BIOPROTA

Key Issues in Biosphere Aspects of Assessment of the Long-term Impact of Contaminant Releases Associated with Radioactive Waste Management

Scales for Post-closure Assessment Scenarios (SPACE)

Addressing spatial and temporal scales for people and wildlife in long-term safety assessments

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PREFACE

BIOPROTA is an international collaboration forum that seeks to address key uncertainties in the assessment of radiation doses in the long term arising from release of radionuclides as a result of radioactive waste management practices. It is understood that there are radio-ecological and other data and information issues that are common to specific assessments required in many countries. The mutual support within a commonly focused project is intended to make more efficient use of skills and resources, and to provide a transparent and traceable basis for the choices of parameter values, as well as for the wider interpretation of information used in assessments. A list of sponsors of BIOPROTA and other information is available at www.bioprota.org.

The general objectives of BIOPROTA are to make available the best sources of information to justify modelling assumptions made within radiological assessments of radioactive waste management. Particular emphasis is to be placed on key data required for the assessment of long-lived radionuclide migration and accumulation in the biosphere, and the associated radiological impact, following discharge to the environment or release from solid waste disposal facilities. The programme of activities is driven by assessment needs identified from previous and on-going assessment projects. Where common needs are identified within different assessment projects in different countries, a common effort can be applied to finding solutions.

This report has been prepared as input to a project to investigate the appropriate spatial and temporal scales for various types of plants and animals in long-term safety assessments, in terms of population level impacts, and compared to spatial scales used for human assessments. In this project, the overall objective is to advance the understanding of temporal and spatial scales for wildlife populations and the commensurability of these with current approaches to human spatial and temporal averaging, specifically within the context of long-term safety assessments. The variety of plants and animals in the natural environment is immense and, as such, the scope of the project has necessarily been limited to the general types of plant and animal representative of temperate terrestrial ecosystems being selected as the focus for evaluation. It is intended that the lessons learnt can then be applied to assessments in other climate conditions and to alternative ecosystem types.

In this report, the rationale behind spatial scales of assessment in long-term safety assessments is examined, both in terms of dose and risk assessments for people and for non-human biota. Both the regulatory context and the available approaches are considered and issues associated with the incorporation of biota-specific scales of assessment in long-term safety assessments identified. A suggested approach for further investigating spatial and temporal scales of assessment for biota is presented and its application is demonstrated for a range of radionuclides that are likely to be released from radioactive waste disposal facilities.

The report provides information that may help to inform consideration of temporal and spatial scales for populations in future dose assessments for non-human biota in relation to radioactive waste disposal facility release scenarios. The content may not be taken to represent the official position of the organisations involved. All material is made available entirely at the user's risk.

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Version History

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Executive summary

This report presents the results of a study called SPACE, which was organised through the BIOPROTA Forum to address the issue of averaging scales within long-term non-human biota (NHB) dose assessments for radioactive waste disposal facilities. The overall objective has been to advance the understanding of temporal and spatial scales for populations of NHB and their commensurability with current approaches to human spatial and temporal averaging, specifically within the context of long-term safety assessments of releases of radionuclides to the biosphere from radioactive waste repositories. The need arises because averaging based on assessment of human exposure, typically used in past assessments, is not necessarily appropriate for the assessment of impacts on NHB.

A critical review has been made of international programmes and associated literature has allowed the rationale for addressing spatial and temporal scales within both human and NHB dose assessments to be evaluated. In addition to information on spatial and temporal scales from biosphere assessments that have recently been undertaken (e.g. in Sweden and Finland), the review included consideration of relevant activities within the context of the International Commission on Radiological Protection (ICRP) and the International Atomic Energy Agency (IAEA). The spatial extent of contamination relative to the area utilised by populations is a key area of consideration in the application of the ICRP framework for environmental protection. However, the review made suggests that there is a general lack of guidance on how best to incorporate scale considerations into long-term assessments for NHB. This is particularly notable at the population level. Given that the protection goal for most NHB assessments is protection at the population level, the lack of guidance in this area is surprising. The application of appropriate temporal and spatial scales in safety assessments would assist in communicating risks in terms of environmental protection objectives to stakeholders and mitigate against situations arising whereby unnecessary effort is expended on environmental protection that is incommensurate with the actual level of risk.

From an evaluation of long-term assessment approaches and critical review of life-history parameters for the range of 'SPACE representative species', it was concluded that, over the timescales for which long-term biosphere assessments are being undertaken, the temporal averaging resolution is unlikely to be significant when assessing doses to NHB. Therefore, although both spatial and temporal parameter data were reviewed for the selected 'SPACE representative species' and appropriate 'SPACE reference groups' established, only spatial averaging considerations were included in the modelling work undertaken to evaluate the influence of selected scales on assessment results. Within the modelling work, the commensurability of NHB and human spatial scales was evaluated using a typical averaging scale for humans, which reflects assumed human utilisation of agriculturally managed ecosystems and their resources. These in turn have been based on actual typical human behaviour today in those ecosystems.

The results presented for a range of different commonly relevant radionuclides with different radiation and other characteristics suggest that, in general, the human spatial averaging assumptions will provide conservative assessments of NHB doses to populations being considered within biosphere assessments. However, it is recognised that the scope of evaluation presented here has been limited. Whilst the results provide some confidence in the use of human averaging scales, they should only be considered provisional and the analysis could be further developed. For example, within future biosphere assessments, account could be taken of the land cover predictions and hence the degree of habitat fragmentation. Linking this with ecological data for each species, a more direct evaluation of the spatial extent of the populations within the assessment area could be undertaken.

