyclical form in Fig. 2 to emphasize its iterative nature are, in overall terms is consistent with the original. Experience has shown that system understanding has a central role in safety assessments for radioactive waste disposal, especially in site specific contexts. This is illustrated, in Fig. 2 which also shows the iteration and feedback with system understanding. The integral role of the biosphere within the overall safety assessment is also emphasized. Lessons learned from ongoing national programmes and international developments have shown that a key to achieving a useful and robust safety assessment is to adopt a holistic strategy. The biosphere is seen as an integral part of the disposal system in which there are links between and interactions with engineered parts and the geosphere. This does not imply that the biosphere part of the system therefore includes safety functions relating to the repository, but rather that site understanding plays a role in the system understanding needed to support the safety case. The BIOMASS methodology described herein therefore emphasizes a strengthened interaction with both the system understanding and the broader overall safety assessment, as shown in Fig. 2.

Although the BIOMASS methodology is focused on the biosphere part of a safety assessment, experience shows that a more integrated approach is required in safety assessments if the approaches and methods used in BIOMASS are to be handled correctly and linked to the rest of the safety case. Also, understanding the processes related to disposal facility development, and acknowledging what, when and why this is needed at each programme step, are crucial when planning and undertaking a safety assessment.

To achieve the site understanding needed to make the right conceptual choices in the safety assessment, the biosphere part of the safety assessment cannot rely solely on data arising in other contexts. Site characterization and safety assessment tasks need constant interaction and cooperation, because the results of assessments inform requirements for further site characterization and accumulated knowledge from site characterization can alter the focus of the assessment, refining and enhancing the assessment approach. The enhancements made to the BIOMASS methodology give greater emphasis to this approach than did the original methodology.

Also of great relevance to BIOMASS are the recent substantial developments in long term climate modelling at global, regional and local scales. The ways to include and to assess future climate variants in the safety assessment have been further developed in the BIOMASS methodology. Given that future climate is a common issue for the whole disposal facility programme, it further emphasizes the importance of having an integrated and holistic approach, so that the overall safety assessment, as part of the safety case, has a common handling of long term processes and far future events.

Given the above, this publication inherits all relevant guidance from the previous BIOMASS methodology but complements this with new insights and approaches based on experience and lessons learned. This publication is therefore to be seen as a new, updated version of the BIOMASS methodology published in 2003 [6]. While the methodological elements within this publication largely supersede the publication Reference Biospheres for Solid Radioactive Waste Disposal, Report of BIOMASS Theme 1 of the BIOsphere Modelling and ASSessment Programme, some of the supporting material in the original document remains relevant, most notably the examples of generic example reference biosphere models; these are referenced herein, as appropriate.

Comprehensive documentation of the approach that is adopted in a specific assessment is a fundamental element of each stage of the implementation of the BIOMASS methodology. This facilitates scrutiny of the assessment, for example for auditing purposes or by the regulatory body, helps to ensure transparency and enables lessons learned in developing the assessment and interpreting the results to be used to revisit assumptions and decisions, either by iteration within the assessment cycle or at some subsequent time when the assessment requires updating. Such information can also be used to refine the assessment, perhaps by identifying particularly important FEPs or sensitive parameters.

A systematic methodology provides a ‘living’ approach for incorporating new information into safety assessments, taking account of experience from using the models and interpreting the results, changing assessment contexts, understanding new scientific information and evolving regulatory requirements and overall safety cases. Maintenance of comprehensive records for each assessment is of great importance where multiple assessment cycles are undertaken at different stages of development for a disposal facility (site selection, site characterization, construction, waste emplacement, closure and post-closure site management). Reference to these records facilitates the maintenance of continuity in the assessment process, while ensuring that the significance of moving to the next stage of development and of incorporating new information and understanding is properly evaluated through a comparison of assessment results.

3. DEFINITION OF THE ASSESSMENT CONTEXT

The surface environment is a very complex biogeochemical system. For post-closure safety assessments, the challenge is to develop simple, robust and readily scrutinized models that address the FEPs that are important for post-closure safety. Complex models can provide more detailed simulations of contaminant transport and accumulation in the biosphere, but these simulations can be subject to substantial uncertainty and might not contribute to the making of robust arguments for safety. Demonstrating the ‘fitness for purpose’ of biosphere models entails finding an acceptable balance between the level of detail and the defensibility of the approach that is adopted, considering the context in which it will be used. Finding this balance requires an understanding of why the assessment has been commissioned, what questions it is supposed to answer and the intended audience(s).

Such assessments apply to the long term future, which can only be characterized in terms of possibilities (alternative scenarios), and only in outline for each such possible future, each of which stands for or represents a multiplicity of possible alternatives, differing in further detail from each other. This lack of a detailed knowledge of potential future conditions could lead to the adoption of very simple models, for example by focusing on a single exposure pathway. However, simple models of this nature are potentially difficult to defend because they could omit important features, events and processes, or they could distort their significance, for example by adopting a spatial or temporal scale larger than that at which the process characteristically operates. Conversely, detailed understanding of present day biosphere systems can result in detailed models being extrapolated to very long timescales, which can be equally difficult to justify and defend. Understanding of the context within which the assessment takes place is an important factor in finding an adequate balance between simplicity and complexity in modelling approaches.

The assessment context defines the information that is needed at the start of each safety assessment iteration during the repository programme stages. These iterations can be seen as a stepwise increase in safety assessment site adaptation through input from ongoing site investigations and site modelling activities, as well as feedback from the safety assessment to further site characterizations or adjustments in repository design. The assessment context is also the appropriate place to document the bounding assumptions on issues that are likely to need consideration to undertake a safety assessment. These assumptions could be provided by those commissioning the assessment but, if they are not, need to be agreed at the start of the safety assessment iteration. These assumptions can involve strategic, technical and social value judgements, and wider consultation could be needed to determine their acceptability. Typically, the assessment context is revisited and updated for each programme stage with the associated resulting safety assessment. These context updates are then to be included and assessed in all programme processes (e.g. safety assessments, site characterization, design and construction).

The context for the biosphere part of the safety assessment is set by that for the overall safety assessment to which it contributes, differing mainly in the characterization of an appropriate source term (see below). Both the overall context and biosphere context answer fundamental questions about the post-closure safety assessment, namely, ‘what needs to be assessed?’ and ‘why does it need to be assessed?’ In a quantitative assessment, these become ‘what needs to be calculated?’ and ‘why does it need to be calculated?’ [6].

The components of the biosphere part of the assessment context are [19]:

- The purpose of the assessment, including the regulatory regime within which it is conducted, or which it is intended to inform;
- The end points of the assessment;
- The assessment philosophy (including management of uncertainty);
- The disposal system and the site context;
- The source term to the biosphere and associated interface;
- The time frames to be represented;
- The societal assumptions to be made.

These components integrate with the context for the overall safety assessment and are significantly driven by the regulatory regime (e.g. Refs [22–24]) and the nature of the overall safety case [3]. However, they can also be strongly driven by the interests of a wider range of stakeholders.

One component that is specific to the biosphere part of the safety assessments is that which defines the contaminant source term to the biosphere. In the biosphere context, the source term usually relates to the time dependent flux of contaminants entering the biosphere. If the biosphere is modelled as a distinct component of the overall disposal system, this also requires the definition of an interface between the biosphere and the rest of the system at which this flux can be specified.

In general, the assessment context will be defined specifically for each assessment cycle. It could be varied during an assessment or between assessments. Only minor changes would be expected during an assessment, as the context, including the stage of facility development, is largely fixed. However, substantial changes in assessment context can occur from one assessment to the next (e.g. from a generic to a site specific assessment). Nevertheless, the assessment context for each assessment will be strongly conditioned by the previously adopted approach to assessment, the results obtained and the reception of those results by stakeholders, including regulatory bodies. In defining the context for the next assessment, it is important to examine this historical legacy critically, to determine lessons learned, but also to challenge previous assumptions that may no longer be appropriate.

In Section 3.1, the components of the assessment context are discussed in more detail.

3.1. THE PURPOSE OF THE ASSESSMENT

The general purpose of a post-closure safety assessment as part of a safety case or in some other context, such as the development of a regulatory regime, is to support a decision concerning radioactive waste management.

In turn, this will generally involve having enough understanding of the disposal system to evaluate the potential impacts of releases of contaminants on human health and the environment through consideration of a range of situations that could apply in the future. In the context of the biosphere, there is a presumption that contaminants will, at some time, be released from the disposal facility and transported into the biosphere and that there is a need to assess their potential impacts quantitatively.

The purpose and method of conducting an assessment could vary from simple calculations to test initial ideas for disposal concepts, to complex analyses to support a disposal licence application requiring detailed site specific calculations with results compared against regulatory criteria and evaluated by the regulatory authorities.

The purposes of assessment include the following [3, 6, 25]:

— Inform development of a national radioactive waste management strategy;
— Support the development of new or modified regulatory requirements;
— Demonstrate compliance with regulatory requirements, for example in support of licence applications and/or as part of an associated regular review and update to safety cases;
— Contribute to regulatory review of an application for permission to proceed to the next stage of development of a disposal concept;
— Contribute to public confidence;
— Contribute to the confidence of policy makers and the scientific community;
— Provide a guide to research priorities;
— Test initial ideas for disposal concepts (site generic);
— Provide guidance in site selection and provide a basis for approval at later stages in the development of a disposal system;
— Facilitate optimization of the disposal system, for example in terms of design features.

3.2. END POINTS OF THE ASSESSMENT

The end points of an assessment will typically be measures of potential impacts on humans and the environment. The structure of a biosphere model will tend to reflect the results that it is designed to evaluate. These will depend largely on the criteria (regulatory or otherwise) that are adopted to judge the overall safety of the disposal system. However, while the overall safety objectives can be similar and correspond largely to those set out in Ref. [2], the level of detail prescribed in regulations on the nature of assessment end points, scenarios to be considered, etc., varies considerably (see the general discussion in Ref. [26] and a specific example from the United States of America in Refs [27, 28]). Several end points could have to be evaluated in accordance with the purpose of the assessment. Examples of measures of radiological impact and the circumstances in which they could be relevant are considered further in Ref. [6], including:

— Individual doses and risks to human health;
— Collective doses and risks to human health;
— Doses or dose rates to non-human biota;
— Modifications to the radiation environment, namely relating to the distribution of radionuclides derived from a disposal system in the environment and their concentrations;
— Fluxes of radionuclides into or through parts of the biosphere.

For any, or all, of the above, it may be necessary to estimate the variability and uncertainty associated with the selected end point.

More recently, the scope of assessments has broadened such that potential impacts on both human health and the environment from chemical releases associated with radioactive waste disposal could need to be assessed. Furthermore, consideration could need to be given to the protection of property and environmental quality through, for example, avoidance of pollution of groundwater bodies. Although the following focuses on radiological consequences, these other considerations could also need to be kept in mind.

It is generally accepted [29] that the effective dose received by a representative individual is an appropriate measure for estimating radiological risks in prospective assessments of planned exposure situations (including the disposal of solid radioactive wastes), for which limited information is generally available on the characteristics of the exposed individuals. However, even if the radiological end points of an assessment, with respect to human health, are expressed in terms of effective dose (or the detriment to health associated with that effective dose), there is considerable flexibility over how those effective doses could be evaluated, for example:

— Generally, the effective dose is evaluated for calculation cases based on each scenario defined in the overall safety assessment;
— In some cases, a probability of the dose occurring is also considered.

Effective doses can be calculated for several groups within a local population characterized by different patterns of behaviour. These groups are often termed potentially exposed groups (PEGs). Often, dose evaluation is for a representative individual of each PEG, typically called the 'representative person', as defined by the International Commission on Radiological Protection (ICRP) [30].

In principle, potential effects on human health could be evaluated for exposed populations, as well as for representative, more highly exposed, individuals (depending on the regulatory requirements). The highest dose to the representative person is usually the quantity to compare with dose constraints (in accordance with national regulations or IAEA requirements [2], for example).

Identification of PEGs and definition of their characteristics are usually made to be consistent with the biosphere system to be developed, but they can also be constrained by other considerations, such as the need to ensure that potential exposures to contaminants are not grossly underestimated or regulatory requirements. Similar considerations apply to assessments of exposure of populations of biota. It is noted that the choice of PEGs can both influence and be influenced by the representation of the biosphere adopted for the purposes of safety assessment. Societal assumptions, identification of PEGs and definition of their characteristics are discussed in Sections 3.7, 4.2.4 and 5.1.2.1, with further consideration in Annex II.

For post-closure safety evaluations, assessment of collective doses is not generally requested by national regulations, and no international standards have been set based on the limitation of collective dose. However, the exposure of larger groups of people (i.e. larger than the most highly exposed group) is considered to be a relevant factor by some countries (e.g. see Refs [31–33]). More generally, the ICRP has noted in the context of geological disposal that optimization of protection is the process to keep the likelihood of incurring exposures, the number of people exposed and the magnitude of their individual doses as low as reasonably achievable, taking into account economic and societal factors [34].

If collective doses are to be assessed as part of the comparison of options or as part of an optimization process, it can be informative to disaggregate the results, so that contributions in different relevant areas, over various relevant future periods, can be evaluated; see the discussion in Ref. [34]. Disaggregation over different ranges of individual dose rate could also be instructive.

Measures of potential impact on biota and the environment could be evaluated at individual, population, community, habitat or ecosystem levels. However, the population level is commonly accepted as the appropriate protection target for evaluating radiological impacts on biota and the environment. This gives rise to considerations of the range of the population of interest, which, in turn, has implications for the spatial extent of the biosphere representation to be adopted (see Section 4.1.2).

In assessment studies undertaken since the original BIOMASS methodology was developed, there has been increased recognition that assessments are not just about demonstrating compliance with regulatory criteria but are also undertaken to satisfy a variety of stakeholder interests. In this broader context, it has also been recognized that over and above impacts on humans and other biota, consideration could need to be given to the protection of environmental quality, for example through avoidance of pollution of groundwater bodies.

For non-radioactive contaminants, comparisons are generally made with standards and criteria for limiting concentrations in environmental media. These include drinking water criteria recommended by the World Health Organization (WHO) [35] and environmental quality standards implemented in various national regulations, such as those set by the European Union [36].

Consideration of potential doses and risks can be complemented by other types of safety indicators that can also draw on biosphere modelling. These might include, for example, comparison of environmental concentrations and calculated fluxes of contaminants against background levels. Alternative safety indicators are discussed, for example in Ref. [37], and will derive from the strategy being adopted in the broader safety assessment context.

3.3. ASSESSMENT PHILOSOPHY, INCLUDING MANAGEMENT OF UNCERTAINTY

The assessment philosophy (or approach) relates to the choices made in calculating the end points of the assessment. These can include restrictions on the pathways to be considered (e.g. see Ref. [22]), use of probabilistic or deterministic methods of calculation and the choice of parameter values in deterministic calculations or distributions in probabilistic calculations. The selection of specific scenarios or types of scenario requiring assessment might also be considered part of the assessment philosophy. The assessment philosophy is typically defined at a high level, for example in specifying the scope of a safety case and is propagated down to the overall safety assessment and hence the way in which the biosphere is represented.

Assessment of the safety of radioactive waste disposal facilities on timescales of thousands to millions of years is subject to profound uncertainties. Therefore, the management of uncertainty is a fundamental component of post-closure safety assessments and the selection of an appropriate approach is a key aspect of the assessment philosophy. The BIOMASS methodology draws on experience to provide guidance on identifying and managing uncertainties.

Specifically, the results of post-closure safety assessments are necessarily associated with explicitly estimated as well as residual unquantified uncertainties that need to be recognized when interpreting those results within the overall safety assessment and safety case (e.g. Refs [3, 26, 38, 39]).

Thus, the assessment philosophy needs to include consideration of how uncertainties will be identified, characterized and propagated through the assessment. For initial assessments, it could be enough to undertake a limited number of deterministic calculations to determine whether the proposed disposal system is potentially viable. However, in most assessments it will be necessary to quantify the robustness of the results obtained. This will involve identification, characterization and propagation through the assessment of those uncertainties with the potential to significantly affect the quantitative results obtained. As discussed herein, these uncertainties are associated with the range of scenarios that can be adopted as a basis for assessment, potential choices of alternative conceptual and mathematical models of the system to be represented, and both systematic (lack of knowledge related or epistemic uncertainty) and random variation (aleatory uncertainty) in the parameter values that will be used in the mathematical models. The assessment philosophy may include consideration of whether deterministic methods, probabilistic methods or a combination of both methods is used to explore these various uncertainties and demonstrate the robustness of the assessment.

In exploring uncertainties, one possibility is to adopt a cautious approach that is thought to overestimate potential impacts on humans and the environment. Caution can be shown in the choice of scenarios, in the models adopted and in the values of the parameters selected for use with those models. However, it is not always clear whether a specific assumption is cautious once it has been propagated through the assessment process (e.g. an increased distribution coefficient could show the slowing of the transport of a contaminant to the environmental medium of interest, but also the enhancing of its retention in that medium — whether these processes together will increase or decrease the concentration of that contaminant in the medium might not be readily determined without explicit calculations). Another consideration is that adopting multiple, modestly cautious assumptions could result in an overall assessment that is extremely cautious. Results from such an assessment, by exaggerating impacts, could distort choices between different sites or disposal facility designs, since it is unlikely to be possible to adopt the same level of caution for each such option. Thus, it can be helpful to maintain a deterministic ‘best estimate’ case alongside pessimistic and/or probabilistic analyses as a point of reference. This can be especially relevant if the assessment is intended to support optimization, particularly if the purpose is to support selection among options, for which realistic or best estimate assumptions may be preferred [40].

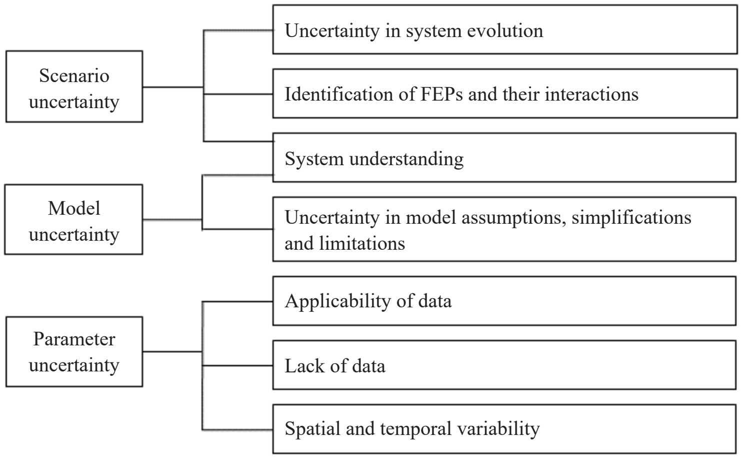

FIG. 4. Types of uncertainty and their main sources in post-closure safety assessment (adapted from Ref. [39]).

It is helpful to recognize the different types of uncertainty associated with post-closure safety assessments, because different approaches can be used to help to understand their potential importance. The types of uncertainty and associated approaches for managing them are summarized below and illustrated in Fig. 4, to provide a foundation for their subsequent discussion in the context of the BIOMASS methodology.

(a) Uncertainty about the broad future evolution of the system being modelled can be addressed by consideration of a range of different scenarios of possible future conditions (termed scenario uncertainty). Examples of uncertainties that are typically addressed through different scenarios include alternative climate sequences (e.g. covering different amounts and temporal patterns of future greenhouse gas emissions, alternative landscape evolutions and possible alternative future human activities and actions).

(b) Uncertainty about the way in which contaminant migration, accumulation and potential exposures are most appropriately modelled (termed model uncertainty) can be addressed through variant calculations/side calculations with differing modelling approaches.

(c) Uncertainty about the values of model parameters (termed parameter uncertainty) can be addressed through probabilistic calculations and/or variant deterministic calculations based on alternative combinations of parameter values. Probability distribution functions reflecting site conditions may be preferred to generic distributions, though care needs to be taken to ensure that they are not unduly constrained by present day site characteristics.

In some circumstances, specific assumptions (e.g. relating to human habits and behaviour) can form part of the assessment basis. Where specific parameters are prescribed (e.g. by regulation), they are not subject to uncertainty within the context of the safety assessment. However, prescribed parameters can still be subject to variability; for example, the consumption rate of a food can be prescribed as a probability density function over the exposed population. Care is often needed to distinguish uncertainty from variability appropriately in safety assessment calculations.

When addressing uncertainties in probabilistic analyses, good practice is to avoid risk dilution. Risk dilution arises where peak risks are uncertain in respect of their locations in time or space, but the approach to probabilistic analysis adopted uses an averaging procedure that does not take account of this, for example by averaging at a fixed location in each realization rather than averaging over the location of highest risk in each realization [41, 42]. Unduly broad distributions of parameter values can produce misleading results. In sensitivity analyses, it can be useful to define the distributions separately for uncertainty analyses and sensitivity analyses.

The difficulty of validation of models for post-closure safety assessments for radioactive waste disposal has long been recognized [43]. Natural systems are always open, and our knowledge is always incomplete and approximate, as discussed in the context of Earth sciences modelling in Ref. [44]. Therefore, safety assessment results are not presented as predictions, but as projections of possible futures, as discussed in Ref. [3]. Transparent recognition of the uncertainties, and a documented approach to addressing them, are important steps in building enough confidence in the results of a safety assessment to support decisions.

In some contexts, assessments can be based directly on characterization of present day surface environments. This approach was adopted by Swedish Nuclear Fuel and Waste Management Company (SKB) [12] and ANDRA [10] based on data derived from comprehensive site characterization programmes. Such a basis provides confidence that the data are coherent and therefore, that dependences between different aspects of the environment are taken into account. However, it does not necessarily result in consideration of the full range of situations that could arise.

Thus, in practice, safety assessments are likely to require the generation of a wide variety of results, so that highly cautious calculations are not given undue weight, but also to ensure that safety is not threatened by very unlikely situations that have not been addressed.

3.4. DISPOSAL SYSTEM AND SITE CONTEXT

As part of the assessment context, the disposal system, inclusive of the site context, needs to be outlined, to inform and set some boundary conditions for the safety assessment. Aspects that can be included are [2]:

- Depth of disposal, host geological medium and waste type;
- Spatial extent, surface topography, current climate, near surface lithology and soil types, plants and animals, local surface water bodies and near surface aquifers.

The amount of detail available depends on the stage of a disposal programme and the type of facility. A typology of the different types of facilities that could be developed to dispose of a wide range of radioactive waste types (from very low level waste to high level waste and spent nuclear fuel) in a wide range of geological contexts is described in Ref. [9]. Examples of safety assessments undertaken for a range of different types of radioactive waste and different types of disposal system and site context are provided in Annex I.

The spatial context of the site can be defined narrowly (e.g. just the area within the designated site boundary) or more widely (e.g. a region that includes a variety of types of land use potentially relevant to the future land use characteristics at the scale of the site and at the regional scale). The size and type of area to consider could be driven by locations of potential contaminant release, which can be outside the site boundary. A wider spatial range could also be needed to cover the area over which relevant human groups or biota populations exploit the landscape.

This range could be considerable in cold or arid climate conditions, in which primary productivity is low, but could also be large for humans in conditions of high productivity, but where a hunter–gatherer lifestyle means that the natural productivity is only exploited to a small degree. For humans, maximal use of local resources obtained from a small area is likely to occur with intensive agricultural use; associated assumptions regarding habits need to be reasonable and sustainable with respect to the size of the area considered, as well as human physiological requirements (e.g. see Refs [2, 34, 45]). A wider spatial range could also be needed if the area includes features (e.g. lakes and wetlands) that do not occur in potential contaminant discharge locations at the present day but could occur at potential discharge locations in the future. This has been illustrated in the programmes undertaken by Posiva [11] and SKB [12].

The issue of site context is closely tied to the way in which site characterization is related to biosphere system representation. The iterative nature of site characterization and its relation to post-closure safety assessment is discussed in Section 4.2.

3.5. SOURCE TERM AND GEOSPHERE–BIOSPHERE INTERFACE

The interface between the geosphere and the biosphere (GBI), that is the boundary or zone where contaminant concentrations or fluxes calculated in geosphere modelling are provided as inputs to biosphere modelling, is not an intrinsic characteristic of the disposal system. Rather, it is imposed as a convenient simplification in modelling because it facilitates handling of the deeper components of the disposal system separately from the often more dynamic and changeable near surface and surface components. However, in some disposal assessments biosphere modelling is integrated with that of the geosphere and near field. In such cases, the GBI is still present within the integrated assessment model (the assessment model is here defined as the model used to calculate dose in the biosphere) and the same degree of care is needed to ensure consistency across all components.

The location and extent of the GBI adopted in the assessment calculations can differ depending on the time frame being assessed. Furthermore, assumptions about the interface can differ for different contaminants and can vary depending on whether impacts on human health or the environment are being assessed. Considerations relating to defining and characterizing the GBI have been discussed elsewhere [16, 46] and these publications can be consulted for more detailed information. The GBI could be defined as a well, an interface between subsurface strata and surface soils, sediments or surface water (including streams, rivers, estuaries and seas), or a combination of these. Near surface strata could also be included within biosphere modelling, such that the interface could be defined as the boundary between bedrock and overlying unconsolidated strata (e.g. see Refs [12, 14, 47]). In general, the most convenient type of GBI to adopt will depend on the type of disposal concept, the site context and the approach to system modelling applied in the overall post-closure safety assessment.

In the context of the overall safety assessment, characterization of the source term relates mainly to consideration of the physicochemical form of the waste and the way in which it is conditioned and packaged, leading to a determination of rates of release from the engineered facility and the chemical speciation of those releases. In some contexts, there may be no projected releases of contaminants from the wastes themselves until they are contacted by groundwater penetrating the packaging. For spent fuel or high level waste (such as vitrified high level waste) that are contained within high integrity waste packages and a highly engineered facility, the delay before release from the waste spent fuel may be hundreds of thousands of years (e.g. see Refs [10–12]). In contrast, for other types of waste and for non-radioactive materials in the closed facility (including materials that form part of the engineered barrier system) releases to the geosphere may begin shortly after closure.

Inventories of contaminants in solid radioactive waste will typically include many radionuclides and non-radiological species. Not all the contaminants will need to be carried through to quantitative post-closure safety assessment. Some will be present in relatively small quantities, others could decay or degrade before potential releases can occur, and others could decay or degrade after potential release but before exposures can occur. Others could potentially reach the surface environment but could be shown to have negligible potential exposure consequences. Such contaminants can be screened out from the post-closure safety assessment, typically using simple, conservative side calculations. This allows effort to focus on assessment of contaminants of genuine interest from a post-closure safety perspective. Although short lived progeny can be screened out from the inventory, their subsequent ingrowth needs to be considered in transport and impact modelling. For example, ^{210}Po can be screened out from the inventory on the grounds of short half-life but will need to be addressed in the assessment because of ingrowth from ^{226}Ra (see also Section 5.2.4).

It is important that the basis for contaminant screening is clearly documented and justified. Examples for low and intermediate level radioactive waste include appendix A of Ref. [48] and examples for higher activities waste are included in Ref. [49].

Once releases from the engineered facility to the geosphere start to occur, release rates will be determined by the availability of contaminants in the wastes, the solubility of those contaminants and their sorption to near field materials and colloids. Release rates from the near field will also be determined by groundwater flow rates through the engineered facility or by diffusion where such groundwater flow rates are low. Similarly, transport through the geosphere will also depend on the solubility of contaminants, sorption on geosphere materials and colloids, groundwater flow rates through porous and fractured materials and by diffusion. Consideration needs also to be given to potential releases of volatile substances in gaseous form, for example ^{14}C incorporated into methane.

In terms of releases to the biosphere across the GBI, the source term can be expressed as a rate (e.g. Bq/a or Bq/m^2/a). Alternatively, the source term can be expressed as a concentration (e.g. Bq/m^3 in groundwater or Bq/kg in material being eroded into the biosphere). There is likely to be a conversion issue arising in this context, for example with a release rate per unit time being supplied from the geosphere model, but with a concentration needed by the biosphere model. It is important that when making such a conversion (in this example, by diluting the release rate, Bq/a, by a groundwater flow rate, m^3/a) that the conversion factor used is justified and consistent with the assumed flows in the geosphere modelling. This can involve detailed modelling of the near surface hydrogeological system. It can also be appropriate to make sure that other properties are consistent across the GBI (e.g. the chemical composition of, and the radionuclide speciation in, the discharging groundwater) when such properties are of relevance in biosphere modelling.

Contaminants can be released to the biosphere in liquid, gaseous or solid form. Contaminants dissolved in groundwater can emerge at the surface, mixing with recent meteoric water in the process, or be subject to well abstraction. Transport of colloids and precipitation/dissolution associated with the groundwater pathway can also be relevant. Volatile radionuclides and other substances, notably hydrogen, methane and carbon dioxide, can be released in gaseous form. Solid releases can arise through erosion into either a developing contaminant plume or into the disposal facility itself, or through human intrusion into the disposal facility or its immediate environs.

The spatial scale of releases to the biosphere can differ substantially depending upon the type of facility under consideration and its hydrogeological context. For example, releases from an intermediate level waste facility could be broadly distributed over its area, whereas releases from some types of spent fuel disposal facility, for example those proposed by Posiva [11] and SKB [12], based on high integrity copper canisters, could arise from only one or a few damaged disposal canisters. In fractured hard rock environments, either distributed or localized releases from a disposal facility could be captured by, and transported through, a small number of major fractures containing flowing groundwater. In turn, these fractures can be expressed as particular features in the landscape, for example fault bounded valleys. In contrast, in sedimentary sequences, releases could become widely dispersed in a local or regional aquifer prior to being subject to well abstraction or discharging to soils or surface water bodies such as rivers and lakes.

For releases over limited spatial extents, the main consideration in determining radiological and non-radiological impacts might not be contaminant concentrations in environmental media within the release area, but average concentrations in a wider area, determined by, for example, the resource requirements of the potentially exposed groups of humans and/or the characteristic exploitation ranges of potentially exposed populations of biota, or by surface water catchment areas. It is, therefore, important to avoid being unduly focused on the area in the immediate vicinity of contaminant releases when considering the biosphere.

3.6. TIME FRAMES

The time frame for the safety assessment is an important component of the context, informing, for example, the degree to which environmental change may need to be considered. The time frame will be determined by the timescale of the contaminant source term to the biosphere (see Section 3.5) and other factors, including regulatory guidance.

A distinction can be made between the time that elapses before contaminants begin to be released to the surface environment in significant amounts and the duration of the period over which such releases occur. For safety assessments, an important focus is on the period over which releases occur. However, in the overall assessment context, both the time elapsed before releases start to occur and the duration of those releases are relevant, as the climate and landscape could change over both these periods, with potentially significant effects on releases of contaminants from the near field and through the geosphere. Projections of long term changes in climate and an understanding of the past and present day characteristics of a site, as determined from site characterization, can be used to inform the development of definitions of future biosphere(s).

The time dependence of projected releases of contaminants to the surface environment needs to be considered in conjunction with the timescales over which those contaminants are retained in the local environmental media. If the timescale of releases is long compared with characteristic timescales of residence in those media, it may be appropriate to estimate equilibrium concentrations in the biosphere model and apply these in assessing impacts. In contrast, if the timescale of releases is short, if the characteristic timescales of residence in environmental media is long, or if the characteristics of the surface environment are projected to change rapidly, an equilibrium situation may never be achieved. In these circumstances, environmental concentrations and impacts could need to be calculated using a time dependent biosphere model.

Changes in the characteristics of the surface environment in the near future may be relatively well understood, even though, even on timescales of a few years, human actions can impose significant changes (e.g. in land use); see also Section 3.7. However, at longer times into the future, greater uncertainties arise. Also, whereas changes could be gradual for an extended period, an event could occur, such as the advance of an ice sheet across the site, that completely resets the characteristics of the surface environment, requiring that an alternative representation of the biosphere needs to be defined for times later than that event. Overall, the increase in uncertainties with time means that the quantitative results obtained from assessments based on present day characteristics become increasingly unreliable and this needs to be taken into account in their interpretation, bearing in mind that the intent is not to predict the future, but to provide a reasonable assurance of safety. Alternative measures of safety can be employed on different time frames (see e.g. Ref. [37]).

3.7. SOCIETAL ASSUMPTIONS

Assessment timescales beyond even a few tens, or at most hundreds, of years introduce profound uncertainty into any description of human behaviour. This means that the biosphere or biospheres adopted for assessing system safety, in which human behaviour is an integral part, can only be considered as illustrative; that is, as providing indicators of the potential impact of the disposal facility.

As there is little technical basis for predicting the nature of future human behaviour, it is necessary to make assumptions to quantify potential impacts on human health and the environment (recognizing that human behaviour is conditioned by, and significantly affects, the environment). Typically, past and present day behaviour and levels of technology at the site and/or other locations are assumed. This leaves open the potential to assume a wide variety of patterns of human behaviour, with attention often focused on local habits and those patterns of behaviour that are likely to reasonably maximize potential exposure to contaminants from a disposal facility, for example through maximum reasonable use of local resources. The use of stylized approaches could become more important for longer timescales [34].

4. REPRESENTATION OF THE BIOSPHERE

Drawing on the assessment context, the next step of the methodology is to identify the set of biosphere systems that are needed to support the safety assessment. This set needs to be sufficient to support the overall assessment. In this section the initial focus is on the landscape level of the biosphere, but biosphere systems ultimately need to be identified at a level that is appropriate for assessment modelling. (See Section 5; this will typically be at the scale of an ecosystem, a discharge area, or some other unit that captures areas with the highest potential concentrations of contaminants in the landscape, e.g. an arable area irrigated with potentially contaminated water.)

The biosphere will need to be represented in each scenario being addressed in the overall safety assessment. The identification of biosphere systems to be represented for each scenario will be closely related to the identification and justification of those scenarios themselves. In particular, both can take climate and environmental change into account, drawing on Ref. [9], for example. EFEPs, such as those associated with global changes in climate, Earth processes and human activities, are typically used as drivers for reference and alternative scenarios to be addressed in the overall safety assessment (i.e. alternative realizations of the EFEPs, e.g. different sequences and durations of climate states) could result in the adoption of different scenarios or calculation cases. The biosphere(s) identified for assessment need to be consistent with, and fulfil the requirements of, the overall assessment scenarios and calculation cases.

4.1. IDENTIFICATION, JUSTIFICATION AND NARRATIVE OF THE BIOSPHERE

This Section describes how the biosphere systems are firstly identified and justified and then characterized through the development of a qualitative and high level narrative. A narrative provides a written account of the development of the biosphere over time, focusing on those aspects that are relevant to the assessment. The narrative provides the basis for a more detailed quantitative description (Section 4.2), that is sufficient to support the assessment modelling (Section 5). Although presented as a linear workflow, in practice, iteration will be needed between these steps, as the biosphere systems to be carried through to the consequence analysis are identified and refined.

The term biosphere system is used to mean surface systems of particular interest for contaminant retention and potential exposure. These systems need to be identified, justified and described sufficiently to support biosphere model development. This can typically only be done in the context of the overall assessment scenarios and the associated assumptions regarding climate and environmental change, and in the context of a larger spatial scale. The IAEA provides general definitions and guidelines on characterization of the disposal system in support of a safety case [3] and other literature provides further discussion of the need to integrate the biosphere, geosphere and designed parts of the disposal system in the overall safety case [50].

4.1.1. Identification and justification

Different scenarios and calculation cases for overall disposal system evolution could lead to requirements for distinct biosphere representations. However, even within one scenario various alternative approaches can be adopted for the definition of biosphere systems. In the current context, the emphasis is on the development of biosphere system definitions suitable for assessing potential effects on human health and the environment due to releases of contaminants from a disposal facility for solid radioactive wastes. At a broader scale, these biosphere representations are usually compatible with the changing climate and landscape projected to apply to the region of the disposal facility under the scenario being investigated.

Figure 5 shows a decision tree for identifying the approach to represent the biosphere appropriate for a specific scenario or calculation case under a specific regulatory regime. In some assessment contexts, the biosphere system(s) can be prescribed by legislation or guidance (Option 1 in Fig. 5), for example present day conditions are to be assumed over a prescribed quantitative assessment period [27, 28].[3] In such a context, all that is needed is to describe the prescribed biosphere system(s) (e.g. a well, a surface water body such as a river or lake, or surface soils). However, this is not necessarily a trivial exercise. For example, a large amount of site specific information could need to be collected and analysed to ensure that present day conditions are described in enough detail for assessment purposes.

If the biosphere system is not prescribed, the principal characteristics of the biosphere need to be identified. At this stage, a detailed description is not needed; it is sufficient to list the principal characteristics of the biosphere at the landscape scale (here 'landscape' is defined as the larger surroundings, containing landforms and ecosystems including and connected to the area of interest to assess dose) and at the scale of each individual biosphere system of interest. These characteristics will form the basis of a subsequent narrative description (Section 4.1.2), which, in turn, will underpin a detailed description sufficient to support quantitative modelling of individual biosphere systems (Section 4.2). The principal characteristics have been identified [6, 19] as:

— Climate and atmospheric composition;
— Geographical location and extent, taking account of discharge locations and areas in the landscape that have potential for accumulation of contaminants;
— Topography;
— Near surface lithology, covering the structure and composition of soils, sediments and weathered material overlying the bedrock;

[3] See US Code of Federal Regulations 10 CFR 63.312, where in relation to a proposed repository at Yucca Mountain it is required that the reasonably maximally exposed individual has a diet and living style representative of the people who now reside in the Town of Amargosa Valley, Nevada. Also, at 10 CFR 63.332 it is required that the exposure pathways to that reasonably maximally exposed individual are those associated with groundwater abstraction at a specified rate.

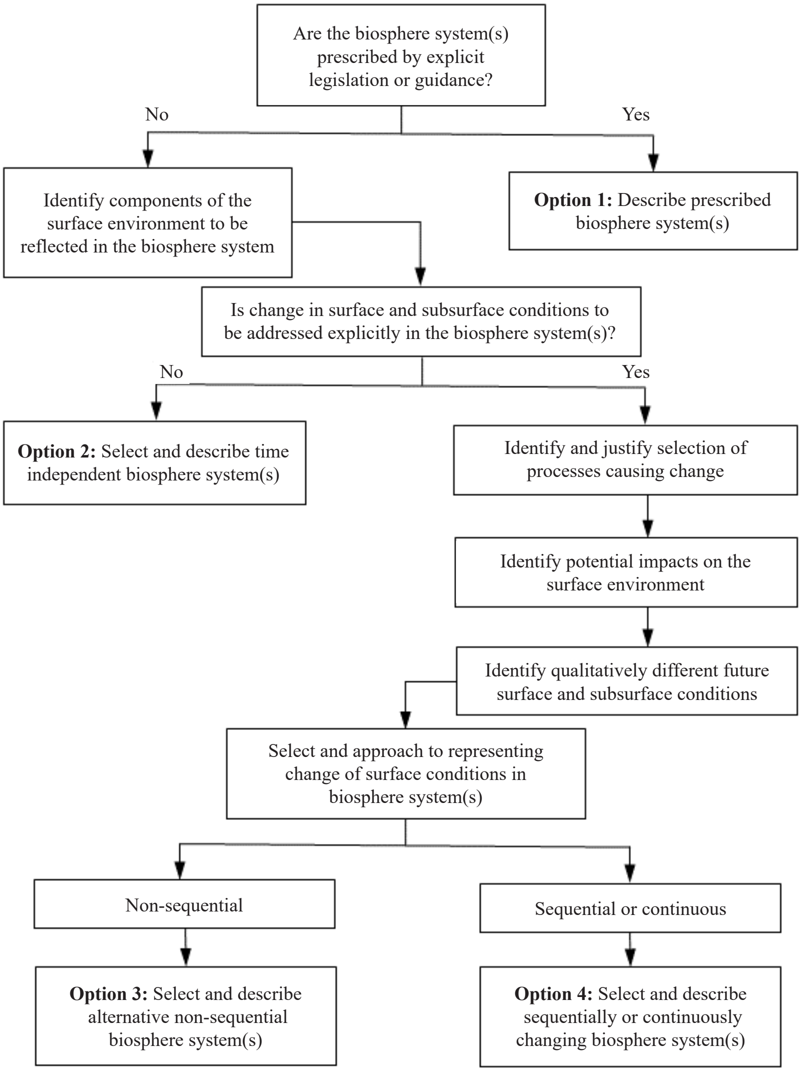

FIG. 5. Decision tree for the identifying the approach to be used for representing biosphere system(s).

— Subsurface and surface water bodies, which may include near surface aquifers, hydrology, hydrogeology, ephemeral surface water bodies in dry areas, and ice caps or ice sheets in arctic, near arctic or mountain contexts;
— Terrestrial and aquatic biota present within the biosphere system(s);
— Human communities within the biosphere system, including their behaviour and land use.

It could be appropriate to use information from site characterization to describe these principal characteristics of the biosphere at the present day and to use climate and landscape modelling (undertaken in the context of the overall assessment) to project how these characteristics can change with time (e.g. Ref. [51]). If biosphere system change is not to be addressed — for example, if the effects on the landscape of climate change due to natural processes and human activities can be taken as limited — then Option 2 in Fig. 5 applies and all that is needed is to describe a time independent biosphere system or systems. More than one such time independent system could be necessary if, for example, various alternative potential groundwater discharge or well abstraction locations are identified.

However, if biosphere change is to be addressed, then consideration needs to be given to the processes causing any changes and their potential impacts on the landscape and biosphere system, with a view to identifying qualitatively different patterns of change that need to be assessed (Option 3 or Option 4 in Fig. 5). Minor changes in biosphere characteristics can typically be addressed through parameter value variation within a single conceptual and mathematical model of a particular biosphere system; that is, they do not constitute qualitatively different patterns of change.

In identifying the causes of changes to the biosphere, it is likely to be helpful to distinguish between those that are internal to the disposal system and those that are external to it. External causes (i.e. EFEPs) could arise as a result of natural processes or human actions and could include, for example, changes in land use, climate change (which may be either natural or human induced) and Earth processes (e.g. isostatic movements of the crust). Internal causes could include, for example, lake sedimentation or development of an alkaline plume from concrete structures in the disposal facility.

All external causes are closely linked, with:

(a) Human activities affecting climate;
(b) Climate affecting Earth processes, for example through changes in ice and water loadings on the continents and ocean basins;
(c) Earth processes affecting land use, for example through changes in shoreline position for a disposal facility located in a near coastal context.

Erosive processes driven by climate and climate change and human land use can also have substantial effects on near surface lithology and hydrology.

The EFEPs can affect the internal characteristics of the disposal system in a wide variety of ways, depending on the type of disposal concept (including location and depth). How EFEPs cause changes in the geosphere and biosphere of relevance to the BIOMASS methodology is illustrated in Fig. 6 (see also

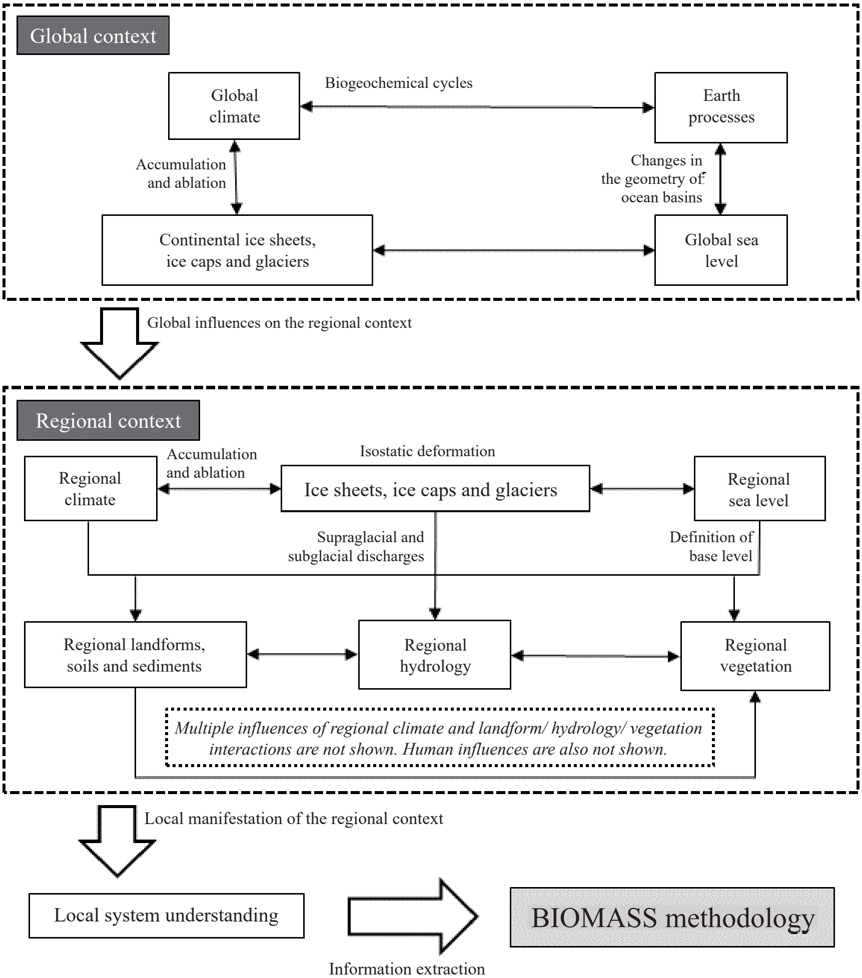

FIG. 6. Role of EFEPs in informing the BIOMASS methodology.

Ref. [9]). Figure 6 is a generic illustration and the importance of the specific EFEPs will vary between different contexts, particularly at the regional scale.

In Fig. 6, key EFEPs are identified as operating at a global scale. Human activities, notably those that give rise to greenhouse gases and particulate aerosol emissions, affect the global climate. The global climate is also affected by Earth processes, notably those that affect the global carbon cycle. These processes operate on timescales ranging from sub-annual (atmosphere-ocean interactions and volcanism) up to more than 100 000 years (rock weathering).

The global climate is a primary control on the growth and decay of continental ice sheets, valley glaciers and ice caps. In turn, the water incorporated in or released from these various ice masses contributes to changes in sea level.

Global climate and eustatic changes in sea level are major factors influencing climate and landscape at a regional scale (typically a few hundred kilometres in extent). As observed at the global scale, the regional climate influences the growth and decay of ice sheets but, at this scale, there is an emphasis on the passage of such ice sheets across the landscape, with associated hydrogeological and erosive effects. In addition, regional ice sheets load and deform the crust of Earth, leading to isostatic depression of the crust beneath the ice and to uplift beyond the ice margin. This means that, at a regional scale, sea levels are determined by the combined effects of global eustatic and regional isostatic changes.

Regional sea level also defines the base level to which major rivers grade (though inland streams can grade to higher base levels, depending on the topography of the landscape and the resistance of different rock types to erosion). In unglaciated conditions, the development of regional landforms is determined by the climate and by base levels. Fluvial processes are typically the main control on landform development, but aeolian processes can dominate in arid conditions. The passage of ice sheets or glaciers across the landscape can have a major influence on erosion and deposition, but also on regional surface water and groundwater hydrology. Irrespective of the presence of ice sheets or glaciers, regional landforms, soils and sediments are closely coupled to regional hydrology and vegetation. The nature of the vegetation cover, for example, influences resistance to erosion, and is a major control on the water balance through the effects of transpiration, as well as contributing to the development of mature soil profiles. Conversely, the topography of the landscape and the characteristics of the soils influence the type of vegetation that may be present. Not shown in Fig. 6 is the consideration that human activities can also profoundly affect hydrological and vegetation characteristics at a regional scale.

The developing regional climate and landscape provides the context within which a description of the local environment can be developed at the site scale (typically a few kilometres in extent). Once such a description has been developed, it can be used to specify the context in which a conceptual model of the biosphere to be used for assessment purposes is to be developed, as described in Sections 3.4 and 5.1, respectively. Thus, EFEPs are identified at the global and regional scales, propagated into the description of the local environment and provide the context in which the FEPs associated with the biosphere model are identified and characterized.

Although the surface environment will change continuously with time, it might not be necessary to model the process of change in assessments. Rather, a set of calculations for time independent biosphere systems could be adequate to determine the range of potential impacts on human health and the environment (Option 2 in Fig. 5). This would be the case, for example, if the key contaminants had short residence times in the biosphere system compared with the timescales over which the biosphere system was projected to change significantly. This does not mean that the surface systems to be modelled would be those of the present day. Substantial changes in the surface system could occur before the release of contaminants began. The key consideration would be whether the released contaminants had short residence times in the then existing surface system compared with its persistence. If releases could occur after different periods of delay, or if there was uncertainty as to what type of surface system might be present after a specific period of delay, it might be appropriate to model several different types of biosphere system, but without making any judgement as to their order of occurrence or as to their probability of occurrence.

These various considerations suggest two broad approaches by which surface environmental change might be represented in safety assessments. In Option 3 (Fig. 5), it is determined that the surface environment has only a limited 'memory' of the time history of previous discharges of contaminants. Thus, calculations can be made for individual and different biosphere system developmental states separately, with no need to evaluate the durations of those states or the order in/probability with which they occur (e.g. see Refs [10, 14]).

Where contaminants are anticipated to persist in the surface environment on timescales comparable with or longer than those over which significant changes in that environment occur, then Option 4 (Fig. 5) is likely to be applicable. Option 4 treats the biosphere system either as a sequence of time independent states with transitional periods (which can be of extended duration) between them (as was done in Ref. [7]) or as a continuously changing system. The sequential approach can be useful where generalized narratives of expected patterns of future environmental change are available, but there is no fully quantitative model of such change. In this case, the narrative can be used to construct descriptions of the individual biosphere states (e.g. based on analogue data from various locations), and the characteristics of transitions can be inferred by interpolation between the before and after end states. However, it can be difficult to justify the interpolated characteristics in the absence of a quantitative model.

Where a quantitative model of climate and landscape change is available, as is typically the case in later iterations of site specific assessments, this can be used to underpin a quantitative model of biosphere system change. Thus, a time dependent model for contaminant transport and accumulation in the biosphere can be coupled to a time dependent model for contaminant release from the near field and transport through the geosphere. This approach can be preferred when enough data are available to construct a time dependent model of the overall disposal system, and where key contaminants are retained in the various components of the system on timescales comparable with, or longer than, the timescales over which those components change substantially. Multiple realizations of the model could be needed to represent various possibilities for the time dependence of the overall disposal system.

The sequential and continuous approaches to representing surface environmental change are included under a single option to emphasize that mixed or intermediate representations can sometimes be appropriate. Thus, for example, a continuous description of larger scale changes (e.g. shifts in the position of the shoreline) can be coupled with snapshot descriptions of various aspects of the landscape that arise because of such changes (e.g. the types of lake that can be present). In some contexts, continuous and non-sequential approaches can be combined (e.g. a continuous model of landscape change can be adopted), but with, non-sequential snapshot modelling of the biosphere system used to assess the impacts of releases of contaminants at different stages of landscape development.

Whichever of the options is adopted, the aim is to produce narratives of changing characteristics of the surface environment and how they are to be considered in assessment studies. These narratives (incorporating both text descriptions and diagrams) provide a context in which biosphere system descriptions are developed and those descriptions are transformed into conceptual and mathematical models. These matters are addressed in Sections 4.2, 5.1 and 5.2, respectively.

Information on consideration of climate and environmental change in safety assessments for solid radioactive waste disposal facilities is provided in Ref. [9]. Recent global climate modelling covering the period from the present day to a million years in the future is described in Ref. [52].

4.1.2. Biosphere narrative

Section 4.1.1 sets out the approach to be used in defining the characteristics of the biosphere and the biosphere system(s) that are to be described and modelled for assessment purposes. In the context of biosphere identification, the biosphere and its systems need to be described in broad narrative/qualitative terms (a more detailed description is provided in Section 4.2).

4.1.2.1. Climate and atmospheric composition

Climate could be described in terms of the Køppen-Trewartha scheme [53] or else this scheme could be used to define a set of climate stations with a range of climates that could be characteristic of the adopted biosphere under consideration. Alternatively, quantitative climate modelling could be used to characterize the range of climatic conditions that could occur for the selected biosphere. The climatic conditions might be characterized for a time independent state (e.g. described as glacial, periglacial or interglacial), with the effect of any potential changes in the characteristics at any time during the existence of that state included in the uncertainties, or changes in climate with time during that state might be represented explicitly. The approach adopted would depend on which option for assessment was adopted from the decision tree illustrated in Fig. 5.

Atmospheric composition might be described only in terms of concentrations of greenhouse gases and aerosols, with consideration being given to ensuring that these concentrations are consistent with the climatic conditions associated with the biosphere. Other potentially relevant characteristics of the atmosphere (e.g. those characterizing relative humidity and wind field) would be included in the description of climate.

In general, the principal characteristics of climate of relevance would be seasonal temperatures and precipitation, but derived variables, such as soil moisture content, potential evapotranspiration and runoff (precipitation minus actual evapotranspiration), might also be described. For some disposal contexts, individual extreme events might be more relevant, for example high precipitation storms influencing erosion and infiltration in arid regions.

4.1.2.2. Geographical extent and location

These are primarily of relevance in defining variations in aspects of the biosphere system(s) to be described. For overall assessment purposes, an area representing the landscape above the disposal facility might need to be defined, since this would inform the boundary conditions relevant to characterizing the hydrogeology and hydrogeochemistry of the disposal system and the potential discharge locations. For the biosphere part of the safety assessment, smaller areas corresponding to individual biosphere systems might be defined. However, these areas might not be located immediately above the disposal facility (except for inadvertent human intrusion, erosion or gas release scenarios in some contexts). Instead, they might be at some distance from the disposal facility in an area or areas where contaminants might discharge into the accessible near surface environment or form a plume in a water body that could potentially be used for groundwater abstraction (an aquifer). For the narrative description, the extent and location need only be described in qualitative terms; further detail will subsequently be developed to support a more quantitative description, as described in Section 4.2.

4.1.2.3. Topography

This influences factors such as land use, settlement patterns and the nature of the drainage network. It can also affect local climatic conditions (e.g. through the influence of altitude and aspect). Once the geographical extent and location of the biosphere have been defined, it could be relatively straightforward to develop a quantitative description of the topography; for example, a digital elevation model (DEM) can be developed for the present day system and geomorphological models can be used to inform how that DEM would change with time due to aeolian, fluvial and coastal processes. This type of supporting work would inform the detailed biosphere description, as discussed in Section 4.2.2.

4.1.2.4. Near surface lithology

As with topography, the lithology of the soils and sediments of the surface environment is likely to be altered from its present day characteristics by erosion, deposition, changing climate and human actions (e.g. agriculture). For a specific site, it can be studied in detail during site characterization by both invasive (e.g. borehole) and non-invasive (e.g. ground penetrating radar, seismic survey) techniques. As with the topography, a model describing the lithology of a site and its evolution may support the biosphere characterization and quantitative description, as discussed in Section 4.2.2.

Both topography and lithology will need to be characterized by reference to the broader context of the overall disposal system, since topographical features can be controlled by underlying geological structures, such as faults, and the composition of soils, sediments and weathered rock will often be closely related to that of the underlying geology (though not for example, where the superficial deposits have been transported from elsewhere by ice sheets and laid down during the subsequent retreat of the ice).

4.1.2.5. Subsurface and surface water bodies

Surface water bodies can be described in general terms, for example the overall drainage density likely to be present in the landscape and the potential locations of lakes. Similarly, subsurface water bodies can be described in terms of those strata in which aquifers are likely to be present. These descriptions will rely heavily on the characteristics of the climate, topography and lithology, which can change with time, as discussed above. Sea level will also be an important consideration in coastal, near coastal and low lying topographical contexts.

4.1.2.6. Human communities and terrestrial and aquatic biota

The biota potentially present within the biosphere system(s) will be determined, in part, by the climatic conditions and by the topography and lithology of the landscape. However, they will often also be strongly determined by land use and human community aspects. Thus, it is appropriate to consider these two aspects together within the narrative description. In turn, this description of human communities and terrestrial and aquatic biota will constrain the types of land use, partially determine the types of biota likely to be present and provide the basis for the subsequent choice of PEGs and, if necessary, PEPs to be adopted in the model (see Section 4.2.4).

4.2. BIOSPHERE DESCRIPTION TO SUPPORT THE SAFETY ASSESSMENT

Narratives of the biosphere system(s) represented in the post-closure safety assessment need to be developed into quantitative descriptions sufficient to provide a foundation for building conceptual and mathematical models. These descriptions can be of time independent states, the transitions between those states, or time dependent biosphere systems, consistent with the approach determined via Section 4.1.2.

The following subsections provide guidance on developing an understanding of interactions within the biosphere (Section 4.2.1), drawing on site characterization, as appropriate, and supportive modelling (Section 4.2.2). There are two interrelated and important aspects of biosphere system descriptions that provide a foundation for the subsequent development of conceptual and mathematical models for contaminant migration and exposure. These are the area of relevance for the biosphere modelling and the choice of PEGs and PEPs that are needed as end points. These aspects are given further consideration in Sections 4.2.3 and 4.2.4.

4.2.1. Understanding biosphere interactions

Tools including interaction matrices and process influence diagrams may be used to help structure, formalize and communicate biosphere system descriptions in a way that facilitates their conversion into a conceptual and then a mathematical model of the system (e.g. see Refs [54, 55]). Some degree of disaggregation of the principal components is likely to be necessary to provide a framework to subsequently develop understanding of the potential for contaminant migration and exposure.

	1	2	3
1	**Soils**	Element supply Habitat supply Relocation	Food supply Habitat supply Material supply
2	Bioturbation Death	**Plants**	Food supply Material supply Stimulation/inhibition
3	Death Material use Relocation	Consumption Material use Species introduction/ extermination Stimulation/inhibition	**Humans**

FIG. 7. Illustrative phenomenological interaction matrix supporting understanding of the biosphere system, based on information in Ref. [56]. Arrows illustrate the clockwise notation of the interactions.

An illustration is provided in Fig. 7 (based on information in Ref. [56]). The system of interest is divided into various components that are listed along the lead diagonal elements (LDEs) of the matrix. Processes that relate the LDEs are entered in the off diagonal elements (ODEs), as shown in Fig. 7. Note that the matrix is read in a clockwise sense, so that processes by which soils affect humans are found in the top right element, whereas processes by which humans affect soils are found in the bottom left element. It is important to ensure that the effects of processes are direct and are not mediated by interactions via a third element listed on the lead diagonal.

All interactions between the components of the biosphere are of interest when developing an understanding of the system to support the description of the biosphere. Similar diagrams can also be used directly in the development of conceptual and mathematical models of contaminant transport and exposure and are illustrated in Section 5.1.3, but it needs to be noted that interactions that are defined when used in the context of assessment model conceptualization are limited to processes that transport contaminants and result in exposures, rather than extending to all influences. So, for example, Fig. 7 includes the influence that humans have on plants through species introduction, which can be important in describing the biosphere system, although this could have no direct role in transferring contaminants in the biosphere. Some processes will switch location when they are translated into the transfer matrix; for example, consumption would move (from cell 2,3 in Fig. 7) to being a transfer from plants to humans (to cell 3,2 in Fig. 7).

4.2.2. Drawing on quantitative supporting information and modelling

This stage of the BIOMASS methodology delivers descriptions of the biosphere system(s) and their evolution that need to be represented in the post-closure safety assessment. Such descriptions are usually underpinned by the outputs from quantitative modelling, including, for example, projections of climate characteristics, landscape development, near surface hydrogeology, surface hydrology and particle tracking. Quantitative modelling that underpins the landscape description and supports the descriptions of the individual biosphere system(s) is subject to its own limitations and uncertainties. Understanding of such constraints needs to be carried through to the safety assessment, such that their implications for assessment results can be considered and potentially evaluated.

In a generic study of a region, it is likely to be possible to describe the climate conditions existing at the present day, the typical topographical contrasts and the variations in the lithology of the soils and sediments of the surface environment. It is also likely to be possible to make broad projections of these characteristics into the future, at least until grossly disruptive external events occur, such as the advance and retreat of ice sheets or glaciers over the region (see Section 4.1.1). Once attention is focused on one or a few sites, it is anticipated that desk studies will be complemented by a site characterization programme and that, as this programme proceeds, information from field and laboratory studies relating to the site or sites will come to dominate the information available to characterize at least some of the selected biosphere systems for such a site or sites.

Figure 2 highlights the importance of the dialogue between those conducting biosphere assessments and those working on the system understanding, the latter of which is informed by characterization of the present day site and/or of sites analogous to the biosphere conditions that are projected to occur in future (see Fig. 2). Site characterization studies will also deliver understanding to other components of an overall safety case, including, for example, baseline conditions for environmental impact assessment and inputs to engineering, construction and operational safety assessments.

Site understanding and site data collection are needed for a variety of purposes and this is an iterative process throughout siting and site evaluation stages, reflecting an increased and refined site knowledge over time. Repository programmes also need ongoing site characterization and associated monitoring in support of safety cases for construction, operation and the initial post-closure/institutional control phase. Site characterization constantly iterates with facility design, environmental impact assessment, and operational and post-closure safety assessments. This iteration provides feedback in both directions and informs planning for the next stages in the repository programme.

Because site characterization is multidisciplinary, it may be structured by discipline (geology, hydrology and hydrogeology, hydrogeochemistry, ecology, demographics, land use, etc.), with an overall site descriptive model being used to integrate the results into a coherent description. The site characterization as a product can be described as a sum of several parts; compiled generic and site data, conceptual and numerical models, descriptions and reports capturing local knowledge and the understanding of experts working with the natural system at the site. Consideration of the last part is especially important given the long time frames involved in national repository programmes.

Site characterization includes the collection of data that can be used in palaeoenvironmental reconstruction. Thus, site characterization addresses not only the current characteristics of the site, but also its past development. This can be very useful in making projections of potential future changes at the site, bearing in mind that anthropogenic contributions to climate change could mean that past changes at the site are not a good model for future changes, at least over the next few millennia to tens of millennia. Also, in making projections, it can be useful to characterize an area larger than the site, such that features that are not currently present on the site, but could be present in the future, are appropriately identified and characterized, especially to identify any changes in the location and size of potential discharge areas to the biosphere from a repository volume.

For post-closure safety assessment, site characterization functions as a basis for conceptual understanding in constructing assessment models, to provide site data for the models and to justify the site characteristics used. Some site characterization programmes have found it useful to specifically identify the surface environment in the overall conceptual model, with its description closely matched in terms of components and spatial extent to the initial state of the biosphere system adopted in the safety assessment. However, the biosphere concept used for safety assessment purposes will necessarily be a simplification and abstraction of what is known concerning the surface environment at the site.

Site information in later programme stages is often extensive, both because the characterization needs to address the various specific purposes outlined above, and because it is fundamental to providing an assurance of site understanding as a component of the safety case in its own right. Research models are often used to synthesize information, build understanding and, in turn, support simplifications made in assessment models. This helps in building confidence in stakeholder groups and, specifically, in people living in the vicinity of the site, providing assurance that their local knowledge has been considered, that specific concerns are being addressed through field investigations and that the results are explicitly being taken into account in safety assessments.

It is important that site characterization programmes and the data they deliver are structured in such a way as to facilitate the various uses of those data, including their use in developing biosphere system descriptions appropriate to safety assessment. Site characterization is also discussed further and exemplified in Annex III.

4.2.3. Determining the area of interest for biosphere modelling

The developed assessment context and narratives provide a framework for determining the biosphere area of relevance to subsequent modelling of contaminant migration and potential exposures. Assessments will typically focus on regions of the biosphere where the highest contaminant concentrations can be expected to occur and over spatial areas of relevance to the defined PEGs and PEPs. In some contexts, potential exposures over larger spatial areas also need to be assessed (e.g. see Ref. [11]).

The location and extent of potential contaminant discharge areas, dynamics of water and mass balances within the biosphere, potential for accumulation of contaminants over time, dynamics of biosphere change, ecosystem and habitat characteristics, biota population sizes, and extents and characteristics of human communities and their associated habits all provide an input to determining the area of interest for biosphere modelling.

The areas of interest for assessment models can be understood as those areas in the landscape where radionuclides and other contaminants can accumulate and where exposure can occur. Topography, lithology and climate govern the processes by which water and sediments move and accumulate in the surface and subsurface environments, carrying contaminants with them. These contaminants can reach the surface environment via different pathways, including natural groundwater discharge or well abstraction. The area of interest in these two cases will differ. The following text focuses on the case of natural groundwater discharges.

National geographical databases, or data from site investigations, can provide information on present day topography, land use and climate. There are a variety of geographical information system tools and public databases that can be used to analyse these data to describe the landscape. This information can be carried forward to define locations in the landscape to be analysed in the assessment model. A DEM describes the topography to which a watershed analysis can be applied. Such an analysis provides the location of topographical boundaries (i.e. the catchment areas), but also the topographical low points within those areas which represent a first approximation to the location of groundwater discharge areas. For a more detailed analysis of the location of recharge and discharge areas, a hydrogeological model can be used. The locations of recharge and discharge areas are also dependent on the climate; therefore, climate evolution could be taken into consideration when defining the area of interest.

Depending on the climate and landscape evolution the spatial extent and location of the area of interest might change over time.

There are different methods that can be used, with different levels of ambition given the programme stage and the assessment of context questions that need to be answered, to identify areas of interest, including, for example:

— Using an early and generic or semi-generic approximation of discharge areas of deep groundwater as the basis for a simplified assessment model with data based on a simplified interpretation of hydrology and landscape development [57, 58];
— Using site specific and distributed, evolving hydrological models and terrain and ecosystem models to provide input data to the assessment model [59–65].

Analysis of the landscape's topography can identify many potential areas of interest in the landscape. Information on potential release locations, coming from modelling of, for example, particle tracks in geological media outside the domain of the biosphere model, can be used to reduce the number of potential areas that need to be included in the assessment model.

4.2.4. Characteristics of human communities and biota populations

The fundamental role played by human communities as a principal component of the biosphere is an important part of the system description (see Section 3.2). If human doses and risks are an end point, it is necessary to explicitly include human communities within the conceptual model, especially to support identification of exposure pathways.

The assessment context provides a description of the contaminant source terms to the biosphere and the narrative description provides information about the potentially evolving environment into or within which those contaminants are released, together with a qualitative definition of human communities and biota within that environment. These inputs support the identification of human communities and biota as components potentially needing explicit representation within the biosphere model.

The next step of the updated BIOMASS methodology is to specifically identify the PEGs and possibly PEPs to be described, with reference back to guidance on the definition of end points included within the assessment context (see Section 3.2).

Several factors merit consideration when defining PEGs and PEPs (if necessary) for explicit representation in biosphere modelling; these are discussed below:

— There is inevitably a degree of iteration between developing a conceptualization of contaminant transport/accumulation (see Section 5.1.3) and choosing and defining the human PEGs and biota PEPs. Lessons learned from experience of identifying and defining PEGs and PEPs are also reflected in Annex II and these support the updated guidance below.
— The source term of contaminants to the biosphere and the behaviour of PEGs and PEPs provide constraints on the areas of interest (see Section 4.2.3). Assessments typically aim to ensure that potential exposures are not underestimated, which focuses attention on areas of potential contaminant accumulation, coupled with behaviours and associated transfer pathways that tend to maximize associated exposures.
— Approaches to the selection of PEGs usually consider the potential activities of PEGs and their associated modes of exposure based on the system understanding, the source term and the narrative. These characteristics may differ substantially in different contexts (see Annex II).

In general terms and in relation to radiological exposures of humans, for radiation protection purposes, the emphasis is on the potentially most exposed groups, which in turn are described by a representative person [30]. The ICRP recommends the consideration of exposure of adults as an adequate representation of long term (lifetime) exposure [30]. However, there could be requirements or requests (e.g. from stakeholders) for assessment of doses to other age groups, which may then lead to a need to consider additional aspects in the conceptual modelling.

PEGs are selected to be representative of plausible patterns of habits and behaviour that might occur in the biosphere systems being represented (see Section 5.1.2). PEGs usually reflect a rural agricultural village community, which, in terms of food production and consumption, can represent or include a self-sustained group or community. The descriptions of human communities and terrestrial and aquatic biota constrain the types of land use and partially determine the types of biota likely to be present and therefore the choice of both PEGs and PEPs to be adopted in the model.

Models for contaminant transfer from a source term to an end point (for either PEGs or PEPs, as necessary) are needed for the exposure modes that could occur, having identified those exposure modes that need to be assessed quantitatively.