For the purposes of the SPACE analysis, population scales were estimated for all of the representative organisms using a generic scaling value of 40. Whilst this was considered appropriate for the provisional analysis made in this study, and also has some provenance within ecological risk assessment, it is unrealistic to think that all populations will scale in the same way. A complex mix of environmental and ecological factors determines population scales. Therefore, whilst the SPACE study provides a useful indication of the influence of spatial scale assumptions within NHB dose assessments, there could be value in extending this evaluation to a range of real assessment situations in the future, and to consider variability in spatial scales under the different climate conditions that may arise over long-term assessment timeframes. To further develop the SPACE analysis, information on territoriality could be used, alongside predictions of landscape change and habitat cover, to evaluate the extent of functional connectivity of habitats of relevance to SPACE representative organisms and hence determine the potential spatial ranges of populations that may be expected to be present. Such work would most effectively be focussed and carried out on a site specific basis, rather than being based on generic considerations as in the current study.

International guidance recommends that human exposure groups should be characterised in terms relevant to the biosphere system that they live in, but no current similar recommendation is made for biota. With international and national legislation increasingly giving specific consideration to the protection of the environment from ionising radiation, there may be merit in giving further consideration to the utilisation of the biosphere by populations of plants and animals that may be exposed due to their possible occupancy in areas potentially affected by discharge zones concurrently to the consideration of human utilisation of the system. Consideration of the biosphere in terms of both people and biota at an early stage in assessments may help alleviate any concern that NHB assessments are undertaken as something of an afterthought to human dose assessments and ensure that model discretisation is appropriate to both human and biota dose evaluations.

CONTENTS

1. INTRODUCTION

Over recent years methodologies have been developed that enable assessments to be made of the potential impact of releases of radioactivity to the environment on species of animal and plant, commonly referred to as non-human biota (NHB). Continued interest in this field is attested by recent developments in international environmental protection requirements. This includes specific mention of environmental protection as a goal by the ICRP and the IAEA.

The ICRP specifically included environmental radiological protection objectives in their 2007 recommendations [ICRP, 2007]. These have been interpreted in the context of radiological protection requirements relating to geological disposal facilities for long-lived solid radioactive waste [ICRP, 2012] such that environmental radiological protection is specifically considered as an additional line of argument and reasoning in building a safety case. The inclusion of environmental radiological protection recommendations is intended as a means of broadening the basis for risk-informed decision making.

The IAEA in their International Basic Safety Standards [IAEA, 2014] also refer to environmental protection, acknowledging that:

"*In a global and long term perspective, protection of people* ***and the environment*** *against radiation risks associated with the operation of facilities and the conduct of activities … is important to achieving equitable and sustainable development*" (para 1.32).

Paragraph 1.33 continues by noting that the system of protection and safety required by the safety standards provides for the protection of the environment at an appropriate level from the harmful effects of radiation. It also acknowledges increasing international awareness of the vulnerability of the environment and the need to demonstrate protection of the environment irrespective of any human connection. In assessing environmental protection "*an integrated perspective has to be adopted to ensure the sustainability, now and in the future, of agriculture, forestry, fisheries and tourism, and of the use of natural resources*" (para. 1.34), including consideration of "*the potential for build-up and accumulation of long lived radionuclides released to the environment*" (para. 1.34).

In appreciation of the developing recommendations to specifically include radiological protection of the environment, a number of radioactive waste management organisations have considered the implications of waste disposal on the environment. For example, approaches to undertaking assessments for radioactive waste disposal facilities have been developing in Canada [Garisto et al., 2008] and dose assessments for wildlife were included in the two latest safety assessments performed by SKB; in the 2011 license submission for the repository for spent nuclear fuel [Torudd, 2010; Jaeschke et al., 2013] and in the 2014 license submission for an extension of the low and intermediate level waste repository [SKB, 2014a], as well as in the 2011 safety case for the low level waste repository in the UK [LLW Repository Ltd, 2011a]; the issue has also been addressed by Posiva within their 2012 license submission for the construction of the Olkiluoto repository in Finland [Posiva, 2014a]. A key driver for the inclusion of such assessments is the development of national policy to take account of international recommendations: for example specific requirements are in place both in Finland and in England and Wales to consider environmental radiological protection in relation to the disposal of radioactive waste [STUK, 2010; Environment Agency and Northern Ireland Environment Agency, 2009].

A working system has been developed that provides a means by which environmental protection may be demonstrated for planned exposure situations that includes tools allowing dose calculations to be made for generalised wildlife groups on the basis of measured or predicted activity concentrations in environmental media (soil, sediment, air, water). However, there are a number of uncertainties in the

application of these approaches to prospective assessments for radioactive waste disposal facilities. Many of the uncertainties stem from the long-term nature of the assessments for which specific advice and guidance on demonstrating environmental protection is lacking, the radionuclides of interest and their behaviour in the accessible environment. In recognition of this, the BIOPROTA collaborative forum[1] has undertaken a series of studies aimed at addressing some of these key uncertainties, including:

- identification of the sensitivity and knowledge quality associated with assessment models and parameters [Smith et al., 2010]; and,
- evaluation of approaches to demonstrating compliance with environmental protection objectives over the long-term, particularly in situations where screening values may be exceeded [Smith et al., 2012a; Jackson et al., 2014].