In broad terms, exposure modes of humans from radionuclides released to the biosphere can be divided into external or internal exposure. External exposures arise due to irradiation from the contaminated environmental media and are determined by the geometry of those media, the sizes and shapes of the exposed organisms, their locations relative to the contaminated media and their residence times in those locations. Internal exposures arise primarily from intake of radionuclides into the body by ingestion or inhalation, although uptake from skin surfaces or from wounds can be of significance in some contexts. Ingestion can arise from a variety of transfer pathways, which broadly can be described as foods, waters and non-food materials, for example soil or sediment attached to foods.

The representative person can be selected to be representative of either a small or a larger population (e.g. a distinction is made in the Finnish regulations, where different standards are adopted for the most exposed small population and a larger population located around a contaminated regional lake [32]).

If the spatial scale adopted in the assessment (see Section 4.2.3) is large enough to provide sufficient resources to support the defined PEGs and PEPs (if needed), it may be convenient to adopt distinct conceptual models for different parts of the system or for consideration of PEGs and PEPs separately. Such separation could be made, for example, at the catchment, subcatchment and/or ecosystem level.

Human communities can also provide a vector for contaminant migration (e.g. via dredging of sediments). It is emphasized that the human communities that play a role in contaminant migration are closely related to the PEGs adopted in radiological impact assessments but are not necessarily identical to them, for example because the dominant exposure modes may be different. Assumptions relating to the contribution of people to potential contaminant migration and accumulation pathways will be implicitly incorporated in the descriptions of the characteristics, configuration and dynamics of other biosphere components (see Section 5.1.2). Therefore, human land use needs to be seen as an important part of the biosphere system understanding needed to model fluxes of contaminants.

5. MODELLING THE BIOSPHERE IN THE SAFETY ASSESSMENT

After identifying, justifying and describing the biosphere systems adopted for assessment, the next step of the BIOMASS methodology is to develop associated conceptual and quantitative models of contaminant migration and potential exposure modes. The biosphere models need to be developed in a way that is both practical and transparent. Model development needs to therefore follow a systematic approach that allows assumptions and simplifications to be recorded and justified in a traceable manner.

The starting point for modelling contaminant transport, accumulation and evaluating potential exposure is the description of the biosphere system(s) (see Section 4.2) in which exposures are assumed to take place, coupled with a description of contaminant releases into or within that system.

The development and justification of a biosphere model and refinement of the associated description will generally be an iterative process from the outset. Initial model development and subsequent iteration aim to ensure that a practicable and justified modelling approach is achieved and maintained at all stages of the disposal programme.

In the BIOMASS methodology, the following basic steps towards model development are identified, building on the description of biosphere system(s):

(a) Develop a conceptual understanding of contaminant release, migration, accumulation and potential exposure for each of the biosphere system(s) carried through from Section 4 (Section 4.5.1):
   (i) Identify those biosphere components that are to be distinguished as separate features in the representation of mass and contaminant transport (i.e. distinct potentially contaminated environmental media);
   (ii) Identify and characterize the human PEGs and biota PEPs (if needed) that are to be explicitly addressed in the assessment;
   (iii) Develop conceptual models identifying the processes that result in contaminant transport between the biosphere components and give rise to potential exposure of the PEGs and PEPs (if needed).

(b) Develop a mathematical representation for the FEPs comprising each of the conceptual models, taking account of the extent and quality of input data that will be available when the model is to be used (Section 5.2). The mathematical representation is developed in an iterative process with feedback to and from site characterization and the overall system understanding.

(c) Collate and justify the input data for the mathematical models, drawing on the description of the biosphere system(s) and taking account of the approach with respect to the treatment of parameter uncertainties (Section 5.3 and see also Section 3.3).

(d) Implement, verify and validate the models (Section 5.4).

5.1. CONCEPTUAL MODEL DEVELOPMENT

An overall conceptual model for the biosphere may be developed that encompasses potential contaminant behaviour and exposure within all the scenarios and calculation cases to be considered within an assessment. Alternatively, several specific conceptual models may be developed for each scenario, or for subsets of calculation cases within a scenario. The choice will reflect the degree of consistency in the timing, location and form of contaminant releases to the biosphere, and the variety of assessment end points that need to be addressed. For example, radionuclide releases in post-glacial conditions could necessitate a different conceptual model to releases in interglacial conditions; radionuclide releases in the gas phase could need to be treated differently from releases in groundwater, releases of radioisotopes of elements that are involved in major biogeochemical cycles (e.g. ^{14}C) could need specific models and impacts on PEPs could require a different model from that used to assess impacts on PEGs.

The conceptual models that are delivered at the end of this stage of the methodology identify the features of the biosphere that need to be represented in the assessment, and the processes that result in contaminant transport and accumulation within the biosphere and that give rise to potential exposure of the PEGs and PEPs (if needed). The process of developing a conceptual model is broken down in the subsection below to:

(a) Identifying the biosphere features (see Section 5.1.1);
(b) Identifying the PEG and PEP end points (see Section 5.1.2);
(c) Identifying the processes that can result in contaminant migration, accumulation and potential exposure (see Section 5.1.3).

In practice, these steps are closely integrated and iterative, but are separated below for the purpose of explanation. An example of a simple conceptual model delivered by these steps is provided in Section 5.1.3, by way of illustration. More detailed examples can be found in the references provided in Annex I.

5.1.1. Identification of conceptual model components

Environmental media to be represented as separate conceptual model components are distinguished not only based on their contribution to potential contaminant impacts (e.g. resources exploited by humans and biota), but also in terms of the role they would play in contaminant migration and accumulation. These conceptual model components and their properties are largely analogous to features within the biosphere system from the perspective of FEP analysis. Their properties are conditioned by the principal characteristics of the biosphere.

These biosphere characteristics that are identified are discussed below from the perspective of contaminant behaviour and potential exposures. Several conceptual model components may be explicitly identified under each of the following principal characteristics to distinguish different locations, dimensions and properties of relevance to assessment of contaminant migration and potential exposure.

(a) Climate and atmospheric composition. The near surface atmosphere within the modelled region comprises one or more components of the biosphere, the properties of which are defined by the local climatic conditions, for example it may be treated as several layers within and above the vegetation canopy. Contaminants can reach the atmosphere as particulate or liquid aerosols and/or gaseous forms. Contaminants reaching the atmosphere may be dispersed relatively quickly outside the biosphere system of interest.

(b) Location, geographical extent and topography. The geographical extent and topography of the biosphere system will determine the spatial extent of the media components to be represented in the conceptual model. It will also determine whether any explicit subdivision of biosphere components (e.g. soils, sediments, water bodies) may be needed to reflect their configuration. For example, where releases may occur to locations with substantially different characteristics, such as agricultural land or a lake, it may be convenient to develop and apply distinct conceptual models for different ecosystems.

(c) Near surface lithology. This is distinguished into subsurface components (e.g. below a rooting/bioturbation depth) and surface soils and sediments:

  (i) Subsurface lithology. If encompassed within the biosphere modelling domain, this comprises material overlying a bedrock. It includes, for example, till and/or other material beneath soils and sediments that has been deposited over time, such as layers of clay, sand and different types of organic sediment. Components of the subsurface lithology help to define the configuration, hydraulic and sorption properties of the hydrogeological system and may also host near surface aquifers (it is therefore linked with water bodies; see below).

  (ii) Soils/sediments. Soils and surface sediments are intrinsic components of terrestrial and aquatic ecosystems and it is, therefore, appropriate to identify physically distinct domains within the conceptual model or models, linked to each ecosystem that is identified as being present within the biosphere. If the interface for release to the biosphere is an irrigation well, it may be appropriate simply to consider a single ‘irrigated soil’ medium. However, depending on the nature of the GBI and on how components of the biosphere system are spatially configured, further subdivision may be necessary to distinguish domains of the same or different soil or sediment types that play distinctly different roles in contaminant transport and accumulation.

(d) Water bodies. As with near surface lithology above, this is distinguished into surface and subsurface components:

(i) Subsurface water bodies. Subsurface water identified as belonging to the biosphere can be considered conceptually as part of the geological strata within which they exist (see above). Alternatively, when developing a conceptual model, it may be helpful to consider soil/sediment solids as distinct from soil/sediment water, that is, to have two conceptually distinct environmental media occupying the same spatial domain. This would be the case, for example, if there was a need to explicitly represent exchange processes between soil solids and soil water.

(ii) Surface water bodies. Each surface water body (natural or artificial) that is identified as belonging to the domain of the biosphere of interest can play a distinct role in the distribution of contaminants and may support a separate ecosystem or sub-ecosystem. Depending on how potential transport and exposure pathways are affected by the assumed spatial and temporal characteristics of surface water bodies, as well as their configuration within the landscape, it may be helpful to distinguish different water bodies as separate conceptual model objects. For example, it may be convenient, or necessary, to distinguish streams/rivers, lakes, estuaries and marine water, and/or distinguish between deep and shallow lake waters.

(e) Biota. Typically distinguished into plants and animals:

(i) Plants. Taken to encompass plants, fungi, algae, lichen, etc. and will comprise cultivated plants as well as wild plants, including both terrestrial and aquatic plants. Plants may be an end point in themselves (e.g. in biota dose assessment) and/or may contribute to other end points (e.g. doses to humans). Plants will also be involved in contaminant migration and distribution in the biosphere, for example by taking contaminants up and then changing their mobility by incorporation into organic matter. Plants, therefore, merit explicit consideration as a component of the conceptual model, being closely linked to assumptions on land use and human behaviour, and may merit further subdivision into parts of plants, as necessary or convenient.

(ii) Animals. Including both terrestrial and aquatic animals. Animals will encompass domesticated livestock as well as wildlife. As with plants, animals can be an end point in themselves (e.g. in biota dose assessment) and/or can contribute to other end points (e.g. doses to humans). Animals will also be involved in contaminant migration and distribution in the biosphere, for example by vertical transport and redistribution of soil and associated contaminants (bioturbation). Like plants, they are closely linked to assumptions on land use and human behaviour. Animals therefore merit explicit consideration as a conceptual model component.

(f) Human communities. Given their typical status as end points for the safety assessment, human communities and associated PEGs will need to be included as explicit components of the conceptual model (see Section 5.1.2).

It is emphasized that, as with principal characteristics, the biosphere part of the safety assessment can determine those aspects that are chosen to explicitly represent components in developing a conceptual model, but that choice needs to be justified.

The list of principal characteristics above is intended as a guide. Iteration could lead to refinement of the conceptual model, with either greater resolution or amalgamation of aspects that can be considered together.

In addition to the main components to be considered within the biosphere system, there are two further aspects that are important for explicit consideration in developing the conceptual model.

(1) Source term(s) to the biosphere. Although often not part of the biosphere itself, the contaminant source terms are an important consideration in the development of the conceptual model of contaminant behaviour and potential exposure. The nature of the releases to or within the biosphere will determine the form of contamination; for example, aqueous phase in well water or emerging groundwater, gaseous phase in releases of contaminated gas to the biosphere and/or solid phase in eroding material or contaminated material (including waste) arising from human intrusion. The spatial distribution of the source term(s) will determine which biosphere media are initially affected and the form of contamination will determine the processes that will affect contaminant behaviour in the biosphere. It is therefore typically convenient to explicitly represent the source term(s) as a conceptual model component.

(2) Contaminant losses. The spatial extent of the biosphere will be based on the objectives and constraints defined in the assessment context. Typically, safety assessments are limited to considering the location where potential exposures will be greatest, which, in turn, will typically be near, include or lie within the areas of releases into the biosphere. The conceptual model will need to explicitly consider mechanisms for contaminants to be lost from the biosphere system to be represented, for example via the flow of air, surface water and groundwater out of the system, or through degradation

or radioactive decay. Failure to recognize these losses would result in an artificial buildup of contamination within the model representing the environment.

5.1.2. Defining potential exposure groups for humans and potentially exposed populations of biota for exposure assessment

The assessment context provides a description of the contaminant source terms to the biosphere and the biosphere description provides information about the potentially evolving environment into or within which those contaminants are released, together with a definition of human communities and biota within that environment. These inputs support the identification of human communities and biota as components needing explicit representation within the biosphere model, as described in Section 5.1.1. The next step of the BIOMASS methodology is to specifically define the behaviours and characteristics of PEGs and PEPs (if needed), with reference back to guidance on the definition of end points included within the assessment context (see Section 3.2) and the system description (see Section 4.2.4).

5.1.2.1. Human potential exposure groups

(a) Defining potential exposure group habits

As mentioned earlier, PEGs are usually selected as representative of plausible patterns of habits and behaviour in the biosphere systems being represented (see Section 4.2.4). Their characteristics are therefore defined in accordance with their activities and associated food habits (e.g. farmer, hunter-gatherer, etc.). Table 1 provides a list of human activities that could result in exposure to contaminated components of the biosphere and can be used to help identify PEG habits and lifestyles [6]. Table 1 also lists parameters relevant to defining the extent of potential exposure and for characterizing PEGs [6]. Examples are provided in Annex II.

It is important to keep in mind that it is not possible to predict societal or human behavioural characteristics over the multimillennial timescales typically of relevance in post-closure safety assessment of solid radioactive waste disposal (see Section 3.7). Selection of both the communities to be adopted for assessment purposes and the PEGs within them could, however, need to take into account stakeholder interests. There could, for example, be interest in including specific characteristics and patterns of behaviour considered distinctive of present day communities in the assessment, even if potential radiological impacts are found to be relatively low. Habits may draw on local surveys or surveys for analogous

TABLE 1. HUMAN ACTIVITIES LEADING TO POTENTIAL RADIATION EXPOSURE [6]

Source/ medium	Exposure mode	Example exposure route	Examples of typical activities	Assumed parameters[a]
Soil	Inhalation	Gaseous release to air	Outdoor activities, indoor activities	A, B, E
		Resuspension of soil particulates	Ploughing, walking, miscellaneous outdoor activities, indoor exposure resulting from soil brought inside	A, B, E
	Ingestion	Incidental soil ingestion	Gardening, fresh fruit and vegetable consumption, recreational activities, occupational activities	A, B, H
		Deliberate soil ingestion	Soil pica	B, H
	External	External radiation exposure	Activities over/near contaminated soil, including dermal contact, living in buildings made of contaminated soil	A, C, F, G
Water	Inhalation	Spray/aerosols/ volatiles	Spray (irrigation, surface water), recreation, domestic (showering, sauna, cooking), recreation/fishing	A, B, E
	Ingestion	Deliberate water intake	Drinking, as a part of diet in other foods (cooking)	B
		Incidental water intake	During swimming, bathing, showering	A, B

TABLE 1. HUMAN ACTIVITIES LEADING TO POTENTIAL RADIATION EXPOSURE [6] (cont.)

Source/ medium	Exposure mode	Example exposure route	Examples of typical activities	Assumed parameters[a]
Water (cont.)	External	Submersion in water	Bathing, swimming	A, C, F, G
		External exposure from water bodies	Working near bulk water (storage tanks, filtration systems), recreational activities near water bodies	A C, F, G
	Dermal absorption	Submersion in water	Farming activities, interception of spray irrigation, swimming, bathing	A, B
Sediments	Inhalation	Gaseous release to air	Outdoor activities on exposed sediments and sediments transferred to soil by dredging	A, B, E
		(Re)suspension of dried sediments	Dredging, maintenance of water distribution system, farming, activities on shorelines and near perennial lakes	A, B, E
		Spray, including suspended sediments	Irrigation spray, showering	A, B
	Ingestion	Incidental ingestion	Dried/exposed sediments as deposits on food, or fingers, suspended sediment with water	B
		Deliberate ingestion	Sediment pica (dried exposed sediments only)	B, H

TABLE 1. HUMAN ACTIVITIES LEADING TO POTENTIAL RADIATION EXPOSURE [6] (cont.)

Source/ medium	Exposure mode	Example exposure route	Examples of typical activities	Assumed parameters[a]
Sediments (cont.)	External	γ irradiation from bulk sediments	Activities (recreational and occupational) near exposed sediments, dried sediments, swimming, bathing	A, C, F, G
Air	Inhalation	Breathing	All activities (indoor, outdoor, including sleeping)	A, B
	Ingestion	Particulate deposition on surfaces/foodstuffs	Eating, recreational activities	B, D, H
	External	Submersion dose	γ exposure from airborne concentrations (all types of activity)	A, C, F, G
Plants and plant products	Inhalation	Particulates from combustion, from plant processing	Burning of plant material (wood, stubble, specific crops (e.g. tobacco)), milling	A, B, E
	Ingestion	Food consumption	Eating, drinking plant material as part of the diet, root and green vegetables, cereals, fruit, etc.	B
	External	γ exposure from plants and plant products	Working/recreation in fields, storage of plants, wearing clothes derived from plants, building materials	A, C, F, G

TABLE 1. HUMAN ACTIVITIES LEADING TO POTENTIAL RADIATION EXPOSURE [6] (cont.)

Source/ medium	Exposure mode	Example exposure route	Examples of typical activities	Assumed parameters[a]
Animals and animal products	Inhalation	Inhalation of animal derived particulates	Derived from domestic activities (cooking), occupational activities (incineration, butchery, tanning)	A, B, E
	Ingestion	Food consumption	Animal products consumed include meat, milk, offal, eggs, dairy products, other products (e.g. gelatine)	B
	External	γ exposure from animals and animal products	Animal husbandry, processing/storage of animal products and materials	A, C, F, G

[a] A: exposure duration (h/a); B: rate of intake (kg/a); C: shielding of source (yes/no, shielding factor); E: resuspension/release rate ((kg soil) (m^3 air)$^{-1}$), m^{-1}, kg/h, etc.; G: relation to source (distance, orientation — above, beside, below, immersed, etc.); D: deposition rate ($kg \cdot m^{-2} \cdot a^{-1}$); F: source geometry (infinite plane, line, sphere, semi-infinite cloud, etc.); H: age specific information.

types of biosphere. However, even if the focus is on the present day community, the consideration of other potential future communities within which potential exposures might differ due to climatic and/or landscape change and activities resulting from those conditions is important (see Annex III).

Prior to undertaking the assessment modelling, it could be unclear which parts of the biosphere will give rise to the highest potential exposures. If it is considered unreasonable to assume that a single PEG could be considerably more exposed than other plausible PEGs, via the range of potential pathways that are to be assessed, then more than one group can be defined. This reflects experience that in the early stages of an assessment, or a sequence of assessments, it is impossible to know which pathways and exposure mechanisms will dominate the results, so it may be necessary to consider several PEGs.

After analysis of initial results, it will be possible to refine the assumptions and identify the more highly exposed group (representative person) in the context of the specific assessment being undertaken. To achieve this, it is important to ensure that a relevantly wide range of PEGs has been considered, while ensuring that exposure assumptions remain reasonable and sustainable. Experience from previous assessments can be helpful in this regard (e.g. see Annexes I, II and III in Ref. [9]).

Throughout the process of defining PEGs, it is important to remember that safety assessments do not aim to predict, but aim to provide an adequate representation of, long term (lifetime) potential exposures (see Section 3.2). Thus, PEGs are hypothetical constructs in which the habits cover all relevant pathways of potential exposure and, as far as can be ascertained, are reasonable and sustainable with respect to the considered area, as well as human physiological requirements [30].

In general, PEG characteristics have tended to be treated deterministically, using point values for occupancies of contaminated areas, consumption of food, water and other materials, and respiratory characteristics (see Annex II). Food consumption might also be supported by local or regional surveys. This approach tends to be adopted in recognition that the exposures being calculated are indicative and are for a representative individual, rather than being measures of the distributions of exposures in a real local population.

Where the area modelled has the capacity to supply all the resource (food and water) requirements of a PEG, this is typically assumed. In evaluating resource areas, consideration needs to be given to the size of the group. Where the area modelled is not plausibly sufficient to supply the resource requirements of a PEG, then local resource fractions are typically used to explicitly take into account a proportion of the group's intake that may be uncontaminated (see Annex II). Such assumptions need to be clearly documented and justified.

It may be noted that the higher assessed potential doses to PEGs are typically dominated by one radionuclide and one exposure mode via one exposure pathway. However, combinations of foods could need to be considered in terms of reasonableness. For example, the PEG can be assumed to have high consumption of one or two food types, consistent with the system in which they exist, but it would generally not be reasonable to assume high consumption of all foodstuffs. Such an assumption would in any case lead to inconsistencies relative to the values of dose coefficients, which are based on reference persons with standard physiology, etc.

Where probabilistic modelling is adopted, the use of broad food categories is suggested, as the variability in consumption rates can be large for individual foods, that is, the aim is to characterize variability in the intake of radionuclides across the diet as a whole.

(b) Average versus cautious exposure group assumptions

In selecting assumptions for PEGs, consideration will also need to be given as to whether to adopt average or more cautious patterns of behaviour. The combination of cautious patterns of behaviour with a requirement that those patterns of behaviour be focused on the more highly contaminated components of the environment could be regarded as unduly cautious.

Deciding on what is appropriately cautious and what is unduly cautious is a matter of judgement, but that judgement needs to be informed by the assessment philosophy and treatment of uncertainties as defined in the assessment context (Section 3.3).

The reasoning behind the 'cautious' approach is that the intention is to ensure that a given dose or risk constraint or target will not be exceeded. This implies that assumptions relevant to dose assessment are chosen to be pessimistic[4] and focus on those people (representative person) who would potentially receive the highest doses.

The assumption behind the 'average' approach is that the objective is to understand the more likely level of potential exposure. This may be useful in optimization studies [40]. Further, to be proportionate, it better allows for the risk from the disposal facility to be regulated to the same level as other risks incurred by individuals. Some of the levels of individual risk that society currently tolerates and that regulatory bodies use in setting standards are based on society wide averages rather than on higher risk subgroups. In contrast with the cautious approach, the average approach implies defining potential exposure groups on a less cautious basis.

It may be entirely appropriate to use either approach to compare with a constraint or target, depending, among other things, on how that constraint or target was selected. Whichever approach is used, the rationale needs to be transparent for both the choice of the constraint/target and the PEG characterization approach.

5.1.2.2. Potentially exposed populations of biota

(a) Selection principles

Radiation Protection and Safety of Radiation Sources: International Basic Safety Standards (IAEA Safety Standards Series No. GSR Part 3) [66] includes

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[4] These can also be referred to as 'conservative' or 'bounding' assumptions.

requirements for the protection of the environment. Paragraph 1.35 of GSR Part 3 [66] states that:

> "These Standards are designed to identify the protection of the environment as an issue necessitating assessment, while allowing for flexibility in incorporating into decision making processes the results of environmental assessments that are commensurate with the radiation risks."

The ICRP also calls for explicit demonstration of environmental protection in its 2007 Recommendations [29]. In response, some national regulations now require radiological assessments to explicitly assess potential impacts on non-human biota from activities resulting in the release of radioactivity to the environment.

To support the explicit demonstration of environmental protection, the ICRP has developed a protection framework for non-human biota [67] with supporting transfer parameters [68] and dose coefficients [69]. The framework is centred around the use of a set of 12 reference animals and plants (RAPs) [70], which are:

> "a small set of hypothetical entities that are representative of animals and plants present in different environments (terrestrial, freshwater, marine) and which form the basis of a structured approach to the assessment of exposures to, and effects of, ionising radiation."

The RAPs are essentially reference models that, together with associated derived consideration reference levels (DCRLs; order of magnitude bands of dose rate for each type of RAP within which some detrimental effects may occur in response to chronic radiation exposure), provide points of reference that can help guide decisions around environmental protection [70, 71]. The intended application of the RAPs and DCRLs for planned exposure situations is described in Ref. [70].

For biota, absorbed dose rates from external exposure plus internal exposure are usually calculated for evaluation of impacts, sometimes with high linear energy transfer (LET) components given a larger weighting than low LET components to allow for their greater radiological effectiveness in inducing deleterious effects. These dose rates are typically compared with screening dose rates below which significant adverse effects on populations are not expected to occur. It remains unclear how dose rates above these screening levels are to be interpreted in terms of potential adverse effects on populations.

Whilst intended as points of reference, it is recognized that for site specific assessments it may be necessary to identify species that are representative of the area of interest [70, 71] (i.e. 'representative organisms' [70]), which are defined as [70]:

> "A particular species or group of organisms selected during a site-specific assessment, taking account of their assumed location with respect to the source. In many cases, the actual representative organisms chosen for this purpose may be the same as, or very similar to, the Reference Animals and Plants; however, in some cases, they may be very different."

The geometrical representations of the ICRP RAPs have been incorporated within the assessment tool[5] developed through the European Commission EURATOM Framework funded ERICA (Environmental Risks from Ionising Contaminants) project to support exposure evaluation in relation to radionuclide concentrations in a range of environmental media. The RAPs sit within a wider range of 'reference organisms' that represent the types of organism typical of marine, freshwater and terrestrial ecosystems throughout Europe. The ERICA tool also has added functionality that enables additional representative organisms to be added, as necessary, to enable a more site specific focus for assessments. Alternatively, dose coefficients for representative species with user defined shapes and masses and positions in the environment relative to contaminated media can be calculated using the BiotaDC tool, described in Ref. [69]. Several alternative tools to support assessments of radiological impacts on biota are publicly available.[6]

Where a site specific assessment is to be undertaken, the use of representative organisms that could be linked to the site in question may be appropriate [70]. Inclusion of site relevant species can address the interests of stakeholders, support communication and help increase confidence in assessments [72]. However, species of specific stakeholder interest might not be those that assessment specialists would identify as most at risk. The assessment context therefore needs to be clear as to whether the objective of the assessment is to address stakeholder interests or to address potentially different aspects of environmental protection, for example from a scientific perspective, or both.

Pragmatically, an assessment cannot take account of all living species and populations within an area of interest and some selection criteria will therefore be necessary to identify a manageable number of representative species that will be adequate to address the assessment objectives. The selection criteria will be informed by the assessment context. For example, national regulations may require the inclusion of certain species (e.g. species of specific conservation

[5] Available at erica-tool.com

[6] Available at wiki.ceh.ac.uk/display/rpemain/Assessment+tools

status, keystone species or species of public interest) or for assessments to take account of specific events or time frames (e.g. to take account of climate and/or landscape evolution). The transport pathways by which contaminants enter the surface environment will also be an important consideration. For example, whether contaminants enter the surface through natural upwelling of groundwater or via extraction of groundwater for irrigation purposes will affect the species of interest for assessments. It is worth noting in this regard that the focus of selection of species for assessment is wildlife rather than domestic farm animals; the ICRP position being that, for typical farm animals living in managed agricultural systems, demonstration of human protection within that environment would be sufficiently protective of those animals [70].

Criteria that have been applied in the selection of representative wildlife species in long term safety assessments to date (e.g. see Refs [72–74]) include:

- Species with a critical role (i.e. keystone species such as wolves);
- Species representative of functional groups/trophic levels (e.g. primary producers, apex predators, etc.);
- Foundation species (those that are very abundant);
- Species of economic importance and/or strongly identified with the assessment area (e.g. game species);
- Species of conservation value (e.g. rare or endangered species);
- Species inhabiting habitats most likely to be affected by groundwater discharges and/or with greater exposure potential due to habitat occupancy (e.g. soil dwelling organisms);
- Species with a limited home range such that populations may be located almost entirely within the area of interest;
- Species resident in the area of interest throughout the year (i.e. non-migratory species);
- Species for which there is reasonable knowledge in relation to ecosystem behaviour.

The use of ecosystem food webs can be useful in ensuring that key trophic/functional groups are represented in the species selected for assessment [73, 75]. Appropriate characterization of the ecosystem in the area of interest, and how it could evolve in the future, is therefore important in selecting species for assessment. Considering species present in analogue areas can also be useful where landscape and/or climate evolution are to be taken into account.

A further consideration in the selection of any representative species will be the availability of dose assessment parameters, particularly body dimensions and mass, and contaminant transfer factors in the environment. Although body dimensions/mass are necessary parameters for user defined representative organisms in the assessment tools, size variation has been demonstrated to have a limited effect on the estimated absorbed dose rates to the organisms [76, 77]. The number of species represented in assessments could therefore be justifiably limited, particularly where transfer factors (recognized as being an important factor governing dose rates) are similar for different species or where there are insufficient data on radionuclide transfer to justify the inclusion of multiple similar species. Depending on the site specificity of the assessment, assessment parameters, including representative species dimensions/mass and contaminant transfer factors can be derived through site characterization programmes and/or use of data from analogue sites. Alternatively, representative organisms could be mapped to reference organism entities for the purpose of dose calculations with or without the use of species/site relevant transfer factors (e.g. see Ref. [72]).

(b) Consideration of spatial context

It is broadly accepted that the focus of protection for non-human species is not on individuals (with the notable exception of endangered/protected species [78]), but rather at higher levels of organization, such as populations, communities and ecosystems [67, 70–72, 75, 79, 80]. These protection objectives were considered in the context of a solid radioactive waste disposal safety assessment framework in Ref. [75]. The conclusion in that study was that the protection of populations provided a pragmatic and accessible target for the demonstration of environmental protection, requiring consideration of the spatial area over which contaminants are present relative to the area occupied by the relevant population. The area utilized by populations relative to the spatial extent of contamination is also a key area of consideration in the ICRP environmental protection framework, which calls for assessments to consider the different fractions of a population exposed to different dose rates [67]. In support of this, data on population characteristics for each of the RAPs are provided (see table 2.1 in Ref. [67]). This is broadly consistent with Radiological Environmental Impact Assessment for Facilities and Activities (IAEA Safety Standards Series No. GSG-10) [81], which recommends averaging of contaminants throughout a reference area around a release point, where the reference area is sufficiently large to ensure that the exposure of an appropriate proportion of a population is evaluated.

While populations can be an appropriate protection end point for non-human species, the calculation of potential exposure is based around individual entities (e.g. an ellipsoidal representation of an individual plant or animal). This leads to the question of how to frame assessments within the context of protecting populations. At the simplest level, the argument could be made that assessments demonstrating protection of the individual are naturally protective of the population, even for the most sensitive species. However, such a strict level of assessment may be unreasonably prohibitive [78]. For example, the use of maximum activity concentrations in environmental media could give rise to dose rates exceeding available benchmarks, but those activity concentrations may be limited to a small area that would not be sufficient to sustain a population. Giving due thought at the outset (i.e. as part of the assessment context) as to the spatial requirements of populations of interest and how a population protection end point is to be evaluated could prevent such issues arising. One approach to addressing this was used in Ref. [75], where average contaminant concentrations over the spatial area needed to support a relevant population were used, so as to evaluate typical exposures across the population of interest. This is broadly consistent with the approach described in Ref. [81], where activity concentrations are recommended to be averaged over an area of sufficient size to support a suitably large number of individuals to ensure that calculated dose rates are representative of the exposure across a fraction of the most exposed population rather than the most exposed individual in the population.

Such an approach has also been explored in the BIOPROTA project SPACE (Scales for Post-Closure Assessment Scenarios) [82]. In this project a set of assessment species was selected to encompass a range of terrestrial animal groups (mammals, birds, reptiles, amphibians, invertebrates). The home ranges of assessment species (reflecting the extent of an individual's interaction with its environment) were investigated and a population scaling approach applied (based on Ref. [83]) to derive an estimate of the spatial area needed for a population of each assessment species. The home range of individuals will be highly variable, being influenced by factors such as habitat availability, food resources, season and climate (noting that climate can affect not only the spatial range of populations, but also the types of plant and animal that may be present in the assessment area). Whilst recognizing the inherent uncertainties, the approach nonetheless provides a simple and pragmatic approach to estimating area requirements for populations. A similar approach could also be applied to plants, with the spatial area for a population being evaluated using the area occupied by an individual multiplied by a population multiplier. In this instance, the population data for RAPs provided in table 2.1 of Ref. [67] can provide a pragmatic basis for calculating the spatial area over which contamination could be averaged to evaluate the exposure of a relevant population. To minimize the need for multiple alternative calculation cases and model discretizations of the area of interest in assessments, the SPACE project [82] grouped assessment species in terms of similar spatial requirements (e.g. population area requirements of <0.5 km^2, 0.5–10 km^2 and >10 km^2), with contaminant concentrations being averaged over these spatial areas as input to exposure evaluation.

Whilst the SPACE approach [82] provides a simple and pragmatic approach to evaluating contaminant concentrations relevant to a population end point, the use of alternative area discretizations in assessment models might not be appropriate in all instances (e.g. where distinct biosphere systems are the focus of assessments). Discussion around dose rates calculated over the modelled area relative to the fraction of the population exposed could be a reasonable alternative. An example of the application of this type of approach is provided in Ref. [84]. The choice of approach needs to be clearly informed by the assessment context and commensurate with the level of risk [70] and the need to ensure stakeholder confidence.

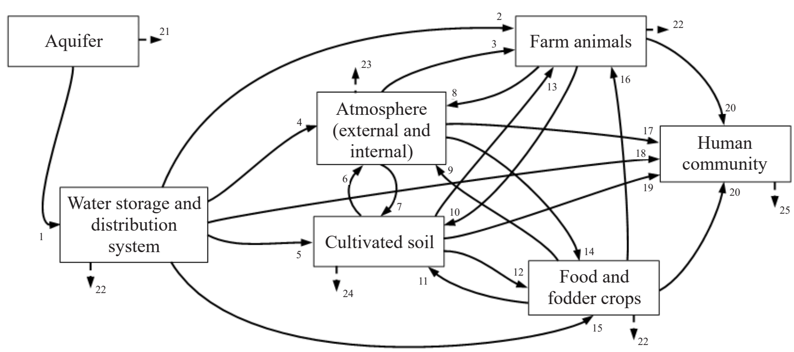
The appropriate selection of potential exposure scenarios will also be an important factor when undertaking biota dose assessments. Traditionally, the focus of safety assessments for solid radioactive waste disposal facilities has been on the protection of people, with assessments of radiological impacts on biota being a more recent addition. As such, safety assessment scenarios have largely focused on potential exposure pathways appropriate for human dose assessment (e.g. groundwater extraction for drinking water and agricultural irrigation) and assessment models developed with regard to PEG habits and resource requirements [82]. A focus on agricultural systems could be less appropriate for biota dose assessment since such managed systems are unlikely to sustain a wide diversity of biota populations and might not be representative of biotopes (areas of uniform environmental conditions) with the highest exposure potential for wildlife populations, such as natural groundwater discharge areas associated with springs or streams [82]. Scenarios that allow for the natural behaviour of plant and animal populations within the area of interest (e.g. occupancy in natural and seminatural ecosystems) could, therefore, be more appropriate for assessments of radiological impacts on biota. For example, Posiva considered the typical exposure of populations across the different biotopes that they could reasonably occupy within the assessment area of interest, including transient occupancy of agricultural lands, by assigning proportionate biotope occupancy factors [73].