Whilst these studies have provided information in support of prospective long-term assessments, uncertainties nonetheless remain. In particular, limited consideration has been given to what would constitute appropriate spatial and temporal scales for NHB dose assessments, especially on the timescale appropriate to releases from radioactive waste repositories. Typically for such assessment contexts, scales of assessment applicable to human radiological assessments have been applied. However, there are significant differences in protection objectives for people and NHB. For people the focus is protection of representative persons (being representative of an appropriately defined potentially exposed group in the case of repository safety assessments), whereas for biota the aim is to protect biodiversity through protection of populations of relevant species [Smith et al., 2012a; Wood, 2011], with the noted exception of endangered species for which individual protection objectives may be applied [Copplestone et al., 2005]. The approach to determining environmental activity concentrations relevant for assessing dose to representative people will not necessarily, nor even likely, be the same as the appropriate approach for determining the radiological exposure of NHB populations, primarily due to differences in the target of protection and in the interactions of human and NHB receptors with contaminants in the environment. There are also issues relating to the inclusion of multiple NHB species within any one assessment and the variation in range of individuals between species; the area required to sustain a relevant population and the timescales over which assessment would be appropriate (in relation to organism longevity) will vary for different species. Implications of these differences to dose assessments are, as yet, unevaluated and may contribute significantly to assessment uncertainty. For example:

- use of a person-relevant spatial scale may fail to identify ‘hot spots’ such as discharge areas associated with springs or streams that may be relevant to wildlife population exposure;
- spatial scales may be optimised in terms of conservative dose assessments for representative people such that occupancy is maximised within individual biotopes (e.g. croplands, forests or mires), which may not be realistic in terms of sustaining wildlife populations. As a result, calculated dose rates for NHB may be unrealistic in relation to population scale impacts; and/or
- failure to take account of temporal scales of exposure may over- or under-estimate impact on wildlife because of the importance of exposure duration in relation to population dynamics and life-history.

[1] www.bioprota.org

Given the potentially significant contribution to assessment uncertainty, there is a need to quantify and evaluate the implications of temporal and spatial scale assumptions, both anthropocentric and ecocentric, when applied to long-term assessments of the impact of solid radioactive waste disposal on representative wildlife groups. Greater attention is therefore now needed to determine the relevant temporal and spatial scales appropriate for averaging radionuclide activity concentrations relevant to wildlife populations that are the focus for the environmental assessment calculations. Such focus will assist in demonstrating compliance with environmental protection objectives.

In order to address such assessment uncertainties, the project reported here was established within BIOPROTA to evaluate the appropriate spatial scales of assessment for plants and animals and the implications of the incorporation of these scales within long-term safety assessments in terms of exposure calculation and impact evaluation. The implications of exposure duration on populations are also considered.

1.1 PROJECT AIM, SCOPE AND OBJECTIVES

As discussed in Smith et al. [2012a], the focus of NHB dose evaluation is on populations. However, due to the wide range of plants and animals present in terrestrial and aquatic ecosystems globally, the scope of the present project was necessarily constrained. Therefore, this project focusses on terrestrial environments within a temperate climate as a means of evaluating scales of assessment within long-term safety assessments with species being selected in terms of their relevance to organisations participating in the project and their likelihood of exposure to radionuclides entering the surface environment from a subterranean source. Constraining the project in this way makes the evaluation of spatial scales manageable within the scope of the project and provides information that is directly relevant to many disposal facility scenarios. However, the general methodological advances and main lessons learned would be applicable to other species and environments.

Through the following specific objectives the SPACE project (Scales for Post-closure Assessment sCEnarios) aims to advance the understanding of temporal and spatial scales for wildlife populations and the commensurability of these with current approaches to human spatial and temporal averaging, specifically within the context of long-term safety assessments.

Specific objectives:

1. Critically review the rationale for addressing spatial and temporal scales within human and NHB dose assessments.
2. Identify relevant spatial and temporal scales for humans, recognising in particular the temporal variability in human utilisation of ecosystems and their resources.
3. Select ‘SPACE representative species’ for evaluation of assessment scales within long term assessments.
4. Develop a database of spatial and temporal parameters for SPACE representative species.
5. Identify relevant scales for populations of SPACE representative species and develop an approach for the development of ‘SPACE reference groups’.
6. Evaluate commensurability between human scales and the newly defined ‘SPACE reference groups’.

7. Identify modelling requirements, including site characterisation requirements, to address scale issues in post-closure NHB dose assessments.

8. Develop a strategy for undertaking 'reference group' assessments.

9. Evaluate the influence of assessment scales on the NHB dose predictions using a hypothetical release scenario from a radioactive waste disposal facility.

10. Present and disseminate findings in a format that supports the needs of those interested in the assessment of post-disposal impacts of radioactive waste disposal, and contributes to international thinking in this area.

1.2 REPORT STRUCTURE

Chapter 2 of this report presents the rationale for how scales of assessment may be applied in relation to human protection objectives within long-term safety assessments and Chapter 3 then outlines how the system for radiation protection of the environment has developed, application within long-term safety assessments and the rationale for considering spatial and temporal assessment scales. Approaches (applied and/or developing) for evaluating impacts on biota that are routinely applied or on-going are described in Chapter 4, with specific reference to their ability to consider spatial and temporal scales of assessments. This primarily focuses on work programmes of the IAEA (EMRAS I and II, MODARIA) and ICRP. Perceived barriers to the incorporation of NHB-specific scales of assessment within long-term safety assessments and key uncertainties are then presented in Chapter 5. Representative species and their associated assessment parameters are then identified to serve as the basis for evaluating temporal and spatial scales in Chapter 6 and their application within a test case scenario is presented in Chapter 7. Overall conclusions are presented in Chapter 8.