5.1.3. Conceptual models of migration and exposure

Having identified the main biosphere system(s) and the distinct features within the system(s) to be represented the PEG and PEP end points that need defining, as needed, a conceptualized description of the dynamics of contaminant transport through the biosphere can be developed. This is based on an analysis of the description of the biosphere system. Such conceptualized descriptions are typically aided by tools such as process influence diagrams and/or interaction matrices, which help to systematically consider both the potential processes operating between the features to be represented and the associated exposure pathways. It can be helpful to consider both aspects in sequence.

(a) Process influence diagrams (see e.g. Fig. 8). The conceptual model components (features) identified can be represented as boxes in a flow diagram. Annotated arrows between the boxes can be used to illustrate potential movement of both mass and contaminants via specific processes that operate between the boxes. Similarly, arrows can be used to highlight exposure pathways to be considered.
(b) Interaction matrices (see e.g. Fig. 9 and Section 4.2 for a description of interaction matrices). The conceptual model components (features) identified above are represented as LDEs. Processes that operate to move mass and contaminants between the features are explicitly listed in the associated ODEs operating in a clockwise direction, with annotation potentially being used to illustrate the significance of interactions. Similarly, potential exposure pathways can be explicitly listed in the off-diagonal elements. Note that this is a different application of interaction matrices compared with their use in displaying influences, as discussed and illustrated in Section 4.2.

The conceptual model of the biosphere may show explicitly, but without the use of equations, how contaminants released into or within the biosphere can be evaluated in terms of doses to people and the environment. Thus, the main aspects that need to be included are the characteristics of the input fluxes,



Key:
Transfer/ exposure process
Loss term

1 Water abstraction
2 Drinking water, sediment consumption
3 Vapour/aerosol inhalation
4 Volatilization/degassing
5 Irrigation, sediment transfer
6 Suspension, volatilization
7 Vapour/aerosol deposition
8 Eructation
9 Transpiration, respiration
10 Manuring
11 Weathering, leaf fall, ploughed in detritus
12 Root uptake, soil splash
13 Consumption of soil on fodder crops
14 Vapour/aerosol deposition
15 Irrigation and interception
16 Ingestion of fodder
17 Vapour/aerosol inhalation, immersion
18 Ingestion, immersion, dermal absorption
19 External irradiation, ingestion
20 Ingestion
21 Decay, groundwater flow
22 Decay
23 Decay, advection
24 Decay, leaching/percolation, erosion
25 Decay, excretion

FIG. 8. Process influence diagram illustrating the conceptual model for contaminant transport and exposure arising from agricultural use of potentially contaminated well water (adapted from Example Reference Biosphere 2A in Ref. [6]).

Aquifer	Water abstraction	x	x	x	x	x	Decay Groundwater flow
x	Water storage and distribution system	Volatilization Degassing	Irrigation Sediment transfer	Irrigation and interception	Drinking water Sediment consumption	Ingestion Immersion Demal adsorption	Decay
x	x	Atmosphere (external and internal)	Vapour / aerosol deposition	Vapour / aerosol deposition	Vapour / aerosol inhalation	Vapour / aerosol inhalation Immersion	Decay Advection
x	x	Suspension Volatilization Gas	Cultivated soil	Root uptake Soil splash	Consumption of soil on fodder crops	External irradiation Ingestion	Decay Leaching / percolation Erosion
x	x	Transpiration Respiration	Weathering Leaf litter Ploughed in detritus	Food and fodder crops	Ingestion of fodder	Ingestion	Decay
x	x	Eructation	Manuring	x	Farm animals	Ingestion	Decay
x	x	Respiration	x	x	x	Human community	Decay Excretion
x (Recharge)	x	x	x	x	x	x	Sinks

FIG. 9. Interaction matrix illustrating the conceptual model for contaminant transport and exposure arising from agricultural use of potentially contaminated well water (adapted from Example Reference Biosphere 2A in Ref. [6]). Arrows illustrate the clockwise notation of the interactions.

their transport through and concentrations in the various environmental media comprising the biosphere system, and the implications of those concentrations for radionuclides in terms of annual effective doses to humans or absorbed dose rates for biota.[7]

5.1.3.1. Conceptual model for contaminant migration and accumulation

The conceptual representation of the biosphere system is developed by identifying all processes (and events) that are associated with contaminant transport between those environmental media that have been selected to be represented as separate conceptual model components. If an assessment is supported by an FEP list, then this list could either be used to as a source of reference for processes to be screened for inclusion within the conceptual

[7] For chemical contaminants impacts on humans are typically evaluated by considering exposures, intakes or concentrations in tissues or organs, whereas impacts on biota are typically evaluated by considering concentrations in environmental media.

model or could be used as an audit tool to review the resulting model (see Section 5.1.3.3). Contaminant transport pathways will depend on the assumed spatial configuration and connectivity of the biosphere system components, as reflected in the system description.

During development of the conceptual model, it is important to bear in mind that more than one process or event can act between two conceptual model components. Also, the features represented as distinct media/conceptual model components can have processes that operate intrinsically within them, for example the processes of sorption that determine partitioning between solid and liquid phases within the soil/sediment. Such internal processes are important in determining the transfer of contaminants around the biosphere and need to be captured in the conceptual model, either explicitly within the diagrams or, potentially, within descriptions for each conceptual model component.

There is potential to further disaggregate the conceptual model components to ensure that processes intrinsic to some of the features are made explicit (e.g. by representing soil solids and soil solution as distinct conceptual model components). If components are disaggregated in developing the conceptual model of contaminant migration (transport and accumulation) and exposure, then the list of conceptual model components defined would need to be updated accordingly. This provides an example of the iteration needed in developing conceptual models.

The conceptual model needs to be structured such that it is readily translated into a mathematical model (Section 5.2), with associated supporting data (Section 5.3). The longer the timescale of an assessment, the more illustrative the results become because uncertainties increase with time. Acknowledging the significant uncertainties associated with long term assessments helps to constrain the level of detail included within a conceptual model (Section 3.3).

An important component of developing the conceptual model of contaminant migration is to explicitly record judgements regarding the relative significance of processes, especially if they are excluded from the model based on such judgements. Such decisions can be supported by qualitative and/or quantitative arguments. Examples of quantitative evidence include side calculations, previous assessment iterations and/or reference to other evidence in the literature. Qualitative arguments may include those relating to natural analogues or expert judgement.

5.1.3.2. Conceptual model for exposure

To complete the conceptual model, human and biota exposure pathways need to be identified and associated with each of the environmental media represented in the conceptual model for contaminant transport. Assumptions relating to the exploitation of biosphere resources by the human community and the presence of biota populations of interest will have been defined within the description of the specific biosphere system. These can be related to the components within the conceptual model, either by annotating the diagrams developed (see example shown in Fig. 9) or separately. If exposed organisms (humans and biota) are included within the conceptual model (e.g. as diagonal elements of an interaction matrix), then the representation and annotation of exposure pathways linking environmental media to exposed organisms will occur naturally.

Explicit methods for exposure pathway analysis have also been developed in recent years, framed in a broader contaminant context than radioactive waste (e.g. see Ref. [85]). Depending on the assessment context, a good practice might be to consider modes of exposure for chemicals and radionuclides together, and this issue has been addressed within a BIOPROTA project [86].

As with other system characteristics, the systematic selection of exposure pathways as part of the overall conceptual model for radiological assessment is undertaken by including, where appropriate, modelling judgements regarding the relative significance of specific pathways.

5.1.3.3. Review and justification of the conceptual models

The approach of building a conceptual model on the foundation of a description of the biosphere system being represented and within the context of the overall assessment is an iterative process and aims to help ensure that the model is fit for purpose. The model itself could have been developed from a generic or project specific FEP list (a bottom up approach), or it could have been developed from the system description based on expert judgement (a top down approach). In either case, once developed, the conceptual model needs to be reviewed to help build confidence that the potentially important FEPs have been adequately addressed.

Where a top down approach has been employed, there is potential for the conceptual model to be checked against a generic (e.g. see Ref. [18]) or a project specific FEP list (e.g. see Ref. [87]). The screening of generic FEPs and EFEPs for relevance to a specific assessment is discussed in Ref. [26]. Reasons for screening out components of the conceptual model can include low probability of occurrence, low radiological impact, irrelevance to the site of interest, and being excluded due to the assessment context (e.g. for regulatory compliance).

For each of the conceptual models that have been developed (recognizing that different conceptual models could have been developed for different scenarios), the relevance of FEPs can be reviewed. For each FEP that is deemed relevant to the conceptual model in question, its inclusion/representation can be checked. Such a review needs to be transparently documented, including arguments as to why FEPs are considered irrelevant and explanations of where any relevant FEPs are captured in the conceptual model.

There is potential for such checking to highlight FEPs that may have been overlooked and it may uncover configurations or interactions that have not been considered. Such findings are helpful, as they enhance the rigour of the assessment. The conceptual model needs to be updated in response to any such findings and the implications of any changes considered.

In addition to auditing against a FEP list, the conceptual model needs to be reviewed and refined. One consideration in refining the conceptual model is that, subject to the overall requirement to satisfy the purposes determined by the assessment context, the model generally needs to also be as simple as can be justified (Section 3). Additional complexity that does not lead to a meaningfully improved estimate of the assessment end points needs to be screened out.

An example of such refinement may be that there is no need to explicitly represent diffusion in a groundwater pathway dominated by advection. Diffusion in groundwater may therefore be explicitly identified in the conceptual model but could be justifiably excluded from the mathematical model based on its limited significance. Such arguments can be supported by quantitative evidence (e.g. side calculations), reference to previous assessment iterations where the case is demonstrated and/or reference to other assessments or analogues. It is important to record the screening process and the justification used to ensure transparency of the model development process and to help ensure that the target audience or audiences for the assessment can follow the associated logic.

The review/refinement process may include individuals not directly involved in development of the model. There is also potential for stakeholders to be involved in reviewing conceptual models and the scenarios/system descriptions from which they have been derived. Such reviews will help to build confidence that important issues have not been overlooked and will also help to ensure a degree of understanding and buy into the safety assessment process.

5.2. MATHEMATICAL MODEL DEVELOPMENT

The conceptual model developed using the procedure described in Section 5.1 involves representation in a quantitative assessment model. Consideration will need to be given to the modelling approach. Because of the distinct media, spatial configuration and connectivity, and timescales typically considered in biosphere models, a compartment modelling approach is typically adopted, though finite element and finite difference approaches can also be considered, where there is a need to provide spatially continuous representations of the modelled biosphere. However, these are typically more relevant to detailed process models that can be used to underpin the assessment modelling approach. Only a compartment modelling approach is considered further in this publication.

A compartment modelling approach provides considerable flexibility in scale and resolution, allowing a model to be developed that has enough complexity to adequately represent the key components and processes that need to be considered. There is a need to include a sufficient degree of complexity so that the results obtained are not biased (e.g. by excessive homogenization when a small number of compartments is used), but not to make the model more complex than is needed by the assessment context (some homogenization can be justified by considering the potential exposure routes associated with e.g. harvesting and food processing for humans and home range area for biota). Each feature of the conceptual model is represented with one or more compartments and the events and processes relevant to the migration of contaminants in the biosphere represented as transfers between the compartments. Contamination could already exist in the compartments at the start of an assessment calculation or could be introduced into the model as a source from elsewhere. Losses from the biosphere can be directed to a 'sink' compartment, which can collect contaminants that migrate outside the region of interest, thereby satisfying balance. Although compartmental models can be deterministic in nature, it is also possible to use them probabilistically, sampling their input parameters from predefined probability density functions. General considerations in translating a conceptual model to a mathematical model with the associated propagation of uncertainties are discussed in Ref. [88].

In addition to describing equations for specific transfer and exposure processes, it is important for the underlying basis of any mathematical model to be clearly stated. If a compartment modelling approach is to be used, then the mathematical basis for that model needs to also be presented, with the number and characteristics of the individual compartments suitably justified. For example, the evolving amount of a contaminant N in compartment i (N_i, moles) can be represented mathematically by the following first order linear differential equation, if the transfers and losses can be represented as linear functions based on the amounts of contaminant present in the donor compartments [89]:

$$\frac{\mathrm{d}N_i}{\mathrm{d}t} = \sum_{j \neq i} \lambda_{ji} N_j + \sigma_{MN} \lambda_M M_i + S_i(t) - \sum_{j \neq i} \lambda_{ij} N_i - \lambda_N N_i \quad (1)$$

where

i and j — are compartments and j is effectively taken to represent compartments other than i ($j \neq i$);

N and M	are the amounts (moles) of contaminants N and M in a compartment (M is the parent of N in a decay/degradation chain);
λ_{ji}	is the transfer rate representing the loss of contaminant N from compartment j to i (a^{-1});
λ_N and λ_M	are the decay/degradation rates for contaminants N and M, respectively (a^{-1});
σ_{MN}	is the branching ratio for decay of contaminant M to contaminant N;
$S_i(t)$	is a time dependent source of contaminant N (moles a^{-1}), such as entering the biosphere from the geosphere, for example in groundwater flow or by groundwater abstraction and irrigation;
λ_{ij}	is the transfer rate representing the loss of contaminant N from compartment i to j (a^{-1}).

The first three terms of Eq. (1) represent a process that can add amounts of contaminant N to compartment i; the last two terms represent processes that result in a loss of contaminant N from compartment i.

Equation (1) allows time dependent amounts of contaminants in each compartment to be determined. There are some components of the conceptual model for which the degree of contamination can be considered to be in equilibrium with other components. This arises where transfer processes between those components and other components are typically rapid relative to processes that move contaminants around the biosphere system as a whole and where the uptake of a contaminant can be taken to not significantly diminish the amount in the associated compartment (this can be because the amount that is transferred is small or can be conservatively neglected). Typical examples include soil to plant uptake, direct contamination of plants via irrigation water, and intake of contaminants by PEGs and PEPs via ingestion and inhalation.

Equilibrium assumptions determine the number of compartments that need to be explicitly represented and the type of mathematical equations to be used. It is therefore helpful to screen the conceptual model to distinguish those components that can be considered in equilibrium with their environment from those for which explicit representation with compartments is needed (e.g. for soils, sediments, near surface strata and surface water). It is important that the rationale for equilibrium assumptions is documented based on the description of the biosphere system(s), so that the justification for the modelling approach is clearly communicated.

There is an extensive history of biosphere studies from which inspiration can be drawn for representing processes included in a conceptual model mathematically (see Annex I for some examples). It is beyond the scope of this publication to prefer one approach above another. For any safety assessment the approach and equations being used to represent contaminant transport and exposure in the biosphere need to be clearly explained and the level of complexity justified. Some modelling topics and issues of relevance to safety assessments are discussed in Sections 5.2.1–5.2.5.

During development of the mathematical model, a list of parameters relevant to the calculations will be identified. Each of these, and its specific meaning within the context of the model, needs to be documented to provide a clear basis for collating or obtaining the associated data. In practice, the mathematical representation of many processes tends not to be explicit but is instead based on an empirical model of effects observed at the system level. For example, the uptake of radionuclides by plants and other biota is often represented in terms of a concentration ratio, as noted above. An empirical model of this type can represent the combined action of several FEPs (e.g. root uptake and translocation), which are identified separately within an interaction matrix. Where this is the case, care needs to be taken to avoid double counting the effects of certain processes or, conversely, the inadvertent exclusion of potentially relevant FEPs. Exposure models typically comprise analytical expressions that relate concentrations of contaminants in environmental media to their impacts on human health and the environment.

The mathematical model can be audited against the conceptual model to help build confidence that the specification is complete. The conceptual model will provide a list of FEPs that need to be considered and the relationships between them (possibly expressed in an interaction matrix), which can be used as a checklist for the components of the mathematical model.

Sections 5.2.1–5.2.5 provide information on biosphere issues that typically need to be considered when developing and justifying mathematical models for the biosphere component of safety assessments. These include:

- The use of biosphere dose conversion factors (Section 5.2.1);
- The level of modelling detail/complexity that is appropriate (Section 5.2.2);
- The importance of discretization for compartment models (Section 5.2.3);
- The representation of short lived radioactive progeny in safety assessments (Section 5.2.4);
- Modelling of contaminants with special or distinct behaviours in the biosphere (Section 5.2.5).

It is noted that some of these issues are also of relevance to other steps in the methodology, for example issues of complexity and the potential need to take the behaviour of contaminants with special or distinct behaviours into account are of relevance to the conceptual modelling stage. This serves to highlight the degree of iteration that is inherently needed when developing and justifying biosphere models in support of safety assessments, especially between conceptualization and the associated mathematical model development. The topics are discussed here because they have been identified through experience as being of particular relevance to the development and justification of the mathematical modelling approach.

Supporting models or models of subsystems, may be used to justify simplifications in the overall biosphere model.

5.2.1. Biosphere dose conversion factors and integrated assessments

It may be appropriate to assess potential impacts of contaminant releases to the biosphere by modelling a constant input of contaminants to the biosphere through to equilibrium in the various components of the biosphere to give constant annual effective doses for humans or dose rates for biota. The ratios between the dose or dose rate end points and the input rates of the contaminants are typically termed biosphere dose conversion factors. Such ratios can be used if the biosphere is being modelled independently of other components of an assessment. Two considerations in determining whether such an approach is appropriate are whether inputs to the biosphere change only slowly compared with the timescale to achieve equilibrium and whether the characteristics of the biosphere also change only slowly compared with the timescale to achieve equilibrium.

These conversion factors, which can differ depending on the calculation case, can then be used to scale time dependent fluxes to the surface environment computed using other components of the overall disposal system model. This is typically a conservative (and potentially overconservative) approach for contaminants that take a long time to reach equilibrium in the surface environment compared with the period of peak release from the geosphere. Examples include biosphere modelling for disposal of higher activity wastes in the United Kingdom [14] and spent nuclear fuel in Sweden [90].

Conversely, if the biosphere is being represented in an integrated manner with other components of the disposal system and/or if the system description identifies a need to explicitly represent biosphere change, then it is not generally appropriate to use an equilibrium approach for the mathematical model. Rather, dynamic calculations within the biosphere model are needed that employ time dependent fluxes obtained using other components of the overall disposal system model. Nevertheless, even in these cases, there will be uses for calculations based on constant input fluxes to the biosphere, for example to examine equilibration times in different components of the biosphere to determine, in retrospect, whether a time dependent biosphere model was needed or whether a time independent biosphere state model would have been sufficient. Thus, it may be necessary to go a step beyond the final aim in biosphere modelling to demonstrate that a simpler approach is justified and leads to similar results. Examples of integrated biosphere assessment include studies for low and intermediate level radioactive waste in Canada [46] and in Sweden [91].

5.2.2. The level of detail and complexity in the biosphere part of safety assessments

Given the uncertainties associated with the biosphere on timescales relevant to post-closure safety assessment, it is important that the aim is that the associated models are no more complex than is necessary. This raises the question of what level of complexity is appropriate. There is no simple answer to this question. The BIOMASS methodology places strong emphasis on defining the context for each assessment, which then helps to guide and justify assumptions that are inevitable when modelling contaminant migration and potential exposure in the biosphere on long timescales. For assessments that build on previous studies, the results and experience gained in identifying key contaminants, processes and pathways can help to refine the level of detail and complexity in subsequent iterations. Previous assessments can provide justification for simplifying the representation of specific processes and/or pathways where they have been shown to be unimportant. Such evidence needs to be quoted explicitly when justifying the modelling approach being adopted. The degree of complexity in modelling approaches and the simplifying and conservative assumptions employed need to be justified in safety assessments.

Although it is important that the aim for assessment models for the biosphere is simplicity, they could be supported by much more complex models. This is particularly the case where site characterization information requires interpretation. A complex model could then be used to fully take into account the detailed spatial, temporal and multiparametric information available. However, an important consideration is then the need to simplify this model for application in an assessment context. This may be done by developing a simplified abstraction of the complex model that is suitable for incorporation in the assessment model. Alternatively, the complex model could be used to compute effective parameter values for use in the assessment model. In undertaking such simplification, it is important to ensure that all key aspects of the response of the complex model to different combinations of input data are appropriately represented.

There is potential to build confidence in the way in which the biosphere is represented in post-closure safety assessments through comparison of alternative modelling approaches. For example, Posiva has used simple biosphere models to cross check the results of their 2012 assessments [92]. Peer review can also build confidence, conducted internally as part of the safety assessment programme and/or formally as part of regulatory and/or formal international peer review exercises (e.g. see Ref. [93]). Independent modelling can also be conducted in support of regulatory review of safety assessments, as has been the case for SSM's review of the SR-Site assessment in Sweden [94–96].

5.2.3. Discretization

A key issue for compartment models is the degree of discretization that is used to reflect each component of the conceptual model that is to be explicitly represented with compartments. Contaminants within a compartment are effectively universally and uniformly available throughout the region of the model represented by that compartment. For transfer processes, this 'numerical dispersion' can result in faster migration and dispersion, and thus also in lower peak concentrations than would be expected in practice. Numerical dispersion in compartment models can be managed by discretizing the model appropriately (e.g. by representing an advective groundwater pathway with five compartments instead of a single compartment). Such discretization can provide a more appropriate representation of the time dependent migration and breakthrough of a contaminant in a transient biosphere representation. The time dependence is important regarding the residence time for contaminants in each part of the system, and hence the potential for decay, degradation and ingrowth for radionuclides and other contaminants, especially where progeny can be more radiotoxic than their parents. It is particularly important for biosphere systems that are evolving over time; the benefit of taking care to characterize the timescale of changes is lost if the timescales for contaminant migration and accumulation are not treated with the same degree of attention. Further guidance on discretization of compartment models is available elsewhere (e.g. see Refs [97, 98]).

Consideration of spatial scale is also an important factor when discretizing biosphere models, including the spatial scales of the distinct features/media to be represented (such as different surface water features and different soil types), as well as the spatial scales for evaluating the end points (e.g. human and biota habits). If only part of the home range (area for living and moving) for humans and/or biota can be contaminated, then there is no need to model the uncontaminated regions explicitly, so long as occupancy factors and intakes of contaminants are suitably specified to reflect the fraction of the area providing resources to the PEGs and PEPs considered. Also, if the human and biota habits mean that exposure will be averaged over certain spatial scales (e.g. harvesting crops across an entire field), then there is no need to impose a greater degree of discretization of those areas, so long as average properties can adequately represent the potential for contaminant migration and accumulation. When considering biota exposures, it is important to recognize that it is the spatial scale over which the biota populations range that is of interest, not the range of individual organisms [82]. The spatial scales of the biota populations of interest need to be defined when the PEPs are identified (see Section 5.1.2.2).

5.2.4. Short lived radioactive progeny

Another aspect typical of biosphere models for solid radioactive waste disposal is the potential to assume that some radionuclides are present in secular equilibrium with their parents. Some radioisotopes have half-lives that are very short in comparison with the characteristic timescales of the transfer processes that move contaminants around in biosphere systems. In such cases, they can be considered to be effectively present in secular equilibrium with their parent (i.e. at the same activity concentration) and their transport need not be explicitly modelled. Where this simplification is adopted, it is important that the contribution of any progeny assumed to be present in secular equilibrium is accounted for when assessing calculated end points, such as effective doses. The contribution of the short lived progeny can be explicitly added to that of their parent radionuclide. Strontium-90 provides a case in point, as illustrated in Box 1.

BOX 1. EXAMPLE OF SECULAR EQUILIBRIUM TREATMENT OF SHORT LIVED RADIONUCLIDES

Strotium-90 decays to the stable isotope ^{90}Zr via ^{90}Y, as shown below based on Ref. [99].

Parent	Half-life	Branching fraction	Daughter
Sr-90	28.79 y	1.000	Y-90
Y-90	64.10 h	1.000	Zr-90 (stable)

At 64.1 h, ^{90}Y has a half-life that is short compared with the timescales of transfer and sorption processes that are typically modelled in post-closure safety assessment, which can distinguish its behaviour from that of its parent, ^{90}Sr. Yttrium-90 is, therefore, typically taken to be present in secular equilibrium with ^{90}Sr, such that ^{90}Y is not explicitly modelled in the radionuclide transport calculations. Each becquerel of ^{90}Sr is instead taken to be accompanied by a becquerel of ^{90}Y. When the modelled environmental concentrations of ^{90}Sr are converted to potential doses to humans and other biota, it is important that the

contribution of ^{90}Y is explicitly added to that of the parent. Ingrowth of ^{90}Y in the body is explicitly included in the dose coefficient from ^{90}Sr, but the ingrowth of ^{90}Y in material inhaled or ingested by the individual is not included in the dose coefficient and has to be addressed in the biosphere model.

Short lived progeny cannot always be assumed to be in equilibrium with their parents, because they can be subject to transport processes that have timescales that are short compared with their half-lives. For example, ^{222}Rn has a half-life of only 3.8 days but its characteristic residence time in unsaturated soil is only a few hours, so it will not in general be in equilibrium with its parent ^{226}Ra in such soils.

5.2.5. Modelling contaminants with distinct behaviours in the biosphere

The same biosphere processes will be relevant to many of the trace contaminants that can be released into the biosphere in the long term as a result of solid radioactive waste disposals. This allows the same mathematical models to be applied, albeit with differing parameterizations reflecting the different physicochemical properties of each contaminant.

However, some contaminants exhibit specific behaviour in the biosphere that need special consideration. In developing conceptual and mathematical models for contaminants in the biosphere, consideration needs to be given to the suitability of models for the full range of contaminants that are specified in the source term to the biosphere and/or that can arise within the biosphere as a result of contaminant releases over assessment timescales. Examples of contaminants for which special considerations arise are given below. It is emphasized that this list is not comprehensive, though it does serve to emphasize the sorts of characteristic that may need to be given special consideration.

— Hydrogen-3: Hydrogen is of fundamental importance to life; ^{3}H can therefore be present in significant quantities throughout the biosphere in various chemical forms (including tritiated water and organically bound forms), each of which can be subject to different environmental behaviour [100].
— Carbon-14: Carbon is a fundamental component of organic compounds and of biota; it is present in significant quantities through the biosphere in differing chemical forms (including gaseous forms, such as CO_2 and CH_4, as well as organic and inorganic substances), each of which can be subject to different environmental behaviour. The behaviour of ^{14}C and its

representation in assessment models have been the subject of a series of BIOPROTA reports (e.g. see Refs [101, 102]) and in reports from waste management organizations (e.g. Ref. [103]). This work has resulted in an improved understanding of ^{14}C behaviour within media relevant to radiation dose assessment over relevant temporal and spatial scales, reducing previously identified significant uncertainties. Application of the 'specific activity' models that are typically adopted both for ^{3}H and ^{14}C can be justified through a sufficient understanding of these scale issues for the system(s) under consideration; that is, if the characteristic timescales for equilibration with the stable element 'pools' of relevance are considered.

— Chlorine-36: Chlorine can be present in the environment as chloride or in organic forms. Chloride is readily available for root uptake but in its inorganic form is poorly retained in soils. In contrast, organic forms of chlorine can be well retained in soils but are not very available for plant uptake. In addition, some compounds of chlorine are subject to volatilization from soil–plant systems. In mammals, chloride is subject to homeostatic control, with retention in the body varying inversely with stable chloride intake. The behaviour of ^{36}Cl and its representation in assessment models has been considered in BIOPROTA [104–106] and in Ref. [107].

— Selenium-79: Selenium is an essential trace element for animals but is also a biochemical analogue for sulfur. It is present in the environment in multiple oxidation states, with oxidized forms being much more mobile than reduced forms. It is also susceptible to volatilization from soils. The behaviour of ^{79}Se and its representation in assessment models has been considered in a series of BIOPROTA reports [108–110].

— Iodine-129: Both iodide and iodate can be present in surface waters, and iodine is also readily incorporated in organic matter and can become immobilized in organic sediments. Volatile forms include both iodine vapour and methyl iodide. Iodine is an essential trace element in mammals, being strongly preferentially taken up in the thyroid and subject to both short term and long term homeostatic control.

5.3. APPLICATION OF DATA

The mathematical model is associated with a set of input parameters that will need to be assigned point values or distributions in the safety assessment. The models, therefore, need to be developed with a view to the availability and quality of data with which to support them; a large degree of iteration and integration between site characterization and safety assessment teams is needed during these two tasks within the methodology. In particular, it is important that model input parameters can be related directly, or indirectly, to measurable properties of the environment and that the increased site understanding is used as the waste management programme moves forward in the stepwise siting process.

The collation of internally consistent data sets, including definition of uncertainty bounds or distributions and dependences, is an important component of safety assessment. The derivation of assessment specific parameters from a wider database of information on surface systems is a general challenge in environmental modelling. In long term safety assessments, the difficulties are compounded by the need to represent both the existing surface environment (where this is known and remains relevant) and potential surface environments that may be present in the future.

The description of the surface environment will provide data needed to support the mathematical model, particularly in relation to the spatial configuration of the biosphere, its physical characteristics, and the dynamics of masses of water, air and solids, as well as the way that it may evolve into the future (if biosphere change needs to be explicitly represented). Where data are drawn from the description of the surface environment, it is important that this is clearly explained and justified, including any interpretation that may be needed to match the format of the mathematical model.

The assessment context will define the approach being adopted to manage parameter uncertainty, including the potential for propagating uncertainties by explicitly representing feasible distributions of values for individual parameters (probability density functions) via probabilistic modelling. Another means is to formulate alternative datasets for deterministic calculation cases exploring uncertainties. The choice between a probabilistic and a deterministic approach will determine the sort of information that will need to be gathered. In some assessments, a mix of deterministic and probabilistic calculations can be undertaken. Whatever approach is used, it is important to keep in mind that the aim of uncertainty and sensitivity analyses is to enable stakeholders to appreciate the degree of robustness of the results of the assessment and to identify topics worthy of further investigation to reduce or clarify the uncertainties.

Biosphere models for radioactive waste disposal typically involve extensive databases due to the range of contaminants and potential exposure pathways that are addressed. Some parameters will depend on site characteristics (e.g. the spatial configuration of the biosphere, properties of environmental media). Other parameters will not depend on the specifics of a site (e.g. effective dose coefficients for humans and dose coefficients for biota). Some parameters could require both site specific and generic data to be assimilated (e.g. distribution coefficients and concentration ratios). Where data are not prescribed, the values and distributions that are used in the assessment need to be justified. Focus needs to be on those data that are important to the outcomes of the assessment.

A possible approach to identifying those parameters that are important to the outcomes of the assessment is illustrated in Ref. [111]. A protocol can also help guide and explain the approach to data selection for safety assessment, as illustrated in Fig. 10.

International compilations and recommendations can be helpful when collating data for biosphere modelling. For example, radiological assessments typically make use of ICRP recommendations regarding dose coefficients (e.g. Ref. [112]) and the IAEA provides international compilations of data that can be used for representing sorption and transfer in the biosphere (e.g. see Refs [8, 113]). Careful consideration is needed to help ensure that data are appropriate for the biosphere system(s) and model formulation adopted in any

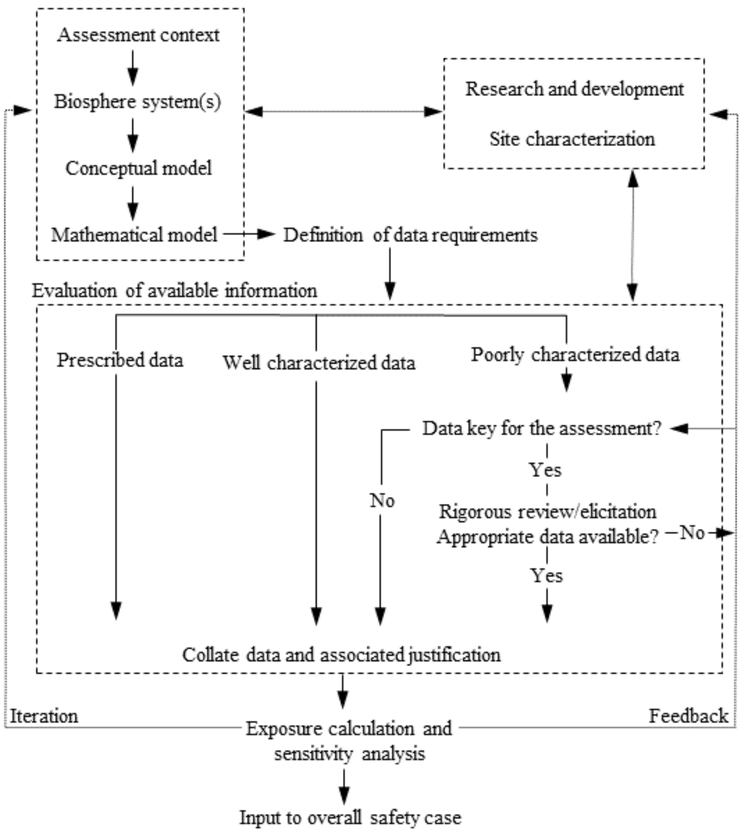


FIG. 10. Relationship between data types, data quality, data availability and data requirements (adapted from Ref. [6]).

specific assessment. Also, assessments may need to strike a balance between using site specific information, where available, versus international compilations.

Data management needs to be considered from the beginning of any safety assessment, especially due to the influence it can have on model developments and because it can be resource intensive. Its treatment needs to be explicit and properly documented, with the aim of trying to avoid confusion and potential loss of information. Data chosen to represent the biosphere in the safety assessment need to be clearly documented and justified, including traceability back to their original sources. There is a need to ensure not only that individual data items are well justified, but that they constitute a coherent set of information, taking correlations between various types of data into account. Quality control and assurance also help build confidence that the correct data have been transcribed from source for use in the assessment (this is part of verification).

5.4. IMPLEMENTATION, VERIFICATION AND VALIDATION

In implementing a mathematical model for the biosphere in safety assessment, consideration will need to be given to the solution method to be chosen. Very simple models may only involve a spreadsheet calculation. However, once the complexity of a model extends beyond a few key features and in contexts for which the time dependent history of contaminant concentrations is necessary, a numerical modelling tool will be needed. Several numerical modelling tools for modelling contaminant behaviour and exposure are available. Important considerations in choosing a software tool include:

- Availability, ease of use, level of support, robust numerical solvers, assurance of software quality/verification, awareness of units and checking for consistency, suitability for automation;
- Transparency of models implemented in the tools, both to help in implementing models suited to the assessment context and to help facilitate quality assurance and review;
- Probabilistic and deterministic modelling capabilities (depending on the approach being adopted);
- Capability to communicate the implementation and support for outputting results.