2. RATIONALE FOR SCALES OF ASSESSMENT APPLIED WITHIN DOSE ASSESSMENTS FOR PEOPLE

The background to selecting assessment scales for evaluating doses to people from present day releases of radioactivity to the environment, and for long-term prospective assessments for waste disposal facilities is discussed in this section. A stepwise methodology for long-term assessments is also briefly presented and examples of resultant assessment areas from the application of this methodology within assessments detailed.

2.1 PRESENT DAY RELEASES

Dose assessments for people require knowledge of, or assumptions for, the distribution of radionuclides in relevant environmental media in the area of interest, such as the breathable atmosphere, and the behaviour of people in that area which relates to how they interact with those media, giving rise to their radiation exposure. For radioactivity in a public environment to which the radionuclide releases have not yet occurred, i.e. for prospective assessments, guidance on human dose assessment is provided in a variety of internationally recognised documents, such as IAEA [2001] and the European Union (EU) sponsored report, Simmonds et al [1995]. These references apply to present day releases and address both the modelling of the distribution of radionuclides in the environment and the types of assumptions which need to be made concerning exposure groups. ICRP [2006] gives further guidance on the definition of exposure groups, described in terms of representative persons[2].

Environment Agency et al. [2012] provides an example of up to date national application of international guidance in this context. Referring, for justification, to both ICRP [2006] and ICRP [2007], Environment Agency et al. [2012] states that:

> *"Because it is not practicable to assess doses to each individual member of the public, the 'representative person' approach is used. The representative person is 'an individual receiving a dose that is representative of the more highly exposed individuals in the population'."*

Environment Agency et al. [2012] also noted that the Euratom Basic Safety Standards (BSSD) Directive [EC, 1996][3] requires doses to be assessed for reference groups of members of the public. Reference groups are defined as:

> *"a group comprising individuals whose exposure to a source is reasonably uniform and representative of that of the individuals in the population who are the more highly exposed to that source*".

Environment Agency et al. [2012] states that this definition of a reference group is broadly equivalent to that of a representative person and can be taken to be the same as the representative person. In line with ICRP [2006] and ICRP [2007], Environment Agency et al. [2012] also suggests that, when deciding upon the habits of the representative person, it is appropriate to consider that the representative person is representing a small group of the more highly exposed individuals in the

[2] Prior to ICRP [2006] the term average member of a critical group was commonly used to refer to the person whose dose should be compared with dose limits or constraints. In ICRP [2006], this is called the reference person.

[3] The Euratom BSSD 1996 was updated in 2013 [EC, 2013]. The update refers to an individual receiving a dose that is representative of the more highly exposed individuals in the population, excluding those individuals having extreme or rare habits;

population and that the dose to the representative person should be the average dose to this group. In 1985, the ICRP referred to this as the critical group [ICRP, 1985] and stated that this:

> "*group should be small enough to be relatively homogeneous with respect to age, diet and those aspects of behaviour that affect the doses received*".

Environment Agency et al. [2012] also noted that ICRP [1985] advised that the degree of homogeneity in this group depends on the magnitude of the mean dose in the group as a fraction of the relevant source upper bound (or dose constraint). In cases where the mean dose is less than about one tenth of the dose constraint, the group should be regarded as relatively homogeneous, if the distribution of individual doses lies substantially within a total range of a factor of ten (i.e. a factor of about three on either side of the mean). Where the mean dose of the group is more than one tenth of the dose constraint, the total range of doses to individuals in the group should be less than a factor of ten, preferably no more than a factor of three.

Further consideration of the issue of the size and/or homogeneity of the exposure group or population that is represented by the representative person was given in ICRP [2006] based on a probabilistic approach. According to this, the ICRP recommends that the representative person should be defined such that the probability that a person drawn at random from the population will receive a greater dose than the constraint is less than about 5% [ICRP 2006]. If such an assessment indicates that a few tens of people or more could receive doses above the relevant constraint, the characteristics of these people need to be explored. If, following further analysis, it is shown that doses to a few tens of people are indeed likely to exceed the relevant dose constraint, actions to modify the exposure should be considered.

The distinction between a representative person, the average member of a critical group, and a member of the reference group may appear esoteric, especially since they appear to be assessed on the same basis and all used for comparison with dose limits or constraints. However, these semantic variations may reflect the difficulty of precisely defining the level of caution and scale of spatial averaging which is appropriate when making the corresponding prospective assessments.

Environment Agency et al [2012] summarises the assessment steps for annual individual dose assessment as follows:

- **Identify / quantify source term -** The amount of each radionuclide released, its chemical form (if important) and the mode of release.
- **Model radionuclide transfer in the environment -** Estimate activity concentrations and dose-rates arising from the discharged radionuclides in environmental media such as air, water, sediment, soils and foods.
- **Determine exposure pathways -** Identify the relevant exposure pathways to people from the activity concentrations and dose-rates in environmental media.
- **Identify habits and data for exposure pathways** – Identify those habits and behaviours together with the associated habit data that could lead to exposure of people through all relevant pathways.
- **Determine candidates for the representative person from realistic combinations of habits** – A number of different groups of people should be determined for a particular source with their habits relating to the different exposure pathways. These groups of people could receive doses that are representative of the most highly exposed individuals in the population.

The determination process should be based on local knowledge and plausible assumptions. Candidates for the representative person expressed in terms of their habits can then be identified to represent each group.

- **Estimate doses to the candidates for the representative person –** Calculate doses for each group for all relevant exposure pathways. This should include identification of the most important exposure pathways and radionuclides in terms of their contribution to the overall dose.
- **Determine the representative person –** This is the candidate for the representative person expected to receive the highest mean dose.

For the purposes of the current report, it is noted that the selection of the candidate representative persons is only to be made within the context of an understanding of radionuclide transfer in the relevant environment, and that in turn, is dependent upon an understanding of the source term. It is noted that while the size of the relevant population, or the area in which they are supposed to live, is not specified, the discussion of homogeneity suggests that the group that the representative person represents should not show a variation of individual doses within it of more than about an order of magnitude. Combined with an understanding of the variation in food consumption and occupancy habits, such an approach puts a limit on the size of the group within any particular assessment context, and, effectively, the area over which concentrations are averaged in order to calculate the doses. In practice, this works against any attempt to increase the area simply in order to reduce the average dose within that area. While these recommendations serve to limit an overly-optimistic approach to assessment, EU BSSD [EC, 2013] requires at Article 66 the assessment of doses to an individual receiving a dose that is representative of the more highly exposed individuals in the population, excluding those individuals having extreme or rare habits. Clearly a balance is needed and, although factors affecting that balance can be discussed in general terms, it is hard to justify specific lines of reasoning or provide quantitative lines of reasoning except within the context of a particular assessment.

2.2 RELEASES IN THE FAR FUTURE

Dose assessments for people have to be made for the long time-frames associated with possible releases from radioactive waste repositories[4] while recognising that:

> *"Any description of the biosphere, including the behavior of humans within it, could appear somewhat arbitrary. A choice of assumptions has to be made as the basis for the assessment. Taken together however, these choices should be consistent with the aim of providing a robust yet reasonable level of assurance regarding the acceptability of possible future releases from a repository into the biosphere. Reference biospheres should provide a practical way of ensuring that an assessment is based on a good scientific appreciation of the key issues and a wide consensus as to what is robust yet reasonable."* [IAEA, 2003].

This view broadly corresponds to the suggestion of the ICRP that assessment biospheres should adopt a stylised approach based on general (human) habits and (biosphere) conditions [ICRP, 2000]. A similar view regarding biosphere uncertainties was included in a more recent report of the Nuclear Energy

[4] These typically extend thousands of years into the future. However, early release scenarios, such as those associated with human intrusion [Smith et al, 2012b] or other factors related to specific assessments, mean that doses may need to be assessed on shorter timeframes.

Agency [NEA, 2012], that refers to the limited possibility to forecast distant-future biospheres and human habits over the very long timescales considered in repository safety assessment.

The reference biosphere methodology set out in IAEA [2003] sought to provide a procedure for meeting the challenge of *providing a robust yet reasonable level of assurance* with respect to long-term dose assessment. The steps in the procedure are illustrated in Figure 2-1. It is noted that the step involving consideration of the potential exposure groups (PEGs) is a logical sequence similar to that adopted for present day releases as presented above, but recognises that more features of the system are likely to be hypothetical, or presented as examples for illustration rather than absolute predictions of impact, and that iteration is likely to be needed in all but the simplest of cases.

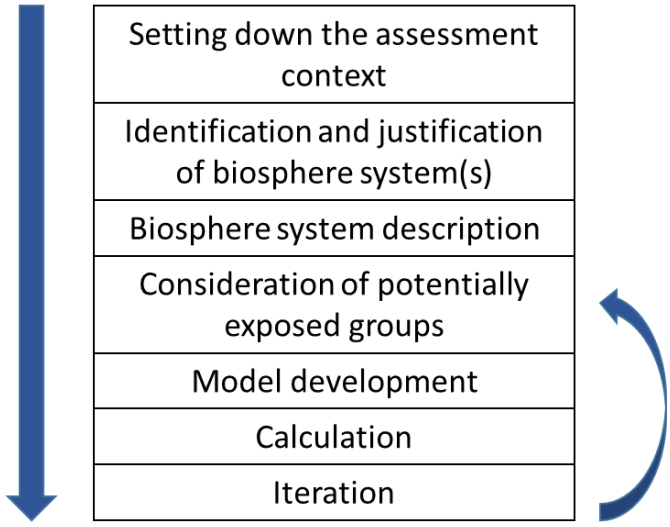


Figure 2-1. ***Steps in Reference Biospheres Methodology [IAEA, 2003].***

Defining the **assessment context** is the first stage and involves the setting down of some basic assumptions about the assessment, needed because the assessments (generally) involve such long time-frames. They include definition of the overall requirements, principally the purpose of the assessment; the assessment endpoint(s); the site and repository context; the radionuclide source term; the geosphere-biosphere interface; the assessment timeframe; basic assumptions about society; and the assessment philosophy (e.g. the level of conservatism to be applied).

Biosphere system **identification and justification** is the second stage of the methodology. Its purpose is to build on the assessment context to identify and justify the biosphere system(s) that is/are to be modelled. Identification and justification takes place in three main steps

1. Identification of the typology of the main components of the biosphere system (e.g. climate type, geographical extent and topography, human activities etc.) using a series of tables.

2. A decision on whether or not the assessment context requires biosphere change to be represented. In deciding this, two components of the assessment context are particularly relevant: the timeframe of the assessment and the geosphere-biosphere interface (GBI). At a coastal site, for example, it may be considered necessary to consider the effect of changes in sea level.