Those responsible for implementing safety assessments need to review the capabilities of the tools that are available and select the most appropriate for the given context.

Model validation is defined as the process of determining whether a model is an adequate representation of the real system being modelled by comparing the predictions of the model with observations of the real system [114]. Model verification is the process of determining whether a computational model correctly implements the specified conceptual model and/or mathematical model. Moreover, it has been noted that the radiological assessment models with the lowest potential for quantitative validation appear to be mainly those used in assessments of the long term impact of geological disposal of solid radioactive wastes [43]. However, the scope for partial validation is potentially important, for example with respect to specific processes and radionuclides, combined with consideration of analogues [115].

Safety assessments will typically involve consideration of several different scenarios, as well as variant calculations exploring the sensitivity of the results to alternative assumptions on conceptual and mathematical model structure and parameterization. This can result in many calculation cases within which much of the structure of the models and many of the data remain the same. Implementing multiple calculation cases within a single model file makes it easier to manage the data consistently between calculation cases and is also more efficient in comparison with having to maintain multiple calculation files. For large numbers of calculations, models can typically be run in an automated manner using script languages.

The same model, with identical input parameter values and discretization (as appropriate) implemented in different numerical tools ought to give the same result. Nonetheless, there is an element of uncertainty in the numerical calculations (e.g. effects of coarse discretization and residual errors from matrix inversion techniques) that could, for example, be explored by implementation in different codes and/or using side calculations to help build confidence in the quantitative results. For example, implementation of models in different calculation tools has been found helpful in support of regulatory reviews (e.g. see Refs [94–96]).

As with earlier steps in the BIOMASS methodology, maintaining an audit trail for the model that is implemented is important for maintaining transparency and building confidence and understanding in the results. The implementation needs to match the specification of the model. Where modifications to the mathematical model are made to facilitate efficient implementation, the model specification needs to be updated to reflect such changes. Furthermore, the overall implementation of the model structure, equations and data needs to be audited against the specification (verification) using documented processes for these quality assurance checks. This is important for building confidence in the assessment as a whole.

The timescale over which safety assessment models are deployed and the level of understanding of some of the processes included over these timescales mean that strict validation of the models against observations is not feasible. Nevertheless, a limited degree of validation of some parts of a model against observations could be possible (e.g. validation of modelling uptake of ^{14}C from the atmosphere to plants in Ref. [116]). Where this is the case, such validation is an important aspect of building confidence in the model.

6. MODEL APPLICATION AND EVALUATION OF RESULTS

Once a model has been implemented, and quality assured and validated to an appropriate level, it can be applied to the range of scenarios and calculation cases to be addressed in the assessment. This section provides some guidance on undertaking calculations and evaluating and communicating results, drawing on experience from historical assessments.

6.1. UNDERTAKING CALCULATIONS

Assessment studies typically encompass a range of calculation cases to help manage and explore the degree of uncertainty that is inherent in undertaking projections over long time frames. The calculation cases can encompass different future evolution scenarios, variant calculations within each scenario and 'what if?' style calculations that all contribute to building confidence that a sufficient degree of understanding has been established for the context of the assessment. Care is needed in tracking the range of calculation cases, their motivations and the iterations that are inevitable as results are explored and cases refined.

Any choices made in undertaking the calculations need to be documented and justified. This includes, for example, the choice of numerical solver used (where applicable and where multiple options are offered) and the time steps specified for the numerical solver (if they are not automatically selected by the computational tool itself).

Calculation files and details of any post-processing need to be retained as part of a quality assurance system. The input data files, any side calculations and the results also need to be independently checked to help ensure that reported results accurately reflect the specification for each case.

6.2. EVALUATING RESULTS

In addition to the verification and validation procedures discussed in Section 5.4, there is a need for scrutiny of the assessment results to determine whether they are generally plausible and can be explained by the characteristics of the model used. It is often found that, whereas it is difficult to forecast the results to be obtained from a complex model, when such results have been obtained, a deductive approach can be used to determine why they arose. Sense checking of results is an important part of building confidence in the overall assessment, providing further reassurance in the implementation and suitability of the modelling for the given assessment context.

Exploration of results can start with the main assessment end points identified in the assessment context (see Section 3.2), typically identifying key contaminants and associated timescales. The transparency provided by the models developed and implemented following a BIOMASS type approach enables the main exposure pathways and transport mechanisms to be identified and traced back to the contaminant source terms.

Intuitive results help to build confidence in the assessment as a whole. There is potential for unanticipated results to arise, such as unexpected contaminants being of importance (e.g. see Ref. [117]). Such results will likely trigger greater scrutiny of the modelling, though they do not necessarily mean that there has been a mistake. Evaluating contaminant behaviour and pathways will help to explain the results. Such understanding can then be communicated through to the target audience.

Having identified important contaminants and pathways, the associated modelling assumptions can be reviewed to help ensure that the assessment remains fit for purpose. The assessment context and the magnitude of results will determine the extent of such reviews. If headline results are close to or greater than an associated criterion, then there is greater motivation to explore the modelling assumptions than if results are many orders of magnitude below a threshold of interest. Early cautious assumptions in conceptual representation or data needs may therefore, during iterative evaluations, have to be revisited and the assessment needs updated.

As calculations are undertaken and results evaluated, there is inevitably a degree of iteration. It is important that any such changes are properly fed back through the methodology. As the biosphere model is adapted and refined, it is important that such changes are properly documented, otherwise there is a danger that the documented model is not the same as the model that is implemented. Such a disconnect would represent a significant risk for the credibility of the overall assessment.

6.3. COMMUNICATING RESULTS

The value of a thorough and robust safety assessment will be diminished if the results and associated degree of confidence are not communicated appropriately to the wider safety assessment and target audience.

The methodological approach embodied in this publication helps to ensure that safety assessments are fit for purpose. The methodology provides a basis for clearly documenting and justifying assumptions and the basis of any results. Exploration of the performance of the biosphere model, described in Section 6.2, will also contribute to the way in which results are presented, helping to provide a chain of logic that builds understanding and confidence in the target audience.

The number of scenarios and calculation cases that could be explored in undertaking assessments has the potential to result in a large number of permutations. Evaluation of the results will help to identify important pathways and processes and could allow the ultimate number of cases that need to be fully explored to be rationalized. The overall assessment context remains an important point of reference when considering the presentation of results; it helps to maintain focus on the underlying purpose of the assessment and perspective.

For probabilistic assessments, the assessment context needs to provide guidance on the output that is relevant for comparison against any stated criteria. This can, for example, be the 'expectation value' or average calculated result. Alternatively, comparison can conservatively be made against other metrics (e.g. the 95th percent confidence in the mean or the 95th percentile of results). It is important for any comparisons to be made on a reasonable basis to help ensure that results are not overly pessimistic, especially as they approach safety criteria that may have been defined. Guidance on balancing realistic and conservative assumptions and on the metrics to be used for comparison against criteria could be included in discussion of the 'assessment philosophy' component of the assessment context (see Section 3.3). Annex B in Ref. [30] provides guidance on approaches to determining compliance when doses to members of the public are estimated probabilistically.

Assessment results at later times are increasingly indicative, given the increasing uncertainties. If the timescale of relevance to an assessment encompasses both shorter (tens to hundreds of years) and longer (thousands to tens of thousands of years, and longer) periods, then consideration ought to be given to explicitly distinguishing results in different periods of time. This can help to emphasize that results presented on long timescales can only be taken to be illustrative of potential impacts, rather than being inadvertently interpreted as predictions.

It is also important to consider the format of graphical outputs. Logarithmic scales need to be used with care because they can appear to overstate very small results. It is often helpful to place numerical results in context, for example by explicitly including background concentrations, fluxes and doses on charts. There is also merit in identifying a magnitude of result that is considered negligible to avoid too much weight being given to very small numerical results.

The assessment context will provide guidance on a range of end points to be considered in the assessment (see Section 3.2). These could include environmental concentrations and fluxes as complementary safety indicators to dose and risk.

The spatial scale of calculated results will be of interest to distinguish, for example, results over a small area from results over an extensive area. Therefore, it can also be helpful to visualize model results in a spatial context. This may include, for example, placing the results in a spatial context relating to the location of the facility and the present day and/or projected site/landscape, which could have a tangible meaning to target audiences.

7. CONCLUSIONS

The original BIOMASS methodology [6] has been shown to be useful in helping to ensure consistency and transparency in the way in which the biosphere is represented in post-closure safety assessments for solid radioactive waste disposal facilities. This publication enhances the original BIOMASS methodology to reflect experience gained and developments made in the period since 2001, when technical work on the original BIOMASS methodology was completed. The components of the original methodology have been retained but have been enhanced and some of the details have been restructured based on experience. This publication is therefore an updated and enhanced version that largely supersedes the original BIOMASS methodology publication; some of the supporting material in the original publication remains relevant, most notably the examples of generic example reference biosphere models.

Important experience has been gained in many areas since 2001, covering:

— Site characterization;
— Understanding of key processes that influence the headline assessment results, and hence areas for focused consideration;
— Approaches to addressing environmental change;
— Approaches for explicitly demonstrating environmental protection;
— Treatment of the geosphere–biosphere interface;

— Regulatory developments and regulatory review, and hence sufficiency of a safety assessment.

The resulting enhanced methodology is provided to assist organizations tasked with undertaking and reviewing safety assessments for radioactive waste disposal. Experience gained through assessments conducted following the original IAEA BIOMASS programme has shown that the overall steps in the methodology remain relatively consistent across assessments conducted in a range of different contexts. The detailed structure of assessments undertaken within different programmes can be expected to differ, depending for example on the assessment context and the overall safety case and safety strategy. Nonetheless, the methodology is useful in managing the uncertainties inherent in assessing safety over long timescales. Consistent with the original BIOMASS methodology, the context for each assessment plays a central role in guiding the approach and assumptions that are necessary (e.g. see Annex I for a list of published assessments).

Experience also shows that there need to be several iterations of the safety assessment during a programme of disposal facility development. In progressing such iterations, the following may be noted.

(a) Key indicators of safety can change as a result of the evolution of regulatory requirements and as a result of knowledge and evaluations of other parts of the disposal system. Focus on one narrowly defined radiological end point is an inadequate approach to managing post-closure safety, even from a basic regulatory compliance point of view. Therefore, the safety assessment methodology needs to be flexible and capable of responding to a range of radiological safety issues and end points.
(b) Key radionuclides that need to be considered can change, as source terms to the biosphere can change between iterations of the safety assessment. Such changes can arise, for example, due to different assumptions for degradation of near field barriers, but also from something as simple as new information about a half-life or the radionuclide inventory. Therefore, it is appropriate to maintain a methodology that is capable of addressing a wide range of radionuclides and other contaminants.
(c) New scenarios could need to be considered due to new interpretations of, or assumptions for, the interface between the geosphere and the biosphere.

The BIOMASS methodology has been strengthened in many ways, including:

— Integrating the biosphere part of the safety assessment within the iterative process of the broader safety assessment and any associated safety case;
— Drawing on updated understanding of long term environmental change in defining the systems to be modelled;
— Drawing on experience of contributing to and synthesizing site characterization and associated detailed modelling;
— Drawing on experience of defining potential exposure groups and potentially exposed biota populations;
— Recognizing the importance of stakeholder engagement in helping to define the assessment context and interpret assessments that address their specific interests and concerns.

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Annex I

EXAMPLES OF BIOSPHERE MODELS USED IN SUPPORT OF SAFETY ASSESSMENTS

This annex provides examples of biosphere models developed and applied in safety assessment for radioactive waste disposal since the original BIOMASS methodology was published [I–1]. In Table I–1 a wide range of assessment contexts are listed; from generic to site specific, from deep geological to surface based and for a wide range of biosphere systems and climate conditions (see also Annex III for discussions on case studies for different biosphere systems) [I–2–I–23].

It is also noted that the original BIOMASS publication [I–1] included generic example reference biospheres for a drinking water well, an agricultural well and for natural release of contaminated groundwater to the surface environment. The original example reference biospheres still provide relevant biosphere modelling illustrations.

It is emphasized that, in drawing on the previous example reference biosphere models and/or the biosphere models listed in Table I–1, care is needed to review and adapt relevant elements to suit the specific context being addressed.

Understanding and development of the studies referenced in Table I–1 have contributed towards the enhancement of the original BIOMASS methodology; the examples themselves predate the enhanced guidance documented herein and therefore, do not provide direct illustrations of its application. One exception is the safety evaluation SE-SFL, which was undertaken by Svensk Kärnbränslehantering AB (SKB) in Sweden in parallel to the work of the IAEA's Modelling and Data for Radiological Impact Assessments Working Group 6 (MODARIA II WG6), and which refers to this version of the BIOMASS methodology [I–24]. Nonetheless, the examples in Table I–1 provide illustrations of the way in which the steps of the guidance have been addressed for a range of different contexts and regulatory frameworks.

For details of the biosphere models and the ways in which they have been developed, justified and applied, the reader is directed to the associated reports. The studies and their contexts are summarized in Table I–1. The examples include both standalone descriptions of biosphere modelling and 'total system' assessments, where the approach to the biosphere is documented along with other components of the assessment (i.e. near field and geosphere).

Table I–1 lists assessment studies undertaken by radioactive waste management organizations. It is also noted that other stakeholders sometimes undertake modelling of the biosphere. In particular, independent biosphere modelling by regulatory bodies has been shown to help build understanding in scrutiny of licence applications (e.g. Refs [I–25–I–27]).

TABLE I–1. ASSESSMENT STUDIES UNDERTAKEN IN RADIOACTIVE WASTE MANAGEMENT PROGRAMMES

Country	Waste context	Sequential or non-sequential	Biosphere dose factors or integrated	Site specific	Biosphere systems	Climate(s)	Biota dose	References
Belgium	Surface disposal of LLW[a] and short lived ILW[b]	Non-sequential	Biosphere dose factors	Yes	Terrestrial, freshwater	Temperate, warmer wetter, warmer drier	Yes	[I–2]
Canada	Geological disposal of L&ILW[c]	Non-sequential	Integrated	Yes	Lake, wetland, freshwater, terrestrial	Temperate/boreal, tundra	Yes	[I–3]
Canada	Geological disposal of SF[d]	Non-sequential	Integrated	No	Terrestrial, freshwater	Temperate	Yes	[I–4, I–5]
Finland	Geological disposal of SF	Sequential	Integrated	Yes	Coastal sea, forest, lake, wetland, terrestrial	Temperate, periglacial	Yes	[I–6, I–7]
Finland	Geological disposal of L&ILW	Sequential	Biosphere dose factors	Yes	Coastal sea, lake, terrestrial	Temperate, periglacial	Yes	[I–7]

TABLE I–1. ASSESSMENT STUDIES UNDERTAKEN IN RADIOACTIVE WASTE MANAGEMENT PROGRAMMES (cont.)

Country	Waste context	Sequential or non-sequential	Biosphere dose factors or integrated	Site specific	Biosphere systems	Climate(s)	Biota dose	References
France	Geological disposal of HLW[e] and L&ILW	Non-sequential	Biosphere dose factors	Yes	Terrestrial	Temperate, humid subtropical	No	[I–8]
Spain	Geological disposal of HLW	Non-sequential	Integrated	Yes	Terrestrial, freshwater	Mediterranean, desert	No	[I–9, I–10]
Sweden	Geological disposal of SF	Sequential	Biosphere dose factors	Yes	Coastal sea, lake, wetland, terrestrial	Temperate, periglacial	Yes	[I–11]
Sweden	Shallow geological disposal of L&ILW	Sequential	Both	Yes	Coastal sea, lake, wetland, terrestrial	Temperate, periglacial	Yes	[I–12, I–13]

TABLE I–1. ASSESSMENT STUDIES UNDERTAKEN IN RADIOACTIVE WASTE MANAGEMENT PROGRAMMES (cont.)

Country	Waste context	Sequential or non-sequential	Biosphere dose factors or integrated	Site specific	Biosphere systems	Climate(s)	Biota dose	References
Sweden	Geological disposal of long lived L&ILW	Sequential	Integrated	Yes	Coastal sea, lake, wetland, terrestrial	Temperate, periglacial	No	[I–14]
Switzerland	Geological disposal of SF and ILW	Non-sequential	Both	No	Terrestrial, freshwater	Temperate, warmer drier	No	[I–15, I–16]
United Kingdom (UK)	Geological disposal of higher activity wastes and SF	Non-sequential	Biosphere dose factors	No	Terrestrial, freshwater, estuarine, coastal and marine	Temperate, warm arid, warm humid, boreal, periglacial	No	[I–17, I–18]
UK	Near surface disposal of LLW	Non-sequential	Integrated	Yes	Terrestrial, freshwater, estuarine, coastal, marine	Temperate, warmer wetter	Yes	[I–19]

TABLE I–1. ASSESSMENT STUDIES UNDERTAKEN IN RADIOACTIVE WASTE MANAGEMENT PROGRAMMES (cont.)

Country	Waste context	Sequential or non-sequential	Biosphere dose factors or integrated	Site specific	Biosphere systems	Climate(s)	Biota dose	References
UK	Near surface disposal of LLW	Sequential	Integrated	Yes	Terrestrial, coastal, local marine	Temperate, warm	No	[I–20]
United States of America (USA)	Geological disposal of HLW	Non-sequential	Biosphere dose factors	Yes	Terrestrial, freshwater	Arid, monsoon, glacial transition	No	[I–21]
USA	Geological disposal of HLW	Non-sequential	Biosphere dose factors	Yes	Terrestrial, freshwater	Arid, monsoon, glacial transition	No	[I–22]

[a] LLW: low level waste.
[b] ILW: intermediate level waste.
[c] L&ILW: low and intermediate level wastes.
[d] SF: spent fuel.
[e] HLW: high level waste.

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Annex II

COMPILATION OF EXPERIENCE RELATING TO POTENTIALLY EXPOSED GROUPS AND POTENTIALLY EXPOSED BIOTA POPULATION DEFINITION

As part of the programme of work of the IAEA's Modelling and Data for Radiological Impact Assessments (MODARIA II) Working Group 6 (WG6), the principles for defining potentially exposed groups (PEGs), representative persons and potentially exposed biota populations (PEPs) were explored [II–1]. To collect the experience of participating organizations, a questionnaire was designed to collect information on the topics associated with the definition of PEGs and PEPs in the construction of the biosphere models for evaluation of radiological impacts.

The objective of the questionnaire was:

(a) To review the current status and ongoing discussions on the handling of issues related to the definition of PEGs within the construction of the biosphere model(s);
(b) To provide a clear overview of the progress that has been made since the original BIOMASS methodology;
(c) To provide a clear overview of the feedback and lessons learned from application of the BIOMASS methodology in the development of safety cases;
(d) To identify areas in which further cooperation at the international level is desirable;
(e) To gather information on the selection of PEPs in the context of dose assessment and protection of the environment.

II–1. CHARACTERIZING POTENTIALLY EXPOSED GROUPS AND POTENTIALLY EXPOSED BIOTA POPULATIONS

The characterization of PEGs and PEPs typically starts with consideration of how humans and other biota could interact with contaminated environmental media such as soils, water and foodstuffs so that they are potentially exposed to radionuclides in the environment.[1] Table 1 in Section 6.1.2.1 of the main text provides examples of potentially relevant human activities; such examples

[1] The same approach may be used for exposure to other contaminants.

can be a useful guide. However, it is important to base thinking about possible exposures on the narratives developed for the specific assessment, and those narratives need, in turn, to be based on system understanding, including the scope for environmental change (see Section 5.1.2).

Potential exposure pathways include internal exposure from inhalation and ingestion of radionuclides and external irradiation.[2] Typical parameters that characterize the degree of exposure include occupancy of contaminated areas, physiological parameters such as breathing rate, amounts of different types of food consumed, and age at time of intake (for ingestion and inhalation). It is important to note that extreme parameters for intakes of radionuclides might imply physiological characteristics that are inconsistent with those used for reference persons in the calculation of dose coefficients.

Figure II–1 illustrates how people might interact with contaminated water leading to exposure. It is introduced here to help with interpretation of the answers to the questionnaire, as set out below, and is not to be used as a general template without consideration of the assessment specific narratives.

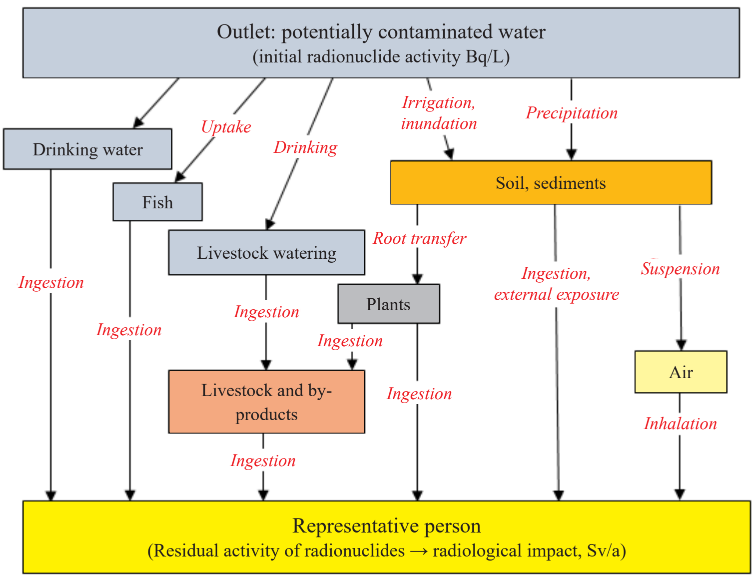

FIG. II–1. Schematic illustration of how people may interact with contaminated water.

[2] Skin absorption and intakes via damaged skin are not commonly considered but may be relevant within some narratives.

II–2. GUIDANCE ON PROVIDING RESPONSES

Questions were organized around the following five subject areas:

(1) Environment of the PEGs and its evolution;
(2) Linking the choice of PEGs to the environment as discussed in the previous subject area;
(3) Food habits and consumption rates;
(4) Populations of non-human biota (PEPs);
(5) National regulations and guidance and international recommendations and guidance.

Continuity in the assumptions for assessment at a site was also raised as an issue in the project discussions, from operation through to the end of regulatory and/or institutional control, and into the very long term. Therefore, the responses to the questionnaire were not to focus only on the post-closure safety evaluation.

It was emphasized that the responses or opinions provided needed to represent the view of the organization and not the individual answering the question.

II–3. QUESTIONS AND OVERVIEW OF RESPONSES

The five subject areas are highlighted in **bold** below and associated questions are highlighted in *italics*; questionnaire responses are then summarized.

II–3.1. Environment of the potentially exposed group and its evolution

Environmental changes: the environment that the PEGs are sitting in, and its evolution in the long term, is generally needed to define the PEGs and representative persons. How is the environment and its evolution described in the long term to support definition of PEGs?

Environmental changes over time are usually considered, including landscape and hydrogeological evolution and their influence on groundwater discharge areas.

The effect of global climate changes is commonly addressed through the description of a set of biospheres, each representative of a different climate state.

Dynamic consideration of climate change can also be considered.

Are environmental changes accounted for when characterizing PEGs, including climate effects? If yes, please describe briefly how.

The effects of climate variation are considered in PEG characterization and the habits. The parameters are selected to be representative of the evaluated or assumed climatic conditions and land use.

Hydrogeological evolution could influence the characteristics of the groundwater discharge area and influence human activities.

Assumptions for PEG characteristics such as food habits commonly follow the changes in climate and other environmental changes assumed or evaluated in the rest of the assessment. However, current behaviours, both locally and nationally, are also considered to be relevant to illustrate the significance of possible future releases.

Some organizations do not consider PEG changes in accordance with climatic evolution. This can be the result of narrower assessment contexts.

II–3.2. Linking the choice of potentially exposed groups to the environment and its evolution

Are environmental changes taken into account for the geosphere–biosphere interface (GBI)? If yes, please describe how they are accounted for within the definition of the 'radionuclide outlet' at the GBI (e.g. potentially contaminated water bodies where water can be extracted by humans or is being used for drinking by non-human populations).

A common response by participants was the need for a clear understanding of the GBI.

Consideration of the potential evolution of the GBI characteristics and location over time could be linked to climatic evolution.

Environmental changes are usually considered for the GBI characteristics, but not always.

Do the location and characteristics of the GBI (together with potentially contaminated water bodies) rely on data from a specific site? If yes, please describe briefly how it is accounted for in the definition of PEGs. If no, please describe the approach for consideration of the GBI for the definition of PEGs.

Responses here depended on whether the assessment is site generic or site specific.

For site specific assessments, the characteristics of the outlets are considered.

For a generic assessment, selection of GBIs and related PEG assumptions depends on the characteristics in the country. Account can be taken of generic approaches to treatment of the GBI.

The GBI or discharge area, the nature of the outlet and local human activities are an important starting point for identification of PEGs. Another factor is the productivity of natural ecosystems.

Depending on the productivity of the discharge area, several types of activity are considered, and several PEGs could be defined to explore different exposure pathways.

Is there a different approach according to availability of the potentially contaminated water (e.g. natural groundwater flow up to the surface, groundwater abstraction by pumping)?

Natural discharge areas and/or abstraction by pumping are considered.

Groundwater abstraction by pumping could be necessary in the assessment context, for example as a part of regulatory requirements or guidance.

At some sites, groundwater abstraction by pumping might be the only way for contaminated water to reach the surface within the time frame and/or spatial area of interest given in the assessment context.

How are potential gas or water releases due to direct human action/intrusion treated?

Exposure due to a drilled well is commonly considered for geological disposal and dose is usually calculated to a future inhabitant using the well as a water resource.

Consideration is also given to exposure of the geological investigation workers who could come into contact with contaminated drilled material. In this case, the PEG characteristics are linked to drilling activities and examination of drill cores.

Releases due to the discharges of radioactive gases to the soil zone are usually treated separately. The main radioactive gas of relevance is usually $^{14}CH_4$.

Do the principles for PEG definition account for current local human activities at the site or on a regional scale? If yes, please describe briefly how it is accounted for in the definition of future hypothetical PEGs. If no, please describe the approach for the definition of future hypothetical PEGs (e.g. use of analogue sites).

Current human activities are accounted for in the characterization of PEGs, for example local activities for site specific assessments and national data for generic assessments. Consideration can also be given to the potential for exploitation of the site rather than actual current activities, for example accounting for historic rather than current human activities.

In cases where the release occurs at times after environmental change, then characteristics of the PEGs can be modified in accordance with that change. For example, the modifications can be based on current activities today at other sites that reflect the anticipated change at the site of interest (i.e. using current analogues for the future of that site).

PEGs for the operational and post-closure phases can include different approaches: actual local observation of habits today versus hypothetical assumptions for the future based on current observations, including potentially relevant analogues, has been noted.

Are other age groups considered in addition to adults in the definition of PEGs and representative persons (e.g. explicit inclusion of children and infants)?

Some assessments have only considered adults, whereas others consider adults, infants (one year old) and children (10 years old). Some stakeholders could have an interest in understanding the implications for different age groups.

The International Commission on Radiological Protection (ICRP) provides dose coefficients for these three age groups, as well as others.

Is there any distinction in the approach for PEG selection and definition according to the scenarios used for building the safety case (e.g. normal evolution scenario, altered evolution scenario, disturbance scenario, what if? scenario)?

The approaches applied are generally considered applicable to all scenarios, with variations in time that correspond to environmental change. However, there can be differences for human intrusion, see above.

II–3.3. Human habits and consumption rates

How are the food habits of the representative person(s) constructed from each selected PEG? For example, are local observed food types (and habits) being considered (e.g. food surveys)? If not, please give the approach (e.g. are biosphere analogues being used, or are stylized approaches being considered?).

Usually, food habits consider relevant exposure pathways that result from utilizing natural resources and water extracted at the GBI. Different pathways (associated with different PEGs) are usually explored, so that there is an understanding of how one set of assumptions implies different results from another. There is also a significant interest in exploring different human activities and behaviours to give confidence to different stakeholders that a range of situations have been considered.

Food habits and consumption rate are usually based on national surveys and sometimes local/regional surveys for site specific studies (for the typical biosphere considered). See also Section 6.1.2.1 in the main text.

Is bioproductivity being considered?

As well as examination of the individual pathways, there is also an approach used by Svensk Kärnbränslehantering AB (SKB), Sweden that considers the overall productivity in the area of interest. It can, for example, set an upper limit on overall exposure from the food chain. Such information can also support assumptions regarding the proportion of food consumed by the PEG that comes from clean sources.

Are similar foods aggregated into a smaller number of types, for example leafy green vegetables are cabbage, broccoli, lettuce, etc.?

Aggregation of some foods is common. Sometimes, this is based on the foods being similar and where data on consumption habits do not distinguish between them. Other times, it reflects that the mechanisms for potential contamination of the food are similar, such as combining types of leafy green vegetable, as opposed to root vegetables or legumes, which do not become directly contaminated by irrigation water. Sometimes aggregation of foods is also considered due to lack of data on the transfer of radionuclides to individual foods.

Are observed (survey) food consumption rates used for each type of food, and at what level in the observed distribution of consumption, average or some percentile? Is specific dietary behaviour at 95th percentile or median value +2 standard deviations considered?

The following two distinct approaches are used:

(a) Consideration of local/regional/national surveys (i.e. use of statistical data), and this could include analogues for the future;
(b) Consideration of total carbon intake (and based on historical records).

Some consider that all food and water consumed by the PEG is derived from the contaminated areas. Others allow for use of uncontaminated food and water, for example based on observed current practice.

Some assume a cautious consumption rate (e.g. at the 95th percentile), while others consider average values, while assuming that all in the group consume all the food types.

The assumption of 95th percentile consumption rates for all foods is considered to be inconsistent with assumptions made in national surveys and would not correspond to present day habits or the assumptions for human metabolism used in ICRP dosimetric models.

How are food consumption rates adapted for specific activities (e.g. fishing, cattle farming, cereal farming)?

There are several examples where food habits are adapted to specific food consumption associated with specific activities (e.g. fishing, cereal farming, cattle farming). In these cases, more than one PEG is described, for example a population group that obtains the bulk of its food from one specific activity rather than another.

Is it assumed that all food is coming from the contaminated area, or some fraction? Are survey data considered, even for the long term?

It is usually considered that all food is coming from a potentially contaminated area but there are examples where a fraction of the food can be assumed to be uncontaminated.

How is the period of occupancy relevant to external exposure and inhalation decided?

Physiological characteristics such as breathing rates are taken from ICRP or similar high level recommendations. Types of occupancy are usually selected on the basis of the narratives supporting the identification of the PEGs. The period of occupancy (external and inhalation) relies upon ICRP recommendations.

II–3.4. Populations of non-human biota

Is the radiological impact on non-human biota assessed?

There is an increasing trend to include assessments of dose rates to non-human biota to support explicit demonstration of protection of the environment.

What specific end points are to be assessed as corresponding with environmental impact (as mentioned above) (biodiversity, populations of particular species, e.g. site specific)?

The end points for the assessment are dose rates to relevant populations of relevant species. Relevance is determined from the system description and narrative, including ecological factors. The focus is usually on protection of populations and not on individuals.

Dose rates are determined using models, such as those in environmental risks from ionising contaminants: assessment and management project (ERICA), to convert environmental concentrations into dose rates to reference animals and plants (RAPs).

The dose rates are then compared with derived consideration reference levels (DCRLs) provided by the ICRP for RAPs, as described in Section 6.1.2.2 of the main text, or similar quantities developed by other organizations.

Is the spatial scale of the biosphere system considered in relation to the population of interest?

There is not a great deal of variation in the approach to dose assessment for non-human biota. However, a potentially relevant factor that is not always considered relates to spatial averaging and is connected with ensuring that appropriate concentrations are used in the calculation of dose rates relevant to a population protection end point.

The spatial scale used to determine the concentrations to calculate the biota dose rates (i.e. the relevant area or volume over which to average the contaminant concentration) is sometimes based on areas used for human dose calculations. In other cases, consideration is given to the occupancy of a relevant population in such areas based on an understanding of the behaviour of the species under consideration. For example, the area considered for human dose assessment could be too small to support the population of interest for protection of the environment. Alternatively, concentrations are calculated based on the area that the selected population would use.

The issue here is to be clear on the population of interest that it is desired to protect. The participants have limited experience on this topic.

II–3.5. International recommendations and guidance and national regulations and regulatory guidance

Do the documents (national or international) that are applied address the issues raised in the questions above or answer them?

International recommendations and guidance are of significant support, both in developing national guidance and in interpreting it on an assessment specific basis.

Different disposal projects in different countries demonstrate different levels of experience and ambition in terms of detail and practical application.

Are there any additional factors requested or differences relative to issues raised in the questions above? If yes, please indicate specific (national) request and observed differences.

No single approach to PEG and/or PEP identification can be developed that will be comprehensively useful for all assessment contexts. However, the sharing of experience is useful for continuing development of recommendations and guidance as well as their application.

Experiences in using the BIOMASS methodology highlight the value in site characterization, site understanding and narratives for future evolution as a basis for the selection of PEGs and PEPs.

REFERENCE FOR ANNEX II

[II–1] GRIFFAULT, L., et al., Approaches to the definition of potentially exposed groups and potentially exposed populations of biota in the context of solid radioactive waste J. Radiol. Prot. **42** (2022) 020515, https://doi.org/10.1088/1361-6498/ac6045

Annex III

EXAMPLE OF DEVELOPMENT AND USE OF SUPPORTING INFORMATION IN BIOSPHERE MODELLING

Part of the Modelling and Data for Radiological Impact Assessments Working Group 6 (MODARIA II WG6) work included presenting and discussing the information needed to model the biosphere as part of a safety assessment. Examples and lessons learned from ongoing national programmes were also assessed in relation to the BIOMASS methodology. This annex addresses ways to acquire information to support biosphere modelling and discusses examples of how to assess uncertainties, long term climate and landscape change. Site characterization for the purpose of biosphere modelling, and how to deal with specific site specific conditions are also discussed. Annex III provides further information on topics related to but tangential to the BIOMASS methodology. This supporting information also helps to show how biosphere modelling links to other parts of a waste disposal programme, and how insights regarding the disposal system as a whole have a role to play when planning, conducting and justifying the dose assessment modelling.

The following examples are described on the basis of information supplied by members of the WG6 and these descriptions are not necessarily comprehensive. Furthermore, the basis for the material presented and the descriptions provided need to be treated as illustrative of approaches and assumptions that have been made to support biosphere modelling within safety assessments.

Those involved in biosphere modelling, safety assessment and safety case development in respect of radioactive waste disposal need to justify the approaches used and determine the level of detail and amount of data to be collected in accordance with the assessment context and a graded approach.

III–1. CONSIDERATION OF LONG TERM CLIMATE AND LANDSCAPE CHANGE IN THE MODARIA PROJECT

Between 2012 and 2015, the IAEA coordinated an international project addressing climate change and landscape development in post-closure safety assessments of solid radioactive waste disposal as part of the overall Modelling and Data for Radiological Impact Assessments (MODARIA I) project [III–1]. The work was supported by the results of parallel ongoing research that has been published in a variety of reports and peer reviewed journal articles. An overview of the work has recently been published [III–2].

The main activities undertaken in the project were the identification of the key processes that drive environmental change (mainly those associated with climate and climate change) and a description of how a relevant future could develop on a global scale; development of a methodology for characterizing environmental change that is valid on a global scale, showing how modelled global changes in climate can be downscaled to provide information that can be needed for characterizing environmental change in site specific assessments; and the illustration of different aspects of the methodology in a number of case studies that show the evolution of site characteristics and the implications for the dose assessment models.

The methodological approach that was developed within the MODARIA I project, together with the technical developments in long term climate modelling made in support of that project, complemented by the development of representations of landscape development in ongoing national programmes, jointly facilitate the consideration of climate change and landscape development within post-closure safety assessment that can be applied to a wide range of site and repository types, and at different stages of the development of a disposal facility, ranging from generic, initial studies to detailed site specific assessments. The methodology has been set out as a road map, as described in Ref. [III–3]. This provides a practical framework and common basis for future assessment work that is consistent with international recommendations and guidance, as well as the latest technical developments.

Global climate results at a 200 km scale were generated in the MODARIA I project for a wide range of carbon dioxide emission scenarios, ranging from no anthropogenic emissions to a prolonged 'business as usual' scenario, using a newly developed emulator underpinned by an ensemble of Atmosphere-Ocean General Circulation Model (AOGCM) runs. These global results could be used in any safety assessment programme worldwide in which the focus is on radiological impacts during the current interglacial episode. Ongoing studies sponsored by Svensk Kärnbränslehantering AB (SKB) and Posiva have the potential to extend this applicability to multiple glacial–interglacial cycles. However, even in the context of applying the results to the current interglacial episode, it would be desirable to explore uncertainties in these results by comparing them with emulators conditioned on the results from alternative AOGCMs. Also, whereas results at a resolution of 200 km could be sufficient for many assessment purposes, there could be circumstances in which downscaling of these results will be appropriate. Since such downscaling depends strongly on the local geographical context, the MODARIA project was not able to present results that are applicable in all locations worldwide. However, it was demonstrated how physical–statistical downscaling, the preferred method, can be applied to AOGCM results for the UK. A similar approach could be applied to results from a climate emulator at any location of interest to provide long time series of downscaled climatic information.

Once climate evolution data are available at appropriate spatial and temporal resolution, they can be used to drive landscape development models along with other relevant data, notably crustal uplift rates, which are relevant to both shoreline regression and river incision.

Again, landscape development is strongly dependent upon local geography, so the MODARIA I project was limited to discussions of the relevant issues and presented as an approach that, in turn, is supported with illustrative examples for warm, arid and temperate conditions, periglacial conditions and glacial conditions, and transitions between them. These illustrate the approach adopted to construction of narratives for environmental change, based on the assumptions used in climate modelling and downscaling. Different aspects of the narratives can be developed and used differently within a single assessment, but it is important that they draw on the same foundation.

The next part of the MODARIA I methodology concerned the use of these narratives in safety assessments. It is highlighted that the narratives can be used in various ways to support the assumptions for assessment models. The choice of simplifying assumptions can be an important consideration, along with how to address uncertainties in the context of present day conditions and the treatment of future climatic and landscape conditions.

The case studies that were reviewed and evaluated demonstrate the value of a step by step approach in building confidence in modelling results, show the potential value of the use of analogues, and illustrate the role of stakeholder engagement in building trust. A description of the landscape at the present and how it could evolve in the future is an aspect of the post-closure assessment process that is particularly accessible to, and understandable by, various stakeholder groups. Descriptions of future biosphere characteristics and human behaviour can appear speculative and subject to challenge. There is, therefore, a need to carefully distinguish those aspects of the assessment that are based on quantitative analyses (e.g. derived from climate and landscape models) from those that are based on regulatory requirements or other judgements and decisions. It is recognized that, in practice, the distinction is not clear cut and that some aspects will be determined to a greater extent by judgmental interpretations of quantitative modelling results. In this context, uncertainty analyses play an important role in investigating the alternative scenarios that arise from different points of view on assessment issues and determining the robustness of safety arguments across these alternative points of view. Engagement with stakeholders is essential both to explain the basis of quantitative aspects of the assessment and to support development of consensus, particularly on those aspects where judgement has the predominant role. An important example is the selection of assumption(s) for anthropogenic CO_2 releases that depend upon a combination of technical, economic and political factors.

Noting these issues, it is highlighted that the results produced through the application of the methodology are only intended as projections of possible futures based on a set of assumptions (i.e. reference futures). Therefore, it is emphasized that assessments are not predictions of the future but are illustrative projections that encompass plausible future situations to an extent that is sufficient to provide confidence in the safety of a disposal system. Notwithstanding the uncertainties that exist, it has been shown through research reviewed and undertaken in the MODARIA I project that quantitative long term climate modelling is sufficiently developed and robust to define an envelope of reference futures for use in safety assessments for radioactive waste repositories, as supported by understanding of paleoclimatic conditions. The climate models that can be used for this purpose have limited spatial resolution and in some cases downscaling is necessary. Physical–statistical methods exist to do this, as described in the MODARIA I report [III–1], but local statistical data are needed to apply them. Qualitative downscaling can also be used.

Quantitative modelling of landscape evolution and the linkage with climate modelling has been significantly developed in recent years, but not for all potentially relevant climates and landscapes. Further work in this area could be undertaken and special attention may have to be given to more detailed understanding of the first few thousand years after disposal. This goes beyond the typical focus of the Intergovernmental Panel on Climate Change (IPCC) (see Ref. [III–4]) but is especially relevant to near surface disposal and the long term management of radioactively contaminated sites and other areas.

Although the focus of the studies undertaken in MODARIA I was radiation dose assessment following releases to the biosphere, the methodology and results obtained are valuable in a wider safety assessment context, for example addressing the effects of climate change and landscape development upon releases to the biosphere (see Ref. [III–5] for a discussion of approaches to representing the interface between the geosphere and the biosphere). They could also be of interest to those with an interest in assessment of the impact of chemically hazardous materials in radioactive waste repositories, and in the general issue of the disposal of hazardous wastes.

III–2. CONSIDERATION OF LONG TERM SEA LEVEL CHANGES

For coastal and near coastal disposal facilities, changes in sea level can have a major impact on post-closure safety, for example through erosion, inundation and the effects the groundwater hydrogeology and hydrogeochemistry. These changes arise through an interplay of local isostatic factors (e.g. due crustal movements arising from glacial loading and unloading) and eustatic factors arising from changes in the volume of the oceans (e.g. from changes in the size of the Greenland and Antarctic ice sheets and of continental glaciers and ice caps, and from the thermal expansion and contraction of seawater). In view of the importance of this topic, a brief account of potential future eustatic changes in sea level is provided here.

As discussed in section 13.2 of Ref. [III–4], the present day global sea level was reached ~6000 years ago. Subsequently, the variation in global sea level has only been ~1 m, with a trend towards increasing sea levels that began worldwide early in the 20th century [III–6]. In terms of forward projections, the IPCC (see section 13.5 of Ref. [III–4]) considers that sea level rise during the 21st century is likely to be between 0.28 to 0.61 m for the representative concentration pathway 2.6 (RCP2.6) scenario and between 0.52 and 0.98 m for the RCP8.5 scenario. However, projections of rises of up to 2.4 m have been reported in the literature (specifically pg. 1186 of Ref. [III–4]).

More recently, a probability density function of the global sea level at 2100 CE has been constructed [III–7] and the probability of sea level rises of more than 1.8 m is less than 5%. An upper estimate of 1.9 m was obtained by summing the highest estimates of individual sea level rise components simulated by process based models with the RCP8.5 scenario [III–7]. The upper part of the probability distribution of sea level rise projections was hard to quantify because of the uncertainties that currently exist in projections from ice sheet dynamical models [III–7].

Relative sea level rise projections for Europe at 2100 CE are given in Ref. [III–8]. These take account of global sea level rise, alterations in the geoid, expressed through the dynamic ocean response, and isostatic adjustments.

Table 13.8 in Ref. [III–4] also provides longer term projections of the global mean sea level (GMSL) rise for low, medium and high emission scenarios (spanning the range from RCP2.6 to RCP8.5). These projections are summarized in Table III–1.

On multimillennial timescales, Ref. [III–4] provides estimates of changes in GMSL as a function of the overall global increase in temperature. Contributions arise from thermal expansion of the oceans, melting of mountain glaciers, loss of part of the Greenland ice sheet and changes in the Antarctic ice sheets. The major uncertainty relates to the Greenland ice sheet, which will be almost completely lost following a global temperature increase of 2°C or more and will contribute ~6 m of sea level rise. Overall, the global sea level rise ranges up to ~10 m in the first 2000 years and up to ~15 m at equilibrium. The GMSLs during previous interglacials (MIS 5e and MIS 11) have been up to ~10 m above that at the present day [III–4].

TABLE III–1. PROJECTIONS OF GLOBAL MEAN SEA LEVEL (GMSL) RISE FOR LOW, MEDIUM AND HIGH EMISSION SCENARIOS [III–4]

Emission scenario	Year of projection (CE)				
	2100	2200	2300	2400	2500
Low	0.26–0.53 m	0.35–0.72 m	0.41–0.85 m	0.46–0.94 m	0.50–1.02 m
Medium	0.19–0.66 m	0.26–1.09 m	0.27–1.51 m	0.21–1.90 m	0.18–2.32 m
High	0.21–0.83 m	0.58–2.03 m	0.92–3.59 m	1.20–5.17 m	1.51–6.63 m

Since Ref. [III–4] was published, further work has been undertaken in relation to the potential sea level contribution from the Antarctic ice sheets. For example, a model was used [III–9] that includes hydrofracturing of buttressing ice shelves and structural collapse of marine terminating ice cliffs to estimate that at 2500 CE the global mean sea level rise for the RCP8.5 scenario would be 12.3 m, to which a further 1.3 m due to warming feedbacks arising from the retreating ice sheet needs to be added. In this model, the projected sea level rise at 2100 CE is more than 1.0 m.

If consumption of unconventional fossil fuels such as clathrates is considered, substantially greater sea level increases could occur. For example, it has been shown through model simulations that with cumulative fossil fuel emissions of 10 000 PgC, Antarctica is projected to become almost ice free with an average contribution to sea level rise of more than 3 m per century over the next millennium [III–10]. In this model, an increased sub-ice sheet melt rate forces grounding line retreat into an area where the ice is grounded below sea level on inward sloping bedrock resulting in progressive instability. The authors propose that the West Antarctic ice sheet will become unstable through this mechanism after cumulative carbon emissions reach 600–800 PgC. A similar instability of the larger East Antarctic ice sheet arises for a release of more than ~2500 PgC.

In the longer term, when the effects of anthropogenic greenhouse gas emissions have declined substantially, glacial–interglacial cycling is expected to resume. During glacial episodes, continental ice sheets similar in size to those of the last glacial maximum (MIS 2) would probably develop, leading to a fall in GMSL of ~120 m relative to the present day. However, because of the long term persistence of a fraction of carbon dioxide releases in the atmosphere, it is likely that glacial–interglacial cycling will not resume for a minimum of some tens of thousands of years and possibly not for 100 000 years or more [III–11]. If glacial–interglacial cycling does resume and exhibits a pattern like that observed over the last 400 000 years, then cycles of ~ 100 000 years are projected to occur, with an oscillatory cooling trend and a corresponding trend in sea level (i.e. generally falling), but with shorter term increases during periods of global warming.

III–3. CONSIDERATIONS RELATING TO RADIOACTIVE WASTE DISPOSAL FACILITIES LOCATED IN WARM, ARID ENVIRONMENTS

Although many existing and proposed geological repositories for radioactive wastes are in temperate environments, this is not universally the case. For example, the existing near surface repository at El Cabril, Spain, the proposed deep repository for high level waste and spent nuclear fuel at Yucca Mountain, Nevada, United States of America (USA), and many others are in warm, arid regions. Special considerations apply to defining and characterizing the biosphere in such warm, arid environments, as discussed below. Cold, arid environments are not addressed here, as these are generally considered in respect of repositories located in zones where temperate conditions apply, but where future, colder conditions that could be appropriately characterized as arid, periglacial environments are also addressed.

A comprehensive account of warm, arid landscapes and the geomorphological processes determining the forms of those landscapes is provided in Ref. [III–12].

In broad terms, three, interlinked factors distinguish repositories located in arid regions from repositories located in temperate environments:

(a) Location relative to the water table. In temperate environments, the regional water table is typically located close to the surface. This means that geological repositories are generally located in the saturated zone beneath the regional water table. Near surface facilities could be located either above or below the water table, but even where they lie above it, the distance from the saturated zone is generally small. In contrast, in warm, arid regions, the regional water table often lies tens to hundreds of metres below the ground surface, so near surface (and possibly some geological) repositories are likely to be in the unsaturated zone well above the water table, making pathways and transport times for radionuclides in the unsaturated zone an important consideration in radiological impact assessment.
(b) Event driven hydrology and hydrogeology. In temperate environments, surface water and groundwater flows typically persist throughout the year (though some aspects of flow could be transient, e.g. discharges of

ephemeral streams). Although both surface water and groundwater flows can show substantial responses to individual precipitation events, these are typically perturbations imposed on a flow regime dominated by longer term changes (i.e. seasonal, interannual and longer). In contrast, in warm, arid environments, individual storm events could be the dominant consideration in defining the hydrological and hydrogeological regime. Thus, a focus on mean seasonal and annual temperature and precipitation is replaced by an emphasis on quantifying the return periods of storms of differing intensity and duration, with the most extreme events seen as the primary determinants of hydrology and hydrogeology, as well as of sediment mobilization, transport and deposition.

(c) Styles of landform development. In temperate environments, the landscape has typically been shaped by ice and or water. Where continental ice sheets have been present in the past, the landscape can be dominated by glacial landforms, though these can have subsequently been substantially modified by fluvial processes, such as stream incision. Even beyond the most extensive margins of former ice sheets, glacial outwash processes and the active hydrological regimes that are associated with the retreat of ice sheets could have profoundly affected the landscape. In contrast, warm, arid environments were typically well beyond the margins of former ice sheets, so their landscapes have largely been formed, sometimes over much longer timespans, by an interplay of aeolian and fluvial processes, with event driven aspects (e.g. intense storms with long return periods) potentially being the determining factor in landscape development. Additionally, whereas landform development in temperate environments can be strongly moderated by dense vegetation cover, in warm, arid environments sparse vegetation cover may enhance the efficiency by which wind and water modify the landscape.

The above considerations are illustrated below, by reference to El Cabril and Yucca Mountain as specific examples.

III–3.1. Characteristics of El Cabril and Yucca Mountain disposal facilities

III–3.1.1. El Cabril

El Cabril is the facility used for the disposal of all low and intermediate level radioactive wastes in Spain (in accordance with the Spanish waste classification system). It is designed to cover all the current disposal needs for these types of waste, including those arising from the dismantling of nuclear power plants.

The facility is in the hills of the Sierra Albarrana, in the province of Córdoba, and its history as a waste disposal facility dates to 1961, when the Nuclear Energy Board transferred the first drums of radioactive wastes to the site, disposing of them in a nearby disused uranium mine.

The facility has two platforms for the disposal of low and intermediate level radioactive wastes and another with specific structures for very low level wastes. In addition, the facility has the resources necessary for the treatment and conditioning of wastes.

The low and intermediate level wastes generated at any location in Spain arrive at El Cabril and are unloaded at a conditioning building or one of the temporary storage facilities. Most of these wastes, generated at the nuclear power plants, are already conditioned on arrival, whereas those coming from hospitals, research centres or industry are treated and conditioned at El Cabril.

The waste drums that are received are placed in concrete containers with a capacity of 18 × 220 L drums. When a container is full, its drums are immobilized by means of injected mortar. The compact block is placed in the disposal cell, which is a structure of reinforced concrete. Once the disposal cell is filled with 320 containers, the upper reinforced concrete closure slab is constructed and weatherproofed. Each of the 28 disposal cells has a sump connected to the seepage control network located beneath the platforms. This allows possible inflows of water to be detected and, if they occur, for the system to be repaired. Once the capacity of the platforms has been exhausted, they will be covered by a sequence of layers of different materials, the last of which will be of topsoil, allowing integration into the environment.

From the time at which the cover is installed, a 300 year site surveillance and control phase will begin.

Very low level wastes are solid materials, generally scrap and rubble, that are minimally contaminated with radionuclides. They can arrive at the facility in sacks, drums or containers and can be disposed of directly in the specific disposal structure or first be taken to the area set aside for their treatment, if necessary. As each structure is completed, it will be covered with various layers, the last of which will be of topsoil, allowing for integration into the environment. From the time at which the cover is installed, a 60 year site surveillance and control phase will begin.

The facility is located on a hillside at an altitude of ~330 m, but the land falls away rapidly to the east, decreasing to an altitude of ~250 m within 1 km of the facility.

III–3.1.2. Characteristics of Yucca Mountain potential disposal facility

The Yucca Mountain site is in Nye County in southern Nevada, USA, ~160 km northwest of Las Vegas. It has been investigated since the 1970s as a potential location for the disposal of vitrified high level waste and spent nuclear fuel. Yucca Mountain comprises a tilted block of mid-tertiary volcanic rocks erupted from a series of volcanic centres that form the southwest Nevada volcanic field. It lies within the north-central part of the Basin and Range Province. Landforms in the region are characterized by regularly spaced, generally north–south trending, mountain ranges and intervening alluvial basins. Yucca Mountain lies near the centre of the upper Amargosa drainage basin, which originates in the Pahute Mesa–Timber Mountain area to the north and includes the main tributary systems of Beatty Wash and Forty Mile Wash. Details of the site are given in two US Geological Survey Memoirs [III–13, III–14]. Details of the post-closure safety assessment conducted in support of the licence application for construction of the proposed repository are given in Ref. [III–15]. Note that the commentary provided here relates throughout to Update No. 1 of the licence application, rather than the original licence application.

The proposed repository at Yucca Mountain would be in the unsaturated regime at a depth of ~300 m and ~300 m above the regional water table.

The rocks of the unsaturated zone comprise a sequence of welded and nonwelded tuffs. Welded units are characterized by high fracture densities, with water flow being predominantly through the fractures rather than through the rock matrix. In contrast, non-welded units are characterized by much less transmissive fracture systems, such that water flow is conceived as being predominantly through the rock matrix. A key unit in this respect is the paintbrush non-welded unit, as this is envisaged as damping the periodicity of percolation arising from event-driven infiltration. However, studies of ^{3}H and ^{36}Cl have identified that fast flow pathways exist through the non-welded units [III–16], while subvertical faults, mainly outside the repository footprint, can also act as water transmissive features.

At the surface, the environment is arid, with only sparse vegetation.

III–3.2. Potential radionuclide transport and exposure pathways

III–3.2.1. El Cabril

Because only very small amounts of long lived radionuclides will be disposed of at El Cabril, it is appropriate to give principal consideration to the first few millennia after site closure. In broad terms, the following pathways of radionuclide transport are identified as being of potential significance. Each of these is then discussed in more detail below:

- — Erosion of cover materials leading to exposure of the wastes;
- — 'Bath tubbing' in which meteoric water fills up the disposal vaults and seeps through the cover materials either above or at the edges of the disposal area;
- — Evolution of radioactive gases that move upward through the overlying capping layers;
- — Downward drainage of meteoric water through the disposal vaults into the underlying strata and subsequent downslope migration of the contaminated groundwater.

(a) Erosion of cover materials

Although the general level of the El Cabril facility lies close to the pre-existing topography, it appears that the cover over the vaults will be somewhat above the existing ground surface. In these circumstances, individual storm events can potentially result in both general erosion of the cover materials and localized erosion, leading to the formation of gullies. The risks of these effects can by minimized by adopting low side slopes (typically $<15^{\circ}$ slope angle) and by using a superficial armouring layer. Incision through the cover would expose the wastes in situ and subsequent erosion could transport the exposed wastes downslope, creating a debris fan. It is expected that appropriate design will be able to prevent these effects occurring on the key principal timescale of a few millennia. However, in the longer term, overall development of the landform with lowering of the ridge on which El Cabril is located could result in exposure of the wastes and their downslope migration.

(b) Bath tubbing

The base of the vaults incorporates a waterproof layer of polyurethane, but the effective lifetime of such a layer is difficult to determine. Below the base of the vaults there is an inspection and drainage system, but it is not clear whether this would remain effective or whether the drainage would become clogged in the post-closure period. If, for whatever reason, the hydraulic conductivity of the base of the vaults, underlying engineered structures and host geology was low, then perched water could develop in the vaults. As this water would be present at some depth below the cover layers, it would not be subject to strong evapotranspiration. Individual intense storm events could lead to a rapid increase in the depth of perched water present and result in contaminated water reaching the overlying soil layer as the storm water mixed with the pre-existing perched groundwater. This potential pathway requires specific consideration over the first few millennia when the engineered structures could be largely intact and form a suitable receptacle for infiltrating water. However, it may be strongly inhibited by the design of the cover, which includes a suitable combination of drainage and impermeable layers to ensure that (if these remain intact) most infiltrating water will be deflected laterally and discharged to the surface at the boundaries of the disposal area. Nevertheless, consideration needs to be given to the possibility that a combination of subsidence and erosion could cause meteoric water to 'pond' in specific locations on the cover and drain through damaged areas of the cover at those locations into the underlying wastes. This is also relevant to the groundwater transport pathway discussed below.

(c) Radioactive gases

The waste accepted at El Cabril includes both metals and organic materials. Therefore, various gases could be generated. In the overlying soil, any methane is likely to be substantially converted to carbon dioxide and ^{14}C present in labelled carbon dioxide, which would then be available for plant uptake by photosynthesis. This potential pathway is of most relevance in the first few millennia after repository closure, considering both the radioactive half-life of ^{14}C and the timescale over which bulk gases may be generated.

(d) Groundwater transport

As discussed above, in the longer term subsidence, erosion and materials degradation can reduce the capacity of the cover layers to laterally redirect infiltration to the boundaries of the disposal area. Thus, infiltration can percolate downward through the wastes, become contaminated and then drain through the base of the vaults to the local water table, where the contaminant plume will move downslope towards a discharge area, likely towards the base of the slope to the east of the repository. In this case, radiation exposures can arise either from abstraction of contaminated water from a well, from groundwater discharge at the ground surface or from groundwater discharge into a local stream.

III–3.2.2. Yucca Mountain

At Yucca Mountain, the nominal (or reference) scenario focuses on groundwater transport of radionuclides. Meteoric water infiltrates at the ground surface above the repository and is transported downward to the level of the repository by a combination of fracture and matrix flow, with the relative importance of these two transport routes being dependent on the rock unit involved and the main distinction being between welded and nonwelded tuff.

At the repository horizon, the pattern of water flow is influenced by the presence of the repository drifts and by the heat output of the wastes. In the first few thousand years after repository closure, the temperature at the walls of the emplacement drifts is calculated to be above the boiling point of water (locally 96°C, because of elevation). Thus, liquid water is initially prevented from entering the drifts but is envisaged as being transported upward in vapour form and condensing above the repository horizon. During this period, water can flow through the intact rock between the drifts and penetrate to greater depths.

As the wastes cool, the walls of the emplacement drifts also cool. Eventually, the wall temperature will drop below the boiling point of water. After this time, water can drip onto the protective titanium drip shields overlying the waste containers. During this period, some infiltrating water can be prevented from entering the drifts by a capillary barrier effect at the wall–drift interface, but the effectiveness of this capillary barrier is disputed, with it being affected by the relative importance of fracture and matrix flows, and by the type of model used to represent those flows. Additionally, as in the earlier high temperature period, some infiltrating water may bypass the drifts and flow through the intact rock between them.

Water entering the drifts can act to corrode the drip shields and waste packages, particularly as water dripping onto these engineered components, or condensing onto them, will evaporate, leaving salts containing highly concentrated liquid inclusions. Eventually, waste packages will fail, and water will enter them, leaching radionuclides and transporting them through the inverts that support the waste packages and into the unsaturated rock beneath the repository. The radionuclides are envisaged as being advectively transported downward to the regional water table and then transported sub-horizontally in the saturated zone, with the first part of the path being in fractured volcanic rock and the second part in alluvium, which is treated as a porous continuum for both flow and transport calculations. Finally, the contaminated water is envisaged as being extracted from relatively deep groundwater wells in Amargosa Valley, where it is used for a wide variety of domestic and agricultural purposes. The warm, arid nature of the climate in Amargosa Valley means that extraction from the alluvial aquifer is close to the limit of sustainability to meet the high local water demand. Also, water use is high in crop irrigation, for example, where the hot climate means that multiple harvests per year are possible, provided that the water requirements of those crops can be met.

Other scenarios are also addressed, including situations in which seismic events damage drip shields and waste packages, and both igneous intrusions into the repository and igneous eruptions in which radioactive wastes are dispersed in

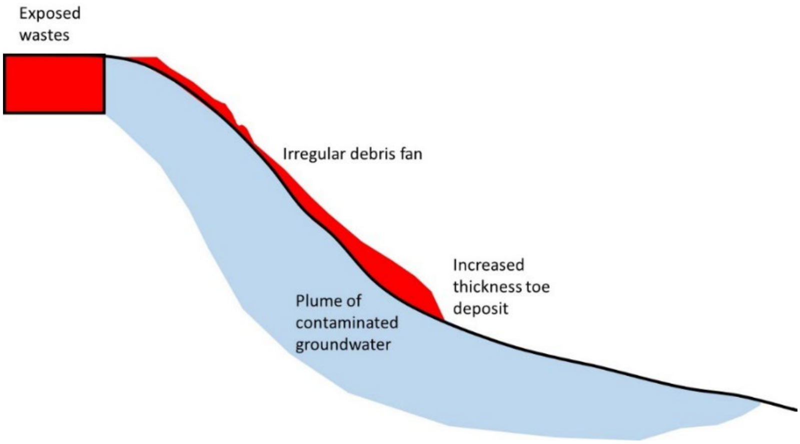

FIG. III–1. Schematic (not to scale) of the development of a debris fan.

the associated cloud of ash and dust released to the atmosphere. These scenarios are of limited interest in the current context and are not addressed further herein.

III–3.3. Considerations in modelling radionuclide transport and exposure pathways

III–3.3.1. El Cabril

(a) Erosion of cover materials

Both general erosion and gullying could expose the wastes and transport them downslope, resulting in a debris fan. Initially, this debris would comprise a mix of waste items and degraded repository engineering materials, but over time weathering processes would tend to degrade this material to smaller particle sizes. In addition, radionuclides could be leached from the wastes due to precipitation events. A schematic of how the debris fan might develop is shown in Fig. III–1. Note that this figure assumes that the hillslope has retreated towards the engineered facility, as well as there being general erosion of the hilltop in which the facility is embedded.

Radionuclide concentrations in the exposed wastes are likely to be similar to, or rather higher than, those in the debris fan. However, both components are of a similar general nature, so the safety assessment calculations can be based on similar principles.

The groundwater plume will develop due to leaching of the exposed wastes and of the debris fan. However, it will be similar to the type of radionuclide plume that could develop from an uneroded repository. Thus, the approach to modelling can be assumed to be the same as that discussed below.

For the exposed wastes and debris fan, a simple transport model can be used. Radionuclide inventories in the disposed wastes can be estimated by decay and ingrowth calculations, neglecting any losses by leaching due to infiltration. Radionuclide concentrations in the exposed wastes can then be calculated by dividing those inventories by the mass of wastes plus the mass of associated engineered components of the repository. For the debris fan, the concentrations in the exposed wastes can be used, multiplied by a dilution factor to allow for the degree to which material eroded from the repository is diluted by material from other sources, for example from the slope itself, as it moves downslope. This could be particularly relevant to the toe of the debris fan. In practice, such dilution is likely to be limited and it may be cautious but reasonable to assume that radionuclide concentrations in the debris flow are the same as those in the exposed wastes. Note that this approach neglects leaching of radionuclides from the debris fan. This is a cautious but reasonable assumption, as much of the radionuclide inventory can be initially incorporated within the solid matrix of material in the debris fan.

The main pathways of exposure are likely to be external exposure, inhalation of suspended particles and adventitious ingestion of contaminated dust. The average thickness of the debris fan is likely to be several centimetres or more (though it might not be spatially continuous) and it is likely to have a lateral extent of tens of metres or more. Therefore, it is considered to be cautious but realistic to assume that the fan is effectively infinitely thick with respect to gamma emissions and to use the gamma dose rate above a thick slab source. Thus, the effective dose rate, H_{ext} (Sv/a), can be given by:

$$H_{\mathrm{ext}} = OF\Sigma_i \, C_i H_{i,\mathrm{ext}} \tag{III–1}$$

where

- O (−) is the fractional annual occupancy (which is likely to be <<1 downslope of El Cabril and might be <0.01);
- F (−) is the fraction of the area of the debris fan over which radioactive waste is present;
- C_i (Bq/kg) is the average concentration of radionuclide i in the debris in the fraction of the area of the fan where radioactive waste is present;

and $H_{i,\text{ext}}$ (Sv/a per Bq/kg) is the effective dose rate from radionuclide i at 1 m above a thick slab source of infinite lateral extent [III–17].

Note that $H_{i,\text{ext}}$ needs to take account of short lived progeny for which concentrations are not explicitly calculated, but that are assumed to be present in secular equilibrium.

In the case of inhalation, it can be cautiously assumed that some of the material is eroded down to a size range of < ~10 μm. Such material is susceptible to resuspension. In this case, the effective dose rate, H_{inh} (Sv/a), can be given by:

$$H_{\text{inh}} = OMB\ \Sigma_i E_i C_i H_{i,\text{inh}} \tag{III–2}$$

where

M (kg/m^3) is the mass loading of locally derived dust in air;
B (m^3/a) is the inhalation rate (taking that appropriate to moderate exercise in outdoor conditions);
E_i (−) is the degree to which radionuclide i is enriched or depleted in the resuspended dust compared with the ground deposit;
$H_{i,\text{inh}}$ (Sv/Bq) is the committed effective dose per unit intake by inhalation;

and other quantities are as defined previously.

Note that O, B and $H_{i,\text{inh}}$ are all age dependent.

For adventitious ingestion, the effective dose rate, H_{ing} (Sv/a) can be given by:

$$H_{\text{ing}} = M_{\text{ing}} F_{\text{ing}} \Sigma_i\ R_i C_i H_{i,\text{ing}} \tag{III–3}$$

where

M_{ing} (kg/a) is the rate of inadvertent ingestion of dust;
F_{ing} (−) is the fraction of that dust that is contaminated;
R_i (−) is the degree to which radionuclide i is enriched or depleted in the ingested contaminated material compared with the ground deposit;
$H_{i,\text{ing}}$ (Sv/Bq) is the committed effective dose per unit intake by ingestion;

and other quantities are as defined previously.

Note that M_{ing}, F_{ing} and $H_{i,\mathrm{ing}}$ are age dependent and that the fractional gastrointestinal absorption of a radionuclide from ingested dust could differ from that of the same radionuclide incorporated in food, resulting in the need to use a different value of committed effective dose per unit intake.

(b) Bath tubbing

In bath tubbing, it is necessary to consider that water present in the wastes transfers radionuclides to water present in the overlying soil (see Fig. III–2). The concentrations in solids can differ, since the distribution coefficients (K_{d} values) that are applicable can differ between the wastes and other near field materials and the overlying soil.

Defining $C_{i,\mathrm{sol}}$ (Bq/m^3) as the concentration of radionuclide i in solution, the concentrations on near field materials and overlying soil (Bq/kg) can be defined as $K_{\mathrm{d},nf,i} \times C_{i,\mathrm{sol}}$ and $K_{\mathrm{d,soil},i} \times C_{i,\mathrm{sol}}$, respectively. Here, $K_{\mathrm{d,nf},i}$ and $K_{\mathrm{d,soil},i}$ are the distribution coefficients (m^3/kg) for the near field and soil components of the system, respectively. Note that the argument below is given for these two components, but it can be readily extended to include a larger number of components.

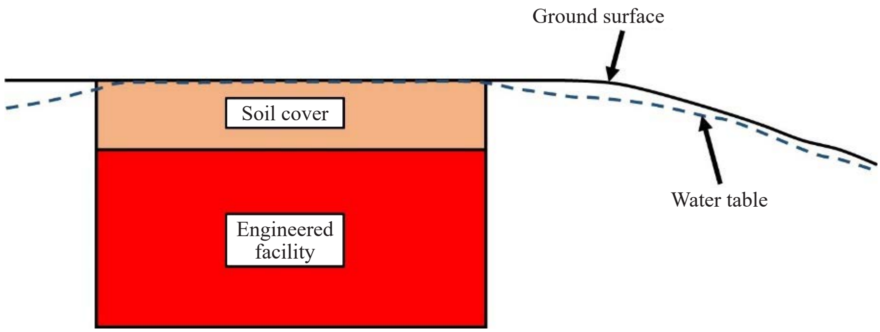

FIG. III–2. Bath tubbing conditions.

From the above, at equilibrium the total inventory of radionuclide i, A_i, is given by:

$$A_i = C_{i,\text{sol}} \begin{pmatrix} V_{\text{nf}} \rho_{\text{nf}} K_{\text{d,mf},\,i} + V_{\text{nf}} \varphi_{\text{nf}} + \\ V_{\text{soil}} \rho_{\text{soil}} K_{\text{d,soil},i} + V_{\text{soil}} \varphi_{\text{soil}} \end{pmatrix} \qquad \text{(III–4)}$$

where

V_x (m^3) is the volume of a component x;
ρ_x (kg/m^3) is the dry bulk density of that component;

φ_x (−) is the water filled fractional porosity of that component.