3. If biosphere change is to be represented, the third step considers how this should be done. One might, for example, simulate the consequences of radionuclides emerging into a set of

separate, unchanging biospheres, chosen to encompass the range of possible futures of interest. Additionally or alternatively, one might wish to consider an inter-related time sequence of biospheres with the interest focussed on the changes from one system to another.

A wide range of illustrations of how these steps can be implemented as far as quantitative dose assessments is provided in [IAEA, 2012a]. Further consideration was given to different types of GBI and important processes controlling the release of radionuclides from the geosphere into the biosphere in a recent BIOPROTA project [BIOPROTA, 2014]. It is noted here that the GBI can be very important in the current context since, for many scenarios for release of radionuclides to the biosphere, it controls the size of the area into which the main radionuclide release occurs, hence defining where radionuclide concentrations are likely to be the highest and hence where doses could be highest.

The next stage is to construct a biosphere system description. This should provide enough detail about the biosphere system (or systems) to be considered in the assessment to justify the selection and use of conceptual models for radionuclide transfer and exposure pathways. To begin, a decision has to be made regarding the assumed level of human interaction with the biosphere system, for instance foraging in a natural or semi-natural environment compared to intensive agriculture. Further illustration is provided in the description of the treatment of future human actions within the safety assessment for the SFR-PSU [Andersson et al, 2014]. Then, for each identified system component, lists of potentially important features, events and processes (FEPs) are screened to determine a short-list of those thought to be relevant to the assessment. Working systematically through these lists allows the main features of the biosphere system to be described, alongside the reasons for the various choices. For example, consideration of the socio-economic context of the local human community provides a basis for the subsequent identification of potentially exposed groups for which radiological exposures are to be considered within the assessment model.

IAEA [2003] illustrated the application of the reference biosphere methodology by reference to two major types of GBI involving radionuclide release in groundwater. One involved abstraction of contaminated groundwater from a well and its domestic and agricultural use, and the other assuming natural groundwater flow to the surface, discharging into different types of soil and water bodies in a hypothetical but realistic landscape. These examples were however limited to assumed constant biosphere conditions.

During the preparation, and since the publication, of IAEA [2003] many post-closure assessments of radioactive waste repository were or have been made on regional or on site specific bases, which have taken account of the above material. Detailed implementation of the methodology can vary significantly due to the different possibilities to be considered according to the different assessment contexts. For example, the regulatory requirements (US Code of Federal Regulations, 40 CFR 197.20) which are applied to the assessment of spent fuel disposal at Yucca Mountain pre-define many features of the GBI and the assumptions for human behaviour. The more prescriptive examples naturally present scope for divergence from each other, and these in turn may reflect geographic and other locally specific factors. This is reflected in the comment in NEA [2012], that,

> *"Greater differences exist between countries regarding the extent to which regulations allow simplified handling of the biosphere in the safety assessment"*,

that is, compared with other aspects of the overall repository assessment.

Many of these assessments have included scenarios corresponding to those considered in IAEA [2003], but taking account of specific information about sites and other factors relevant to the assessment, such as the need to take account of environmental change. See IAEA [2012a] for examples.

Also since IAEA [2003] was released, guidance on the safety case and safety assessment for the disposal of radioactive waste has been published [IAEA, 2012b]. Relevant paragraphs here are as follows:

> *5.29. Normally, it is assumed that the representative person is located within the region of potential radionuclide contamination in the accessible biosphere giving rise to the highest radiological impact. It may also be assumed that radioactive contamination of the biosphere due to releases of radioactive material from the disposal facility is likely to remain relatively constant over periods that are considerably longer than the human lifespan. It is then reasonable to calculate the annual dose or risk by averaging over the lifetime of the individuals.*
>
> *5.30. In {ICRP [2006]} it is recommended that three age categories be used for estimating the annual dose to the representative person for prospective assessments. These categories are 0–5 years (infant), 6–15 years (child) and 16–70 years (adult). For practical implementation of this recommendation, dose coefficients and data on habits for a 1 year old infant, a 10 year old child and an adult should be used to represent the three age categories.*
>
> *5.31. For long term dose assessments, it can be assumed that radioactive contamination of the biosphere due to releases of radioactive material from the disposal facility is likely to remain relatively constant over periods that are considerably longer than the human lifespan. It is then reasonable to calculate the annual dose or risk by averaging over the lifetime of the individuals, which means that it is not necessary to calculate doses to different age groups; the average annual dose can be adequately represented by the annual dose or risk to an adult.*[5]
>
> *5.32. It should be ensured that the characteristics assumed for the individuals in the group are consistent with the capability of the biosphere to support such a group. For example, depending on the assumed environmental conditions (location, climate, etc.), the agricultural capacity or other productivity of a particular setting may limit the size of the group that can reasonably be expected to be present.*

Based on the above considerations the following points are made about the rationale for long-term dose assessment for people for releases from repositories.

- Selection of assumptions for the human exposure groups needs to be considered alongside other features of the assessment within an iterative process.
- The selection of values of key parameters for biosphere compartments is fundamentally dependent on processes in the geosphere-biosphere interface which control the release from the geosphere, and which in turn depend on the results of modelling of the geosphere. The system should be assessed in an integrated fashion, to take account of these connections.
- Regulatory requirements also have had a significant impact on the selection of parameter values, especially as regards what they say about:
    - timeframe for assessment;
    - the need or otherwise to address environmental change; and

[5] Practically the same point was made in ICRP's most recent recommendations on radiological protection in geological disposal of long-lived solid radioactive waste [ICRP, 2013].