From this expression, $C_{i,\text{sol}}$ is readily obtained, as is the concentration on soil solids, $K_{\text{d,soil},i} \times C_{i,\text{sol}}$.

As with the debris plume, the main exposure pathways are likely to be external exposure, inhalation of resuspended material and inadvertent ingestion. For these pathways, the same equations can be used as set out above. The equation for inhalation applies to dry soil and a cautious approach is to assume that when the soil dries out after a bath tubbing event, the radionuclide content present in the pore water deposits on the solids. Therefore, the appropriate concentration, Ci (Bq/kg) to use is given by:

$$C_i = \left(\rho_{\text{soil}} K_{\text{d,soil},i} + \varphi_{\text{soil}}\right) C_{i,\text{soil}} \,/\, \rho_{\text{soil}} \qquad \text{(III–5)}$$

This approach is also cautious for the external exposure and inadvertent ingestion pathways, since the gamma attenuation of dry soil is less than that of wet soil and inadvertent soil ingestion is generally expressed on a dry mass basis [III–18].

(c) Radioactive gas release

The two main gases that could potentially be released are ^{222}Rn and ^{14}C. For the latter, $^{14}CH_4$ is likely to be the main gas of relevance, though it might migrate to the surface as a trace component in bulk hydrogen. In the case of ^{222}Rn, the rate production, $P_{\text{Rn-222}}$ (Bq/a), would be $\lambda_{\text{Rn-222}} \times A_{\text{Ra-226}}$, where $\lambda_{\text{Rn-222}}$ (a^{-1}) is the radioactive decay constant of ^{222}Rn and $A_{\text{Ra-226}}$ (Bq) is the time dependent inventory of ^{226}Ra in the waste. In the case of $^{14}CH_4$, the rate of production, $P_{\text{C-14}}$ (Bq/a) needs to be derived from a specific waste degradation model.

In the case of ^{222}Rn, not all the activity produced would escape at the surface. Allowance needs to be made for the fraction trapped in the waste form until it decays and the fraction that decays in the period after it is released from the waste form, but before it is released at the surface. Thus, the release rate, $Q_{\text{Rn-222}}$ (Bq/a) = $F_{\text{waste}} \times F_{\text{trans}} \times P_{\text{Rn-222}}$, where F_{waste} is the fraction released from the waste and F_{trans} is the fraction that does not decay in transit to the surface.

In the case of ^{14}C, its long half-life (5730 years) suggests that losses between production and release might be substantially less for ^{14}C than for ^{222}Rn. However, this is not necessarily the case. There are some waste forms, such as reactor graphite, that are extremely effective at retaining ^{14}C, and losses during transport might not be by decay, but either by dissolution in solution or by chemical reactions, notably with cementitious materials. Thus, again it is appropriate to write $Q_{\text{C-14}}$ (Bq/a) = $F'_{\text{waste}} \times F'_{\text{trans}} \times P_{\text{C-14}}$, where the primes are used as a reminder that the F values will differ between ^{222}Rn and ^{14}C.

In general, movement of gas from a facility will tend to be vertical, unless a high integrity cap of low gas permeability is in place. If this is the case, the gas will tend to be deflected laterally and emerge around the edges of the cap. For initial assessment studies, it is appropriate to assume that either the cap is of higher gas permeability or that it has failed at various places. For these initial assessment studies, it would then be reasonable to assume that radionuclides in the gas phase are uniformly released over the area, S (m^2), of the facility. Thus, the fluxes, $\Psi_{\text{Rn-222}}$ and $\Psi_{\text{C-14}}$ (Bq·m^2 per a), are given by $Q_{\text{Rn-222}}/S$ and $Q_{\text{C-14}}/S$, respectively.

In the case of ^{222}Rn, the relevant pathway would be inhalation of ^{222}Rn and its progeny in indoor air, assuming that a building was constructed over the engineered facility. Taking the plan area of the building to be A_B (m^2), its volume to be V_B (m^3) and the air exchange rate in the building to be k (a^{-1}), the average air concentration of ^{222}Rn, $C_{\text{Rn-222}}$ (Bq/m^3) can be calculated using:

$$C_{\text{Rn}-222} = \Psi_{\text{Rn}-222}\, A_B / \left\{V_B\left(k+\lambda^{\text{Rn}-222}\right)\right\} =$$

$$\Psi_{\text{Rn}-222} / \left\{h_B\left(k+\lambda^{\text{Rn}-222}\right)\right\} \qquad \text{(III–6)}$$

where h_B (m) is the height of the building. Cautiously, a single storey building of height 2.5–3.0 m is typically assumed. Note that this formulation takes account of losses by ventilation and radioactive decay. Plate-out of the progeny of ^{222}Rn is taken into account by computing an equivalent concentration of ^{222}Rn, $C_{\text{Rn-222,eq}}$, that is somewhat less than the ^{222}Rn concentration, and using this concentration to compute the annual effective dose, $H_{\text{Rn-222}}$ (Sv/a), from the expression:

$$H_{\text{Rn}-222} = O_{\text{B}} C_{\text{Rn}-222,\text{eq}} H_{\text{Rn}-222,\text{inh}} \quad \text{(III–7)}$$

where O_B (−) is the fractional occupancy of the building and $H_{\text{Rn-222,inh}}$ ((Sv/a)/(Bq/m^3)) is the effective dose rate per unit concentration of ^{222}Rn in air in equilibrium with its short lived progeny as given by UNSCEAR [III–19 to III–21] and the ICRP [III–22].

In the case of ^{14}C, the main pathway of relevance is incorporation in plants through a combination of foliar and root uptake followed by photosynthesis, with losses by plant respiration also being taken into account. Detailed resistance analogue models have been developed for this purpose [III–23] and these can be used to estimate the specific activity of ^{14}C in plants, C_{plant} (Bq/kg C, from the flux of ^{14}C (assumed to have been converted to $^{14}CO_2$ in the soil zone), using:

$$C_{\text{plant}} = \Psi_{\text{C}-14} F_{\text{RA}} \quad \text{(III–8)}$$

where F_{RA} (Bq·kg [C]/ per Bq/m^2 per a or equivalently m^2/a per kg [C]) is a flux to concentration ratio derived from a resistance analogue model. This typically has a value of ~0.04.

Once the specific activity in plants has been determined, the specific activity in animals, C_{animal} (Bq/kg C) can also be calculated using:

$$C_{\text{animal}} = C_{\text{plant}} \eta \quad \text{(III–9)}$$

where η (−) is the fraction of the carbon intake by the animal that derives from the contaminated plants. Similarly, the specific activity in humans, C_{human} (Bq/kg C), can be calculated using:

$$C_{\text{human}} = \eta_{\text{plant}} C_{\text{plant}} + \eta_{\text{animal}} C_{\text{animal}} \quad \text{(III–10)}$$

where η_{plant} and η_{animal} (−) are the fractions of dietary carbon obtained from contaminated plant and animal products, respectively. The concentration of ^{14}C in human tissues expressed on a fresh weight basis, $C_{\text{human},f}$ (Bq/kg), is then given by $f_C C_{\text{human}}$, where f_C (−) is the fraction of carbon in human tissues and organs (0.22).

Given the concentration of ^{14}C in human tissues, the effective dose rate, $H_{C\text{-}14}$ (Sv/a), can be computed using:

$$H_{C-14} = \left(3.156\times10^{7}\right)\times\left(1.602\times10^{-13}\right)\times C_{\text{human}}\times E_{C-14} \quad \text{(III–11)}$$

where 3.156×10^{7} is the number of seconds in a year, 1.602×10^{-13} is the conversion factor from MeV to J and $E_{C\text{-}14}$ (MeV) is the mean beta energy emitted per transformation of ^{14}C.

(d) Groundwater transport

For groundwater transport, it would first be appropriate to calculate the time dependent flow field in the unsaturated and saturated zones using a physically based, surface water catchment hydrogeological model such as MIKE-SHE [III–24]. The resulting flow field could then be used to calculate radionuclide transport downslope using the standard advection–dispersion relationship. A possible geometry of the flow system is illustrated in Fig. III–3, where the plume is seen to first migrate downward through the unsaturated zone and then sub-horizontally to discharge towards the bottom of the slope. An alternative geometry, where the plume moves sub-horizontally in the near surface layers, is illustrated in Fig. III–1. Erosion is taken to be of less significance in Fig. III–3 than in Fig. III–1.

Application of such a model for groundwater transport would give radionuclide concentrations in soils and surface water bodies. Thereafter, conventional radiological impact calculations, such as that undertaken in the terrestrial biosphere model used by RWM [III–25], could be made.

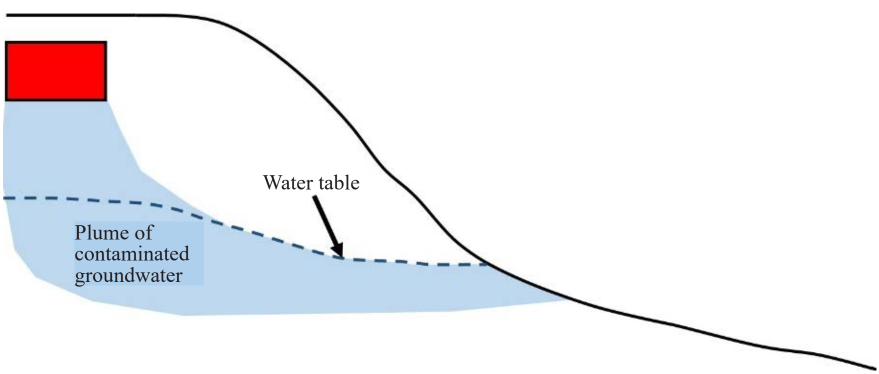

FIG. III–3. Illustrative groundwater plume.

III–3.3.2. Yucca Mountain

For the nominal scenario adopted in the safety assessment report (SAR) [III–15], two main considerations are how water flow and radionuclide transport could be modelled in rocks with different degrees of fracturing, alternative fracture fill and fracture wall characteristics, and differing matrix properties (including composition, pore shapes and sizes, and pore connectivity). In unsaturated media, computing flow patterns and flow rates is complicated by the close connection that exists between degree of saturation, hydraulic conductivity and matrix potential. Highly non-linear relationships exist between these three quantities, which can be represented using the Richards equation, which is the extension of Darcy's law that is applicable to unsaturated conditions.

In developing a flow model for Yucca Mountain, the US Department of Energy (USDOE) considered several alternative numerical methods, including explicit discrete fracture network approaches, but adopted a dual permeability continuum method based on its ability to match different types of moisture, geothermal, pneumatic and geochemical field data (see section 2.3.2.4.1.1.2 of Ref. [III–15]). In this approach, two continua are assumed to exist at each location in the model. These two continua represent the fractures and the matrix, respectively, with exchanges existing between them. The Richards equation for unsaturated flow is applied to both these continua. Darcy's law and the van Genuchten model for relative permeability and capillary pressure are generalized for multiphase flow under non-isothermal conditions. The solution of the relevant equations is described in Ref. [III–26] and was implemented in a special version of TOUGH2 (version 1.6) [III–26]. For fracture flow, an active fracture model was adopted, in which only a portion of fractures in a connected, unsaturated fracture network contribute to flow. The model presumes gravity dominated, non-equilibrium, preferential flow in fractures, such as fingering flow in unsaturated porous media. It does not assume that the fractures with smallest aperture are occupied first on wetting.

The applicability of van Genuchten relationships to flow in fractured media has been evaluated [III–27, III–28]. The van Genuchten approach reasonably matches water retention curves, but generally underestimates relative permeability, except at low fracture saturations.

In relation to the treatment of heterogeneity in rock properties, the USDOE states (see section 2.3.2.4.1.1.4 of Ref. [III–15]) that a geology based deterministic approach is adopted in which an entire model layer is assigned uniform properties. The justification offered for this approach is that the overall behaviour of unsaturated zone flow and transport is mainly determined by relatively large scale layered heterogeneities associated with the geological stratification of the mountain. Limited property variations within layers are attributed to the homogenization introduced by the process of tuff deposition.

Application of the model requires specification of a suitable grid and appropriate boundary conditions. Mountain scale grids were developed to capture large scale flow processes, with drift scale grids developed to model detailed flow processes near emplacement drifts and test beds (see section 2.3.2.4.1.2.2 of Ref. [III–15] and also Ref. [III–29]). The site scale model comprises 32 flow model layers with different hydrological properties and 59 computational grid layers. The boundary condition at the top of the model is a steady state, spatially variable flux. Lateral boundaries are treated as no-flow or closed boundaries that only allow flow along the vertical plane (based on the eastern boundary being located at the Bow Ridge Fault and the other boundaries being located far from the repository). The top and bottom boundaries of the model are both treated as open, with constant but spatially varying gas pressure and temperature.

Site scale coupled process models simulate the impact of heat released from emplaced nuclear waste on unsaturated zone flow, including thermal–hydrological, thermal–hydrological–chemical and thermal–hydrological–mechanical processes. These models are only described briefly in section 2.3.2.4.1.2.5 of Ref. [III–15]. However, it is relevant to note that the site scale model predicts little flux accumulation above the emplacement drifts, because most of the heat induced liquid reflux drains through fractures in pillars between the emplacement drifts (see section 6.2 of Ref. [III–30]). However, the drift scale model predicts increased saturation and flux due to condensation above the drift. This distinction emphasizes that grid refinement might not simply result in model results incrementally approaching a converged solution. Instead, when grid refinement passes a threshold in resolution, qualitatively different model results can be produced.

The results of thermal–hydrological–chemical modelling by the USDOE indicate that mineral precipitation and dissolution will not significantly affect the hydrological properties of fractures or the percolation flux in the unsaturated zone. The results of thermal–hydrological–mechanical modelling indicates that permeability changes range from a factor of 0.3 to 5 near the repository drifts.

The USDOE Engineered Barrier System flow and transport model estimates radionuclide transport through the degraded waste package and waste form, and subsequently through the invert.

The magnitudes of the fluxes F_1 to F_8 are spatially and temporally variable. Thus, for example, if there is no seepage into the drift and no condensation on the drift walls, F_2 to F_6 are zero. Subcomponents of the model are:

— Temperature and water content dependent diffusion within a degraded waste form and waste package;

— Advection within the degraded waste form, waste package and invert;
— Competitive equilibrium sorption–desorption of radionuclides onto fixed corrosion products and colloids in a degraded waste package;
— Reversible sorption onto crushed invert tuff.

A kinetic model is used for sorption of plutonium and americium on stationary corrosion products. The forward rate coefficient is a sampled parameter, and the backward rate coefficient is then calculated using the sampled K_d value. Desorption of plutonium and americium is not permitted from colloidal corrosion products.

Transport through the unsaturated zone below the repository is based on the same type of flow model as described above in relation to the unsaturated zone above the repository.

The USDOE approach to the representation of groundwater flow and radionuclide transport in the saturated zone is set out in section 2.3.9 of Ref. [III–15]. This can be considered in three parts, comprising the modelling of the three dimensional (3-D) flow field in a site scale model embedded in a regional scale hydrogeological model; modelling of the transport of radionuclides in the flow field derived from the site scale model; and abstraction of the radionuclide transport model to provide a representation of radionuclide transport for use in the overall assessment model. This last is achieved using breakthrough curves for single member decay chains or using sequences of GoldSim pipe elements for multimember decay chains. The overall approach adopted is described in Ref. [III–31].

As with the unsaturated zone, although discrete fracture network modelling approaches were considered by the USDOE, they were not adopted. Instead, a dual porosity effective continuum approach was implemented using the FEHM code [III–32] with orthogonal hexahedral elements and a horizontal grid spacing of 250 m. A non-uniform vertical grid spacing with a minimum value of 10 m near the water table was used. Because the model adopts boundary conditions from the regional model, it implicitly includes the effects of the pumping of wells that lie outside the explicitly modelled domain. The model is based on sub-horizontal zones, but crosscut with features that primarily represent faults, fault zones and areas of mineralogical alteration (see section 6.5 of Ref. [III–33]). The properties of these features are averaged over 250 m × 250 m grid blocks, though the features themselves may be much smaller. This was judged to be adequate for modelling the overall flow regime, but the USDOE recognized that for transport parameter values could need to be adjusted to take the smaller dimensions of the actual transport pathway into account (section 2.3.9.2.3.1 of Ref. [III–15]).

III–3.4. Major considerations in representing warm, arid biosphere environments

It is important to recognize that warm, arid environments vary greatly and there is debate amongst geomorphologists as to whether or not they can be considered as a single class. Indeed, it can sometimes be informative to compare styles and rates of processes between arid and more arid sub-humid environments, rather than limiting comparisons to just a single classification of arid environments [III–12].

Although aridity is not the sole criterion in distinguishing different types of arid region [III–34], it is a useful broad classifier. The United Nations Environment Programme (UNEP) World Atlas of Desertification [III–35] has classified deserts based on an aridity index defined as the ratio of precipitation to potential evapotranspiration computed from mean monthly data for the period 1951 to 1980. Hyper-arid regions (e.g. much of the Sahara) have an aridity index of <0.05 and occupy 7.5% of the world land area. Arid regions (e.g. central Australia) have an aridity index of 0.05 to 0.2 and occupy 12.1% of the world land area. Semi-arid regions (e.g. much of the southwest of the USA) have an aridity index of 0.2 to 0.5 and occupy 17.7% of the world land area. Finally, dry, sub-humid regions have an aridity index of 0.5 to 0.65 and occupy 9.9% of the world land area.

There is a complex interplay between the ecology, geomorphology, hydrology and hydrogeology of desert areas that is only now beginning to be explored in transdisciplinary and interdisciplinary studies [III–36]. Thus, the modelling of how arid areas will develop in response to changes in climate is challenging, especially as extreme events could be of great importance, as changes in extremes are less well predicted in climate models than are changes in average values of quantities such as temperature and precipitation.

As shown in the two examples discussed in Section III–3.1, under arid conditions, the biosphere is of importance in determining the fluxes of meteoric water reaching the repository horizon, interacting with the disposed wastes and percolating to greater depths to reach either the water table or a perched water body. Thus, the modelling of water flow and contaminant transport in unsaturated, heterogeneous and sometimes fractured media requires consideration. This is much more complex than the corresponding modelling in saturated media due to the highly non-linear relationships between the degree of saturation, hydraulic conductivity and matrix potential, as well as a lack of knowledge concerning the degree and kinetics of sorption of key radionuclides in such conditions. In addition, under altered climatic conditions, the characteristics of the surface water and groundwater flow systems can change markedly. Thus, fluvial episodes can occur in which ephemeral surface flows are replaced by continuously flowing streams and rivers, as well as lakes. Also, groundwater levels can rise by many tens of metres, as evidenced, for example, by paleodischarge deposits local to Yucca Mountain (see Ref. [III–37] for a detailed discussion of the paleohydrology of Yucca Mountain and its vicinity).

The limited vegetation cover that is typical of arid regions makes them susceptible to erosion. This could include generalized denudation, slope retreat and gullying, with both aeolian and fluvial processes of potential significance. In young landscapes, such as that at Yucca Mountain, recent tectonic uplift can mean that the landscape is far from equilibrium, tending to further enhance rates of erosion [III–38]. A detailed account of the erosion of badlands, such as those that characterize the area around Yucca Mountain, is provided in Ref. [III–39].

In terms of the biosphere into which radionuclide releases could occur, it is possible that those discharges can be to streams and spring lines in valley locations. This can apply where the uplands exhibit an arid environment, but deeply incised valleys exist and are characterized by a more humid climate and denser vegetation cover. On the other hand, as illustrated at Yucca Mountain, the high demand for water and lack of surface or near surface supplies can lead to an emphasis on extraction from deep boreholes and can also mean that deep groundwater resources are used at close to their sustainable yield. In these circumstances almost all of the contaminant plume could be captured by a single well or a small well field. In an arid environment, intensive use of this water could occur, for example in respect of high irrigation rates, to obtain several harvests per year. Indeed, a significant degree of overwatering could be necessary to prevent salt buildup in the soil. Also, special approaches to irrigation can be used to limit evaporative losses, for example irrigation directly to the root zone rather than to the surface. Where discharges of contaminated water occur at the surface, intense evaporative effects may result in the formation of solid contaminated discharge deposits similar to the paleodischarge deposits that exist in various arid areas at the present day.

III–4. CONSIDERATIONS IN PLANNING AND THE USE OF SITE CHARACTERIZATION INFORMATION IN BIOSPHERE MODELLING

III–4.1. The role of site characterization for modelling of the biosphere

During the MODARIA II WG6 work to enhance the BIOMASS methodology, a need for examples was recognized, showing how site characterization has been conducted to support modelling of the biosphere in post-closure safety assessments. Even though site characterization as an activity is not part of BIOMASS, the information gained and the use of site data and site understanding in safety assessments are crucial to support and build a case for safety after closure. Site data and models are already used from a generic stage to find and evaluate sites as well as when to verify a specific site and its conditions. Different types of site investigation and site modelling activity will also be conducted during repository construction and operation. Therefore, examples from ongoing national programmes can function as a guide for programmes that are at an early planning stage.

This text is a summary view on work performed during site investigations and site modelling activities in Sweden and Finland associated with safety assessments and repository construction applications, with a focus on recent developments. It is compiled to provide examples of strategies and methods to characterize a site, with a focus on the surface environment used for assessing safety after closure. All repository projects differ in their overall assessment contexts and unique site specific properties, features and/or processes. Therefore, the reader needs to carefully adjust the tasks to fit with the assessment context at hand, if the general methods described herein are applied. Also, the level of ambition in site characterization is causally linked to the needs. Therefore, a graded approach that evaluates the proportionality is also relevant to use in site characterization planning.

The efforts to characterize reasonable locations for spent nuclear fuel repositories in Sweden and Finland started in the 1970s, and the work was arranged to gradually focus on a few sites and then ultimately a single one (Fig. III–4). However, to benefit from the latest knowledge, this annex has been arranged to describe the earlier stages only briefly and is focused on the integrative site description methodology, noting that it would be reasonable to also apply this methodology from early considerations of the geographical site context. Thus, the discussion below describes the general strategy used by SKB and Posiva in recent site characterizations to describe the natural system of the Forsmark and Olkiluoto sites to inform safety assessments for deep geological disposal of spent nuclear fuel (see e.g. Refs [III–40, III–41]). Due to the extensive nature of the site characterizations, the references given here primarily concern the original work, and summarize the step by step building up of relevant site understanding in repository projects [III–42–III–45). When referring to the specific methods described below, the readers need to go back to these original publications for further details.

The site characterization serves as a basis for all activities related to nuclear waste management, from supporting the site selection to applications for construction, operation and closure. Needs based on site specific properties and processes can be found in all different functions within the programme, including repository design, other aspects of the site characterization, environmental impacts, radiological baseline and discharge monitoring, operational safety and safety after closure (‘long term safety’). Therefore, depending on the stage of the programme (Fig. III–4), site understanding will have different purposes and play different roles for different end users. In addition, the step by step buildup of site understanding contributes directly to overall confidence in system behaviour and function, supporting applications to move forward in the programme. This stagewise strategy results in an iterative process where site understanding is constantly updated and fine tuned [III–46]. For the present context, however, the focus in this annex is on those needs governed by (long term) safety assessments.

National programmes have always been a driver for developing and enhancing strategies in site characterization methods. The concrete tasks and regulatory demands in real national nuclear waste management programmes are guiding and focusing the work. The need for scientific support for the results given, and in the applications filed, ensures that best practice continues to reflect the latest scientific findings. However, the international forums and organizations in this field have an important role in taking the overall and generic framework of best practice forward. For both SKB and Posiva, the international guidelines are an important basis, together with the lessons learned from earlier stages in the repository programmes. Publications that have been guiding the site characterization programmes in Sweden and Finland include:

— IAEA Safety Standards Series No. SSG-35, Site Survey and Site Selection for Nuclear Installations [III–47];
— IAEA Safety Standards Series No. SSR-1, Site Evaluation for Nuclear Installations [III–48];
— IAEA Safety Standards Series No. SSR-5, Disposal of Radioactive Waste [III–49];
— IAEA Safety Standards Series No. SSG-23, The Safety Case and Safety Assessment for the Disposal of Radioactive Waste [III–50];
— IAEA-TECDOC-1755, Planning and Design Considerations for Geological Repository Programmes of Radioactive Waste [III–51];
— IAEA Safety Standards Series No. SSG-31, Monitoring and Surveillance of Radioactive Waste Disposal Facilities [III–52];
— IAEA Safety Standards Series No. SSG-14, Geological Disposal Facilities for Radioactive Waste [III–11];
— NEA/RWM No. 6923, Methods for Safety Assessment of Geological Disposal Facilities for Radioactive Waste [III–53];
— ICRP Publication 122, Radiological Protection in Geological Disposal of Long-Lived Solid Radioactive Waste [III–54].

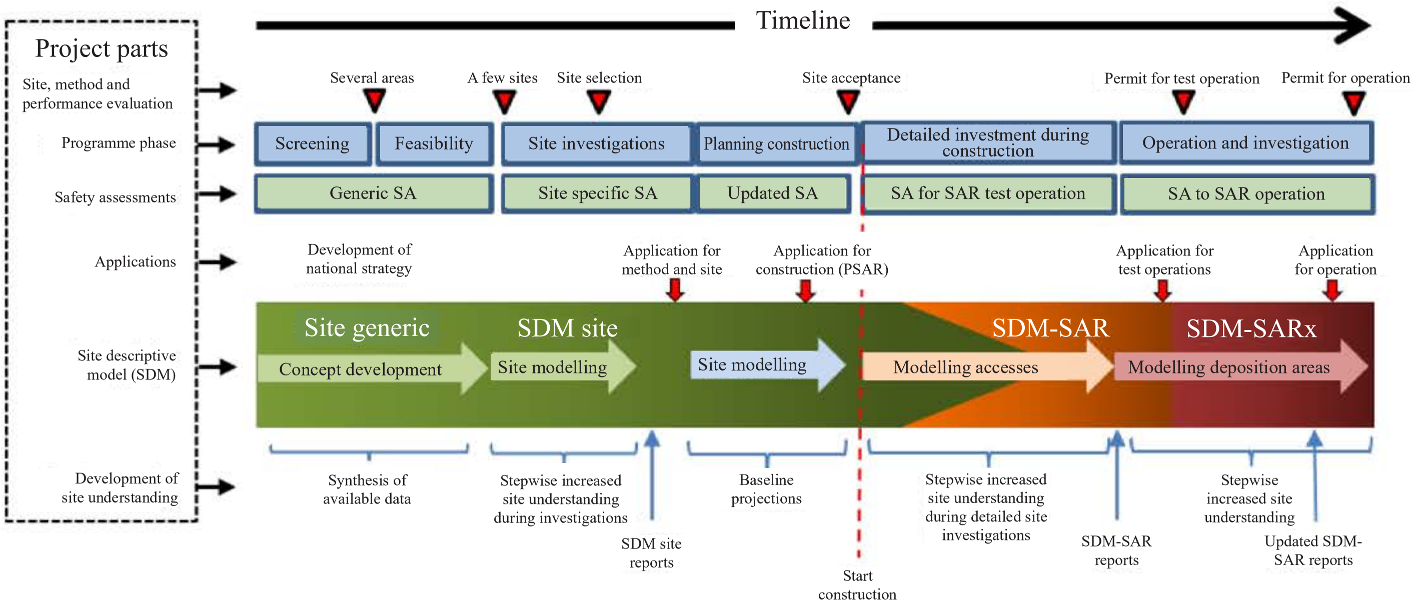

FIG. III–4. Schematic timeline showing the stepwise process of site characterization in the Swedish nuclear waste management programme, going from site generic to application for operation. SDM — site descriptive model; SA — safety assessment; SAR — safety analysis report; PSAR — preliminary safety analysis report.

During feasibility studies on a national scale, a general characterization is already needed. This need stems from both testing the repository method suggested and defining suitable environment types. This work mainly comprises bedrock type screenings, or ‘the search for suitable host rock blocks’ (see e.g. Ref. [III–55]). However, since the whole natural system needs to be considered when a site is selected, a general understanding of site characteristics is needed for regional areas suggested during the feasibility phase. This includes descriptions of land use, logistics options, societal factors and avoidance of environmental protection targets [III–42, III–56]. The generic safety analyses for different types of environment suitable for suggested repository methods need to incorporate the engineered barriers, the geosphere and the biosphere system to assess the total system performance (see e.g. Refs [III–57–III–59]). These first generic site characterization efforts are useful for planning future site characterization programmes and are also a good tool for developing exposure models and safety assessment techniques adapted to environments that may be encountered once sites of interest have been selected.

The results from screening and feasibility studies that considered several areas and regions were the basis for the selection of a site or a few sites to investigate and characterize in more detail. The general strategy was:

(a) Site or sites need to be selected that provide sufficient long term safety;
(b) If no clear difference can be detected in long term safety, other aspects will guide the selection [III–42, III–55, III–60].

In Sweden, two sites, Laxemar/Simpevarp [III–61] and Forsmark [III–62–III–64] were initially chosen. Of these, SKB later chose Forsmark as the site for a repository construction application [III–65]. In Finland, the more intensive efforts after the feasibility studies were directly focused on a single site, Olkiluoto [III–55].

III–4.2. Planning the site investigations

The site investigation and site modelling strategies have followed a stepwise advance approach in the overall repository programmes. This means that site investigations in general have a graded approach going from generic feasibility studies, through initial surface based investigations to detailed site investigations in the bedrock at repository depth, by drilling or even underground pilot construction, as in the Finnish example. Between these steps, a review is undertaken by the authorities and a go/no go decision is made for the overall programme, depending on the results, as well as possible specific requirements regarding specific topics of concern. The need for a constant dialogue with stakeholders has been and is a crucial aspect of the constantly updated planning on further site characterization.

Data from the site investigations has several key end uses, including:

- Concept and repository design and construction;
- Environmental monitoring and management;
- Post-closure and operational safety (including monitoring of repository performance);
- Development of sufficient understanding of the site as part of the total disposal system.

Site and system understanding are difficult to specify in terms of needs but are nevertheless of extreme importance. The support that a proper site understanding contributes to the whole programme can never be overestimated. The site descriptive model, together with the repository layout, if properly integrated, will show professionalism and site adaptation, and build confidence in the national programme, which have been crucial to outcomes in the SKB and Posiva cases so far. To execute investigations to obtain the data and models needed for the above tasks, thorough requirements planning was performed [III–42, III–43] and has continued (see e.g. Ref. [III–66]). Much effort was invested in integrated data sampling campaigns, monitoring installations and laboratory analyses to make sure that the needs of all the end users were fulfilled in each activity as far as feasible. For instance, sampling for environmental monitoring and for radionuclide transport parameter (distribution coefficients, concentration ratios, etc.) analyses can be achieved together within the same field campaign, saving significant resources.

Two main strategies were used in parallel to plan the site investigations and modelling activities at the Forsmark and Olkiluoto sites. The first comprised listing the needs and requirements of end users (see e.g. Ref. [III–42]) and the second comprised generally constructing conceptual and numerical models of the natural system for each scientific discipline [III–67]. The second method was built on what questions there were to be answered. This type of applied scientific method provides a tool to define the output needed for the suggested answer, then the supporting models to test the hypothesis, and finally the site data needed to run the models.

As an example, take the question, ‘How long have there been lakes close to the shoreline at Forsmark?’ To answer this question, an understanding of the processes involved in relative sea level changes and lake succession is needed. This exercise results in a process list that can be transformed into several field activities that collect the data needed. Once the data have been collected, it is possible to backtrack the method by producing models driven by site data and process rates and produce an answer, as well as answers to other related questions; for example, ‘How long will the lakes be there if present processes and their rates do not change?’

A few well defined questions related to landscape evolution and element transport will help conceptualize the whole site for assessment purposes (see Refs [III–68–III–70]). Also, by reporting interim results and having feedback loops between end users and the site investigations, the site characterization planning can be (as in the present case) constantly updated and refined. This strategy can also be found in the BIOMASS methodology, in which ‘system understanding’ sits in the middle of Fig. 2 in the main text.

III–4.3. Site investigations and site descriptive modelling strategy

The overall site investigation and modelling tasks are divided into two main stages in the programmes performed at SKB [III–42] and Posiva [III–43]. First, there is a surface based phase, with investigations and modelling performed to describe the sites to support decisions on the repository concept and siting. A second phase is then initiated when underground excavations are permitted. Therefore, a division is made between site characterization with an initial surface and drilling based investigation programme and a detailed programme also exploring the bedrock at depth during repository construction. For the surface environment part of site characterization, the investigations can be seen as a continuous programme with interim goals and updates consistent with the stages and aims of the overall repository programme (i.e. corresponding to the maturity of the site understanding needed for safety assessments and key decision to proceed onto the next stage).

Learning from the experience of SKB and Posiva, the environmental sampling supporting the integrative site description needs to be oriented as early as is feasible into coherent datasets recording as many parameters as possible. It is recognized, however, that this is not always possible — or even reasonable — for several practical reasons, but it needs to be strived for by allocating sufficient resources in planning and coordination. In any case, it is important that the survey, sampling and analytical methods are compatible with each other to allow for data integration [III–71], even if everything cannot be implemented within the same campaigns.

With respect to collecting and managing information on ecosystem properties, SKB in Sweden and Posiva in Finland have utilized different approaches, partially due to the site characteristics and in part due to the pacing of the overall programmes. In Sweden, SKB’s sample collection has been conducted in parallel with conceptual model development and classification of the landscape, resulting in the description of the mass balances in terms of catchment areas (see e.g. Ref. [III–72]). In Finland, Posiva's site is an island on a water divide, hosting only two main catchment areas that are reasonably homogeneous in their ecosystem properties. Given this and the need to monitor the impact of the rather heavy land use pressures at the site and in its vicinity, Posiva has developed a hierarchical network for managing information. This network includes information from surveys (e.g. general descriptions of land use and vegetation patterns), from a number of measurement plots, and from a few 'intensive level measurement plots', which are instrumented for more or less continuous monitoring activities (see e.g. Refs [III–41, III–66, III–73–III–75]). In Posiva's case, there has also been a greater need for present day analogue studies further away from the site to satisfy the information needs of safety assessments, where similar measurement and sampling plot designs originally oriented to terrestrial systems have been applied and further adapted to aquatic environments. However, in both of the Swedish and Finnish site characterization programmes there have been numerous exceptions from the overall scheme for many valid reasons; the survey and sampling methods are explained for each campaign in the respective reports referred to in the summary reports indicated in this annex.

Another strategic aspect of striving for integrated and coherent quantification of the ecosystem fluxes and storages employed by both SKB and Posiva is the use of stable element information as a proxy for the behaviour of long lived radionuclides relevant to the safety assessments (see e.g. Refs [III–76, III–77]). Even though there are general needs for studies of environmental radioactivity at the site and of laboratory experiments with radiotracers (e.g. batch sorption experiments), the experience is that the stable element analogues allow for considerable expansion of the data basis on the behaviour of the assessment relevant elements as well as many others, information on which can be used to provide supporting data. The stable element concentrations and inventories are also, mostly, directly measurable from the samples taken throughout the ecosystem components, and they also are an effective means to build models of landscape level matter fluxes and storages, and to detect uncertainties through imperfect closure in the ecosystem level or landscape level mass balances (e.g. see Refs [III–76, III–78]).

III–4.4. Field surveys and sampling

The initial and surface based site investigation programme at Forsmark is described in Ref. [III–42]. The input from all scientific disciplines is used to describe the data and analyses needed, as identified from the generic conceptual understanding provided by the preparatory work described above. The description of site field sampling and synthesis for the surface environment can be found in three reports and references therein that describe the terrestrial ecosystem [III–79], the limnic ecosystem [III–80] and the marine ecosystem [III–81]. These reports were extended by the same group of people in the safety assessment SR-Site, with information gained during the use of site data, (e.g. element and K_d data and biosphere models; see Refs [III–82–III–84]). The strategy used by SKB to have the same group of people who planned and reported the site characterization be involved in the safety assessment was a key decision and is a lesson for others on how to organize experts in a programme to successfully transform the site understanding gained during the site characterization into the safety assessment.

For the Olkiluoto site in Finland, an iterative succession of such summaries has been presented (see e.g. Refs [III–44, III–45, III–85] while the 'biosphere description' series (see e.g. Refs [III–73, III–86, III–87]) similarly provides further references to a number of individual reports that elaborate in detail.

Basically, the survey and sampling methods regarding the surface environment are similar to those applied to the field research in the respective scientific disciplines. However, there is much higher demand for coherence in the methods and for integration of the results, and thus the methods need to be selected carefully and often somewhat adapted for the site characterization. Also, the broad but focused combination of individual methods that was implemented in SKB's and Posiva's programmes is rather unusual and has therefore attracted interest from universities for conducting research within site investigation areas. The benefits of this type of stakeholder involvement in terms of confidence and acceptance can be key to a successful programme.

It is important to note that field work typically comprises a major share of the cost and effort in site characterization, compared with laboratory and data analyses. One approach to saving resources is to acquire self-consistent multiple parameter data ('snapshots') of the ecosystems by taking samples for as many purposes as possible within the same campaigns. This also aids in drawing the overall picture for the integrated site descriptive modelling compared to trying to join together individual pieces of data, although it requires dialogue, coordination and planning.

III–4.5. Site modelling

The modelling on site properties and processes performed during the site characterization is an iterative process both in respect of modelling disciplines and model versions and regarding the feedback to and from field work and analyses of the samples [III–42, III–46]. The surface environment modelling method as well as results for the Forsmark site are described in Ref. [III–67] and a further synthesis is made in Ref. [III–88] for issues related to safety after closure. For the Olkiluoto site, the most recent summaries of modelling activities supporting the surface environment part of the site descriptive modelling can be found in Ref. [III–73], with complementary information given in Ref. [III–77] and also references therein.

The site descriptive modelling, as a multidisciplinary interpretation of the geosphere and biosphere (surface environment), uses site investigation data from the surface as well as soil/bore holes. The modelling comprised evaluation of primary data, descriptive and quantitative modelling, and overall confidence evaluation in an iterative manner.

Data were first evaluated within each discipline and then the evaluations are checked between the disciplines. As a starting point, the site geometries were modelled (elevation and bathymetry) together with soil and sediment stratigraphy and thickness. Hydrological modelling and ecosystem description and mapping, together with determination of the chemical properties of the biotic and abiotic parts of the sites, followed. After the individual discipline modelling, a phase of overall confidence evaluation followed. This included integrated conceptual modelling describing the site properties and division of the site in relevant domains for the identified processes that govern element transport, including water. The findings as well as the modelling results were documented in a site description.

The site descriptive model (SDM) [III–46, III–63, III–89] is a discipline integrated numerical and qualitative description that, by iterations during the repository programme, sets out the latest site understanding. It consists of mutually supporting models from all disciplines, including both the bedrock and the surface environment, and provides the data needed to carry out the next steps in the repository programme in respect of safety assessment, repository design and environmental impact assessment. This product is updated (and reported) for each major application or assessment of safety after closure.

The key stages of developing such a spatially explicit model for the surface environment (‘biosphere’) include [III–71]:

— Building a tentative conceptual ecosystem model describing the matter storages and fluxes based on initial data and expert judgement (also described in Section 5.1.3 of the main text);
— Collection of site specific and generic data for mathematical representation of these storages of fluxes in an appropriate spatial and temporal resolution (e.g. long term average storages and fluxes for each catchment area of interest, including the vertical layering);
— Description and quantification of the processes affecting the flow and accumulation of matter within (e.g. the terrestrial, limnic and marine ecosystems and across their boundaries);

— Construction of a site specific linked ecosystem model to address the transport and accumulation of matter at the landscape/site scale;
— Active exchange of information between disciplines, as well as between the levels of implementation (from field work to data synthesis, and back), leading to iteration and updating of the stages above with the accumulating data, information and knowledge.

III–5. SITE ANALOGUES

Safety assessments for radioactive waste repositories cover a time span from closure up to one million years or more. Scientific information and knowledge on processes related to the far future characteristics of a site are therefore needed. For areas associated with possible future climate variants showing glacial and/or periglacial conditions, an understanding of cold climate environments is needed. Also, earlier safety assessments in Sweden [III–90] showed that, for sites located in previously glaciated terrain, the impacts of glacial (ice sheet) and periglacial (permafrost) processes need to be included and addressed. These processes influence the environment and have the potential to directly or indirectly affect repository safety (see e.g. Refs [III–90–III–92]). In addition, these processes affect the landscape development at a repository site, which in turn will affect the radionuclide behaviour in the biosphere as well as land use.

Analogue sites and the information gained from analogues have a broad range of uses, from repository design criteria to input in scenarios used to describe the far future and associated risk for release scenarios. This means that the use of analogue sites is part of the iterative process going from generic assessments through feasibility studies, siting, construction and finally the operational phases. Site analogues can therefore be used as a support in all repository stages and for all questions at issue.

It also needs to be noted that the use of site analogues is not limited to the biosphere and that geosphere studies within the same site or region increase the overall system understanding needed for assessments of possible far futures (see e.g. Refs [III–2, III–93].

Different types of site analogue can be distinguished:

(a) Climate analogues with a present day climate that mimic assumed future climate conditions for the site of interest;
(b) Space for time analogues that can be used to capture the effects of time dependent processes at a site by studying site analogues that exhibit a historical succession captured in features, events and processes.

For example, it is possible to consider the potential effects of shoreline displacement processes by studying other sites going from coastal sea level to inland areas in a transect. Other types of analogue can be of use in specific assessment contexts.

Analogues can be used to capture the conceptual understanding needed to construct models of radionuclide behaviour in a future biosphere and/or to provide safety assessment with data that are otherwise not available. Below follows a short example from a site analogue project conducted by SKB in Sweden that used a catchment area in Greenland as an analogue for future periglacial conditions at the Forsmark site.

In 2010, an SKB funded project (GReenland Analogue Surface Project, GRASP) was initiated with the aim of better understanding features, events and processes associated with periglacial climate conditions [III–2, III–94, III–95]. A lake catchment was selected in the Kangerlussuaq area to function as a study site (Two Boat Lake, TBL) (see Fig. III–5). Also, a parallel project (Greenland Analogue Project, GAP) was initiated at the same time looking at ice sheet and bedrock processes and is reported in Ref. [III–93].

The relatively small catchment of TBL covers an area of 1.56 km^2, with the lake constituting 25% of the total catchment area (Fig. III–6). Two Boat

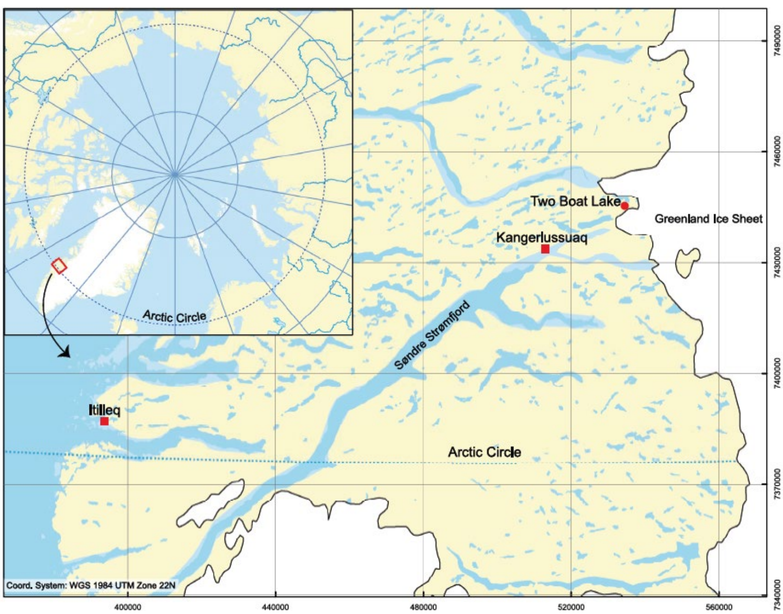

FIG. III–5. Map over West Greenland and the location of the Two Boat Lake study site (reproduced with permission from Ref. [III–95]).

FIG. III–6. Photograph (facing southwest) showing the Two Boat Lake catchment used as periglacial analogue for the present temperate climate at the Forsmark site (photo credit: T. Lindborg).

Lake (TBL) is situated approximately 30 km northeast from the settlement of Kangerlussuaq, western Greenland, close to the Greenland ice sheet. The region around Kangerlussuaq is characterized by a hilly tundra landscape with numerous lakes and continuous permafrost interrupted by taliks beneath lager water bodies.

The GRASP project had two components. The first part focused on how ecosystems develop and behave from a long term climate change perspective during an entire glacial cycle [III–70]. The second part aimed to improve understanding of water exchanges between surface water and groundwater in a periglacial environment [III–94]. Hydrological and biogeochemical processes and conditions not considered in temperate climate regions could be of great importance in periglacial areas. Hydrological responses in these cold areas differ in fundamental aspects from those in catchments in boreal and temperate regions. Most importantly, the hydrology in periglacial environments is intimately connected with the presence of permafrost and the active layer dynamics. Snow related processes have also been shown to be of great importance for the annual water balance. These special hydrological conditions in turn have large impacts on biogeochemical processes, since hydrology and aeolian transport are the main drivers for the transport of elements in the landscape. The aim of the hydrological field studies illustrated here was to identify and quantify the main hydrological processes in a periglacial lake catchment, providing input to conceptual and mathematical modelling (see Fig. III–7).

The overarching target when planning the hydrological TBL field programme was to identify and quantify the main hydrological processes, including the interactions between surface water in the lake and that in the surrounding catchment, and the role of both supra- and sub-permafrost groundwater. In addition to the scientific questions to be answered, the hydrological measurement programme was, to a large extent, determined by

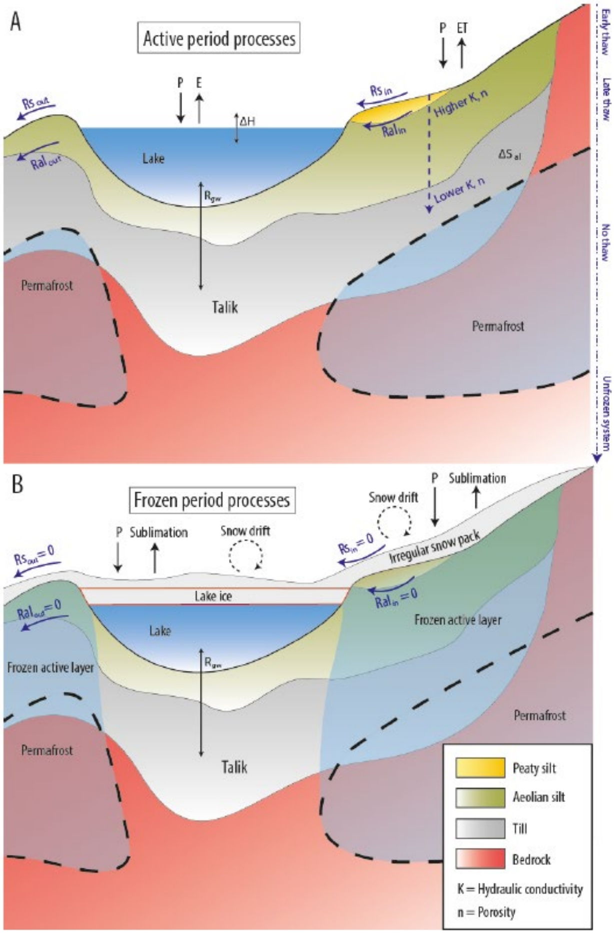

FIG. III–7. Conceptual model of the hydrological flows during the active (a) and frozen (b) periods in the Two Boat Lake catchment (adapted with permission from Ref. [III–94]).

what was possible, given the hydrological, climatic and logistic conditions at the site. Due to its remote location, but also depending on the harsh climate, people were only on-site for relatively short periods. This limited the possibilities for long term or continuous observations of some parameters that could not be measured automatically (e.g. surface water inflow to the lake or groundwater monitoring in the active layer). Lack of infrastructure for electrical power and telecommunications was also a limiting factor. Typically, three field campaigns were organized per year (in April, June and August–September) during the period for which data are presented.

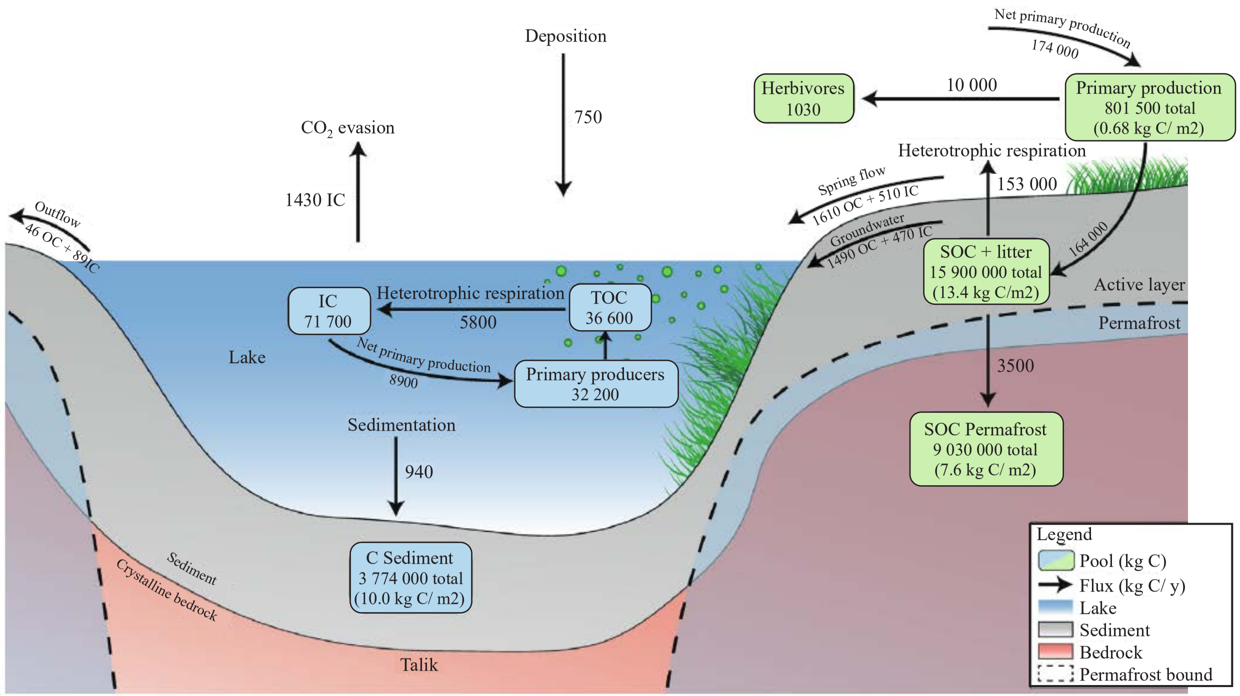

FIG. III–8. Carbon landscape ecosystem model for TBL catchment showing major pools and fluxes. Illustration adapted with permission from Ref. [III–78].

The general biogeochemical sampling and investigation strategy was to obtain a broad picture of element distribution and not to focus on any specific target element [III–95]. The aim was to provide understanding of the general fluxes of dissolved and particulate matter in the system [III–78]. Sample analyses were performed to determine concentrations of carbon, nitrogen and phosphorus, together with associated species and isotopic composition, as well as total concentrations of a long list of major and trace elements and isotopes, which together with the hydrological model helped in understanding and calculating fluxes in the landscape. Age determinations for soils and sediment layers, together with estimates of biomass and primary production in both the terrestrial and aquatic systems, made it possible to calculate the accumulation and mass balances of various elements in different landscape and ecosystem units (see Fig. III–8 and Ref. [III–78]).

III–6. DEFINING THE AREA OF INTEREST

III–6.1. Introduction

This text builds on the discussions and presentations made during the MODARIA II WG6 workshops on representing the site in a safety assessment model. It shows, as an example, the issues that need to be explored, the background information to assess, and a way to handle the linkage between the site understanding and the dose assessment model representing the site. The example given below summarizes the development done, and lessons learned, during the work performed by SKB to underpin licence applications for constructing and extending repositories in Sweden [III–90, III–96, III–97]. All repository projects are different regarding their overall assessment contexts and will encounter both global general and unique site specific features or processes. Therefore, the tasks need to adjusted carefully to fit the assessment if the general methods described here are applied. When referring to the specific methods described below, it is important to consider and refer directly to the original referenced publications. The emphasis is on providing an easily digested summary that focuses mainly on the method parts of the task to identify the areas and volumes to be used in dose calculations.

Any dose calculation needs information on the natural system and human behaviour within that system. Many methods to obtain this information have been proposed and applied [III–40, III–98–III–101]. The aim here is to give an example of one of several approaches for identifying areas of interest for dose calculations. Furthermore, this section describes the use of synthesized site specific understanding to define not only the area of interest, but also the implications on land use and human population size relevant for dose calculations. This includes a description of the methods used by SKB in recent assessments to identify areas and volumes, with site specific properties, to inform the SKB dose model (see Refs [III–102, III–103]). The assumptions made and rationales used are presented, together with a step by step strategy to delineate the physical parts (areas and volumes) of the site of interest relevant in dose assessments for the Forsmark site [III–67, III–104].

III–6.2. Site understanding

In all safety assessments, and particularly in a site generic safety assessment, it is necessary to make assumptions about the site, for example a land based site within the territory of Sweden. This site information gradually improves as steps are taken towards site selection and geological facility site adaptation and construction. The emerging site understanding builds on models describing, for example, the topography/bathymetry, soils, hydrology, chemistry, geology, hydrogeology, ecology and human land use. If time dependent site properties are to be used in the assessment, then it could be possible to develop a landscape development model from the information.

Also, assumptions need be made about the repository. This repository could be displayed either in the geosphere as a planned layout or as a generic repository over an area at relevant depth to illustrate all possible repository locations (potential release points/areas) within the geological volume.

If no site data are available or no site has been selected, generic site descriptions and assumptions made on regional settings for a preferred site can be used. Areas with good data and site understanding can be used as analogues even if they do not represent potential candidates for repository construction (e.g. SKB's safety assessment for an ILW repository) [III–105, III–106].

If the time frame of the assessment covers a long time period (e.g. several hundred years up to one million years), the major driving forces on landscape development need to be considered [III–1, III–2, III–72, III–104]. Global, regional and local climate changes have been found to be a major driver for site specific processes [III–107]. Depending on the location and site properties, the site will respond differently in a changing climate [III–70]. Long term changes due to, for example, climate, affect process rates, for example, erosion, shoreline displacement, lake development, soil and sediment accumulation and land use. These changes can be assessed by applying models describing processes associated with identified future site developments. The combined synthesis of long term processes affecting the landscape can be integrated into a landscape development model. For further discussions of landscape development methods and strategies see Refs [III–1, III–72, III–104].

III–6.3. Discharge areas of contaminants

A first step to define the area of interest to be used in a safety assessment is to identify where a possible future release of radionuclides could occur. Even before any modelling has been performed, expert judgement on potential discharge areas can be used to establish a first conceptual model for the site. For the Forsmark site, the surface water bodies were assumed to be sites of discharge of groundwater.

Next, the site description with its hydrogeological models was used to simulate particle tracks from the repository (see Fig. III–9) [III–108]. Potential flow and transport paths from the repository via the geosphere to the biosphere are shown in Refs [III–109, III–110].

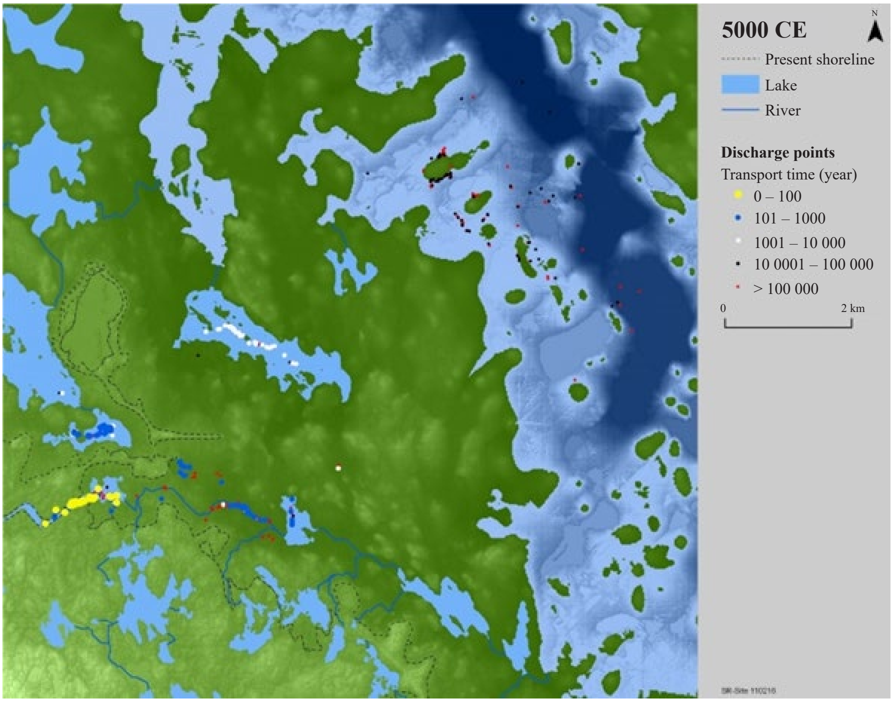

FIG. III–9. Example of discharge locations for a specific future scenario displayed on a 'map'. The discharge locations are coloured in accordance with advective travel times for the future 5000 CE flow field at the Forsmark site in Sweden. The topography and bathymetry at 5000 CE are indicated by different shades of green and blue, respectively; darker shades correspond to lower elevations. Note that infilling of lakes has not been accounted for in the map; instead, the largest extent of each lake is shown. Figure adapted with permission from Ref. [III–104] based on data from Ref. [III–108].

It may be useful to use several software codes to ensure that the processes involved are properly captured, for example specific codes for hydrogeological modelling and specific codes for hydrological modelling of surface processes. Once the particle tracking simulations have been performed, manual comparisons and expert judgement can be used to validate the results; the question can be asked as to whether the modelled areas as discharge areas can be identified at present. If it can, then the potential discharge areas of deep groundwater from the repository volume for that specific site context have been identified. If not, the supporting site models and the conceptual site understanding need to be revisited. Usually discharge areas in Fennoscandia are in local depressions such as sea basins, lakes, wetlands and streams [III–110].

According to the particle tracking models, contamination released to deep groundwater will migrate in accordance with the pressure gradient. This requires a good understanding of the local and regional hydrological properties of both the bedrock and the surface system to identify potential discharge areas at the surface. For a site with appropriate properties, contamination released from a modelled repository will travel long distances and for long times before entering the surface system. Therefore, understanding of the potential for discharge of deep groundwater to the surface is a joint task that requires close cooperation between the biosphere modelling team and those involved in site characterization and hydrological/geohydrological modelling. The resolution of the hydrogeological model and the features contained therein will have an impact on the point release pattern on the surface. In particle tracking simulations, this could lead to particles emerging in the surface system at places with no other support for discharge. Therefore, the discharge areas can only be defined after removing obvious outliers (modelling artefacts) caused due to model constraints and it may be necessary to conduct more detailed modelling exercises and site investigations.

III–6.4. Identify areas of interest for modelling of the biosphere

In the section above, discharge areas were described as identified using site models together with other information directly from the site. This will give a general picture of the potential discharge pattern at the landscape level, as well as specific locations for discharge points. To delineate areas of interest the pattern can be used to identify areas with typical properties associated with groundwater discharge and land use. Present day ecosystems can be displayed on a map that overlies the discharge areas to link specific discharges with typical ecosystems (e.g. lake, sea, wetland) [III–104]. Such linked areas of discharge and ecosystem type are sometimes termed ‘biosphere objects’. A biosphere object is defined by SKB as an area in the landscape that potentially, at any time during the assessment period, could receive discharge of deep groundwater associated with the repository volume [III–104].

The steps for identifying biosphere objects include the following:

— Use expert judgement to assess the discharge pattern at the present day by visual inspection. This includes considering regional bedrock fractures, the regional hydrogeology, the site geometry and the ecosystem pattern.
— Use a groundwater model and particle tracking studies to simulate flow paths from the repository to the surface and identify areas of potential groundwater discharge. This step could have to be repeated iteratively for several flow paths and/or repository release scenarios. The set of all possible discharge areas will therefore be larger than that from any individual release scenario.
— Apply visual techniques and/or frequency/geostatistics to the groundwater discharge point pattern to establish the major clusters and identify discharge outliers. If time dependence is applied in the assessment, this is also done for time steps of relevance to the landscape model.
— Confirm (if possible) present day discharge areas in the field and by using other supporting site specific information (e.g. vegetation, chemistry, hydrology).
— Allocate present day ecosystems to the discharge areas identified in the steps above and use their delineation to encircle areas of interest. The final areas (biosphere objects) can be used as the basis for site data input to dose calculations.

It is notable that the pattern of calculated discharge locations is not the main information used as a basis for delineating the objects. The objects are primarily delineated based on ecosystem considerations, and the discharge locations are used to help identify which objects to include in the transport and dose modelling. One practical advantage of having the object delineation partly separated from specific release scenarios is that the delineation is more likely to match with alternative sets of discharge locations, early as well as late in the assessment period. Experience shows that changes in release patterns are to be expected several times during the repository programme time frame. This is due to causes related to changes in repository inventory, repository layout, future climate variants, release scenarios and, in time, increased site understanding leading to new model results.

III–6.5. Biosphere objects and their use in modelling

The delineation of biosphere objects starts with the identification of sea basins (potential future catchments below present sea level) and lakes (present and future). This implies that the whole landscape on a regional scale is used to describe the discharge pattern of deep groundwater. However, the biosphere objects are always associated with specific ecosystems with homogeneous properties and potential for human land use within that landscape.

The size of a biosphere object is likely to change in time. For the Forsmark site, active crustal rebound processes have been acting on the relative sea level since the latest glaciation (shoreline displacement). When a biosphere object is below the sea, the whole basin is the biosphere object. As the sea basin emerges from the sea, the biosphere object size changes accordingly (see Chapter 7 in Ref. [III–104]). If the object turns into a lake (i.e. at the time when the bay is cut off from the sea and becomes a lake), the object size equals the lake size. After this stage, the object size is constant, but the lake/land ratio within the object will change continuously due to infilling and ingrowth of the lake, until the whole object area is land. This natural landscape succession is highly site specific and associated with uncertainties not discussed here (see Refs [III–70, III–72].

The delineated and parameterized biosphere objects can be used to estimate radiation doses and risks to humans and other organisms. This task in the safety assessment needs to produce results that are reasonable and representative of the considered site. This may sound obvious, but to be able to relate to the present site behaviour when interpreting the results is an important cornerstone when arguing that there is confidence in the safety of the proposed disposal system. The use of a broad set of biosphere objects allows the analysis to explore the effects of uncertainties (e.g. the sizes and positions of discharge points and ecosystems in the landscape, different land use practices).

The size of a biosphere object is the size of the homogeneous ecosystem (marine basin, lake, wetland, stream) in the landscape. To make sure the size (and the potential number of objects) was reasonable, the present Forsmark area was used to constrain the identification of objects. Thus, the smallest existing lake with potential to support future farmlands was used to define the lower size limit for a biosphere object also in the future landscape. No upper limit on object size was imposed. In some cases, when wetlands that never go through a lake stage were identified as potential discharge areas, the biosphere object was delimited based on future surface water/groundwater levels, as predicted by means of a hydrological model, and particle tracking information on discharges from repository depth [III–104].

The potential discharge of radionuclides with groundwater to a biosphere object that has the size of a lake, basin or a mire, for example, is unlikely to be homogenous, and groundwater flux rates and sediment/soil depths and properties will vary within the biosphere object. Areas with a high upward flux of groundwater are likely to receive more radionuclides than other areas, but the dilution will probably be greater in high flux areas. Thus, environmental concentrations can be expected to vary spatially within any outlined biosphere objects. Also, mixing from cultivation practices and processing of crops from the drained peatland will add to the dilution.

Depending on the modelled groundwater discharge locations within the ecosystem and the size of the area, a subdivision of the biosphere object into one or several smaller areas may be needed. This could also be the case if the identified area contains sub-areas with different properties for land use. Further arguments and rationales for this can be found in Ref. [III–111].

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GLOSSARY

The IAEA Safety and Security Glossary is provided in Ref. [114]. Additional terms used in this publication are defined below.

aeolian. Pertaining to the wind. Used in relation to erosion originating from air movements, for example by the suspension of sediment particles and their abrasive effects on rock formations.

calculation case. A specific assessment calculation used to determine impacts on human health and/or the environment. Several calculation cases can be used to study different aspects of a single scenario. Individual calculation cases can be deterministic or they can be probabilistic and involve multiple realizations of the model adopted.

catchment. Used as a short form of surface water catchment. The area over which precipitation is captured by a single stream or river. Where rivers are fed by tributaries, subcatchments may be defined for each tributary, with the overall catchment of the river comprising the subcatchments of the tributaries plus any areas that drain directly to the river.

contaminant. Any substance that impairs the purity of another substance or of an environment. Here used to include both radioactive substances and chemicals. A pollutant is a contaminant that has unwanted or adverse effects.

discretization. The representation of a continuous system as a finite number of elements. Here used in respect of mathematical models comprising sets of ordinary differential equations that are represented as meshes of cells (that may be one, two or three dimensional). Discretization permits numerical methods to be used to solve the equations of the continuous system under specified initial and boundary conditions. In general, the finer the discretization, the more accurately will the equations be solved.

drainage network. The system of surface water bodies, primarily streams and rivers, that transport meteoric waters away from an area, for example to a lake or the ocean.

eructation. Expelling gases from the gastrointestinal tract through the mouth.

eustatic. Related to or characterized by a worldwide change of sea level. Eustatic changes in sea level result from changes in the volume of ice sheets and ice caps, thermal expansion of sea water and changes in the shape of the ocean basins.

fluvial. Pertaining to the action of water. Typically used to describe landscapes formed through the action of flowing water.

ice cap. Mass of glacier ice smaller than an ice sheet. Ice caps could be topographically constrained, but this is not necessarily the case.

ice sheet. Large mass of glacier ice that is unconstrained by topography and has an area that exceeds ~50 000 km^2. At the present day, includes Greenland, the West Antarctic and East Antarctic ice sheets. However, in the past, large ice sheets have also existed over northern North America and Scandinavia.

isostatic. Relating to deformation of the crust of Earth by the emplacement or removal of mechanical loads, for example depression of the crust due to the growth of an ice sheet.

keystone species. A plant or animal that plays a unique and crucial role in the way in which a community or ecosystem functions, such that, without the keystone species, the community or ecosystem would be dramatically different or cease to exist.

leaching. Removal of a contaminant from waste either by solubilization and loss in solution from the waste matrix or by dissolution of the waste matrix liberating the contaminant.

non-human biota. All living organisms excluding humans. Approaches to environmental protection often distinguish protection of human health from protection of non-human biota.

percolation. This is the process by which a liquid is transmitted through a filter. However, in hydrology it is used in a more restricted sense to describe the process by which precipitation penetrates soils and sediments. The same process can be described as infiltration. Some distinguish between transport downwards across the soil surface, which they describe as infiltration, and the further downward movement of the water, which they describe as percolation.

till. Unsorted material deposited directly by glacial ice and showing no stratification.

LIST OF ABBREVIATIONS

AOGCM	Atmosphere-Ocean General Circulation Model
BIOMASS	IAEA Programme on Biosphere Modelling and Assessment, 1996–2001
BIOMOVS	Biospheric Model Validation Study, 1985–1990
BIOMOVS II	continuation of BIOMOVS, 1991–1996
DCRL	derived consideration reference level
DEM	digital elevation model
EFEPs	external features, events and processes
EMRAS	IAEA programme on Environmental Modelling for Radiation Safety, 2003–2007
FEPs	features, events and processes
GBI	geosphere and the biosphere
GMSL	global mean sea level
HLW	high level waste
ILW	intermediate level waste
ILW-LL	intermediate level waste — long lived
K_d	solid:liquid distribution coefficient (also termed sorption coefficient)
LDE	lead diagonal element (within an interaction matrix)
LET	linear energy transfer
LLW	low level waste
L&ILW	low and intermediate level waste
MODARIA	IAEA programme on Modelling and Data for Radiological Impact Assessments, 2012–2015
MODARIA II	continuation of MODARIA, 2016–2019
PEG	potentially exposed group
PEP	potentially exposed population
RAP	reference animals and plants
SKB	Swedish Nuclear Fuel and Waste Management Company
TBL	Two Boat Lake
USDOE	United States Department of Energy
WG6	Working Group 6 of the MODARIA II programme

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Interim Working Group Meetings

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