- the definition of endpoints for human dose assessment, such as:
  - individual dose or risk to critical or representative groups,
  - doses to other human exposure groups,
  - the nature of critical and/or representative exposure groups, including the level of prescription regarding exposure groups and assumptions for society and land use.

- International recommendations suggest that human exposure groups should be characterised in terms relevant to the biosphere system that they live in. It is also noted that the biosphere may change significantly with time due to natural or anthropogenic causes. Thus the assumption for human behaviour may go beyond the behaviour of the potential exposure groups.

- For site generic assessments, the assumptions for the local biosphere compartments in which peak radionuclide concentrations would occur, have been selected on a theoretical and/or regional basis. They should be scientifically and physically coherent and it can be helpful to apply regional models to understand the system in enough detail so as to be able to select parameter values coherently.

- For site-specific assessments, local site characterisation work has been used to support biosphere compartment discretisation (the size of compartments), as well as values of parameters related to processes that support radionuclide migration between those compartments. However, the degree to which site characterisation has been done varies from project to project, particularly because of different levels of prescription in regulatory requirements and guidance on how to conduct the assessment.

- One approach is to limit the model for radionuclide migration to cover only the area within which the discharge locations to the surface environment are expected to occur, and to the areas potentially contaminated by radionuclide transport away from those discharge locations. However, the characteristics of the biosphere in such an area can be affected by the wider environment, especially if environmental change has to be considered.

- Accordingly, it has been shown that connections between biosphere objects can be derived from models of the wider terrain. These connections determine primarily how the biosphere objects, represented by compartments or collections of compartments, are hydrologically connected, i.e. they determine if one object is upstream or downstream of another, either because water is the vector of dissolved material or the cause for erosion of solid material.

- Catchment-scale and coupled geochemical/flow models can be framed by regional understanding of geology, hydrogeology and hydrogeochemistry. These, in combination with similar modelling deeper within the geosphere, then provide the context and boundary conditions of the model of the local catchment, where release from the geosphere and exposure might occur.

- In some cases, this regional scale biosphere system modelling has taken account of the dynamics of environmental change of the landscape. These changes are said to be largely driven by climate change, which can have many effects on the biosphere, but may especially affect sites located on the coast by changes in sea-level and coastal erosion. The models used for dose assessment are derived significantly from the landscape models and include varying

degrees of dynamic features. They are largely driven by local landscape features rather than any general rules. The dose assessment model can be relatively simple, as in the case for site generic assessments, but the justification for the simplification comes from a large range of site characterisation measurements and interpretation using a complex range of models.

- There are significant differences in the types of biosphere system that have been assessed, and in the methods used to support selection of parameter values for compartments. Despite these differences, it is suggested that the key parameters and processes are:
    - volumetric flow in surface water bodies, in the case of groundwater discharge to surface waters;
    - area of soil contaminated by irrigation water or by release from the geosphere, upward into soil from below; and
    - processes for loss from soil, especially water flow though soil and, to a lesser degree, erosion of soil.
- The area assumed can vary according to the assumed characteristics of the exposure group and the productivity of the soil etc., but it is noted that example assumptions for areas and critical groups include:
    - A range from 1E5 to 3E6 m^2 suggested in work by Nagra based on consideration of a range of specific sites [NAGRA, 2013];
    - A crop area for a critical group of 1E4 m^2 in work carried by CIEMAT for ENRESA [Perez-Sanchez, 2013];
    - A range of areas for different types of exposure groups ranging from 500 m^2 for a group whose exposure is linked only to growing of vegetables, up to 1.2E5 m^2 in the case where a wide range of animal products and other sources of exposure are considered [LLW Repository Ltd, 2011b]; and
    - 50 people needing 2E4 m^2 in JNC [2000].
    - The approach in the site specific dose assessments performed by SKB is somewhat different as the contaminated area is delineated as “biosphere objects” based on aspects such as topography and groundwater discharge areas (for details see Chapter 6 in [SKB, 2014b]). The size of the most exposed group is then identified based on the carrying capacity of each object varying with time and depending on the land use considered. The size of the contaminated areas varies over time due to land uplift and e.g. lake ingrowth. In the safety assessment SR-PSU [SKB, 2014a] the smallest biosphere object area used was 1E5 m^2
- The area contaminated by irrigation water is usually selected to be consistent with the need to produce enough food to meet the dietary needs of the exposure group within which the representative person resides (or the critical group or the reference group). This approach was suggested in IAEA [2003] and has been adopted in various assessments. However, if the well abstraction rate is too small to irrigate this area, or the area contaminated from below via natural release via groundwater from the geosphere is smaller than this area, then the occupancy and contamination in food consumed by the critical group should be reduced in proportion.

3. RATIONALE FOR SCALES OF ASSESSMENT APPLIED WITHIN DOSE ASSESSMENTS FOR PLANTS AND ANIMALS

In contrast to human radiological protection, procedures for demonstrating compliance in terms of protection of the environment from ionising radiation entering the biosphere from radioactive waste facilities are less well developed. Indeed, with a framework for environmental protection from ionising radiation having been developed relatively recently, there is relatively little practical assessment experience as compared with human dose assessments, even for current release situations. Dosimetric assessment tools and supporting databases have nonetheless been developed that allow dose rates to biota to be evaluated, and regulatory recommendations for protection objectives and mechanisms by which compliance may be demonstrated are available [ICRP, 2008]. However, little consideration has, as yet, been given as to appropriate averaging approaches for radioactivity in environmental media in relation to these protection objectives. This section provides background on the development of the current environmental protection framework, the current status of guidance on its application to long-term dose assessments for plants and animals, and approaches taken to date in repository safety assessments.

3.1 BACKGROUND TO THE FRAMEWORK FOR ENVIRONMENTAL PROTECTION FROM IONISING RADIATION

Traditionally, the system of radiological protection has focused on the protection of people, in line with recommendations of the ICRP which, until recently, did not deal explicitly with environmental protection, but rather applied the rationale that if humans are protected then so too are other species within the environment they inhabit. However, largely in response to various national initiatives on the subject of protection of the environment from radiation, the Commission set up a task group in 2000 to address the issue of environmental protection and, in 2003 Publication 91 [ICRP, 2003] was published. This addressed the ethical basis for environmental protection and recommended the development of a flexible framework for radiation protection of the environment. The task group concluded that that any developments in terms of an assessment framework for non-human species should draw upon the lessons learned from the development of the systematic framework for the protection of humans.

In response to the recommendations of the task group, ICRP established Committee 5 in 2005. Committee 5 had the specific objective of taking forward the recommendations in ICRP [2003] in developing a framework for the assessment of radiation exposure and effects on non-human species that would be applicable to planned, existing and emergency response situations and consistent with the framework for humans.

Subsequently, in 2007, updated recommendations on the system for radiological protection were published, broadening the scope of earlier recommendations to directly address the subject of protection of the environment [ICRP, 2007]. In the 2007 recommendations, the Commission concluded that there was a need for a systematic approach for the radiological assessment of non-human species to support the management of radiation effects in the environment in order to address a conceptual gap in the radiological protection system. Whilst acknowledging that there is no simple or single universal definition of ‘environmental protection’, and noting that the approach to environmental protection should be commensurate with the overall level of risk, the Commission set out general aims for environmental protection [ICRP, 2007]:

> *“to prevent or reduce the frequency of deleterious radiation effects in the environment to a level where they would have a negligible impact on the maintenance of biological diversity, the*

conservation of species, or the health and status of natural habitats, communities, and ecosystems."

The following year, the first report of Committee 5 was published that set out an assessment framework for environmental protection, based around the concept and use of reference animals and plants (RAPs) [ICRP, 2008]. This framework is described further in Section 4.1.

The IAEA has also considered the issue of environmental protection from ionising radiation and, in 2002, published a report detailing the ethical basis for protection of the environment and outlining a series of protection goals [IAEA, 2002]:

- Any radiation exposure should not affect the capability of the environment to support present and future generations of humans and biota (principle of sustainability);
- Any radiation exposure should not have any deleterious effect on any species, habitat, or geographic feature that is endangered or is under ecological stress or is deemed to be of particular societal value (principle of conservation);
- Any radiation exposure should not affect the maintenance of diversity within each species, amongst different species, and amongst different types of habitats and ecosystems (principle of maintaining biodiversity);
- The management of any source of radiation exposure of the environment should aim to achieve an equitable distribution of the benefits from the source of the radiation exposure and harm to the environment resulting from the radiation exposure, or to compensate for any inequitable damage (principle of environmental justice); and
- In decisions on the acceptability and appropriate management of any source of radiation exposure of the environment, the different ethical and cultural views held by those humans affected by decisions should be taken into account (principle of respect for human dignity).

Subsequently, in line with ICRP developments in the field of environmental protection and taking account of national experience in countries with environmental legislation and methodologies in place, the IAEA reviewed the need for revised safety standards, that would also take due regard of Principle 7 of the IAEA Safety Fundamentals [IAEA, 2006] that "*people **and the environment**, present and future, must be protected against radiation risks*". Revised International Basic Safety Standards were published in 2014 [IAEA, 2014] and addressed the IAEA's fundamental safety objective "*to protect people **and the environment** from harmful effects of ionizing radiation*". Relevant paragraphs to protection of the environment are as follows:

1.32. In a global and long term perspective, protection of people and the environment against radiation risks associated with the operation of facilities and the conduct of activities — and in particular, protection against such risks that may transcend national borders and may persist for long periods of time — is important to achieving equitable and sustainable development.

1.33. The system of protection and safety required by these Standards generally provides for appropriate protection of the environment from harmful effects of radiation. Nevertheless, international trends in this field show an increasing awareness of the vulnerability of the environment. Trends also indicate the need to be able to demonstrate (rather than to assume) that the environment is protected against effects of industrial pollutants, including radionuclides, in a wider range of environmental situations, irrespective of any human connection. This is usually accomplished by means of a prospective environmental assessment to identify impacts

on the environment, to define the appropriate criteria for protection of the environment, to assess the impacts and to compare the expected results of the available options for protection. Methods and criteria for such assessments are being developed and will continue to evolve.

1.34. Radiological impacts in a particular environment constitute only one type of impact and, in most cases, may not be the dominant impact of a particular facility or activity. Furthermore, the assessment of impacts on the environment needs to be viewed in an integrated manner with other features of the system of protection and safety to establish the requirements applicable to a particular source. Since there are complex interrelations, the approach to the protection of people and the environment is not limited to the prevention of radiological effects on humans and on other species. When establishing regulations, an integrated perspective has to be adopted to ensure the sustainability, now and in the future, of agriculture, forestry, fisheries and tourism, and of the use of natural resources. Such an integrated perspective also has to take into account the need to prevent unauthorized acts with potential consequences for and via the environment, including, for example, the illicit dumping of radioactive material and the abandonment of radiation sources. Consideration also needs to be given to the potential for buildup and accumulation of long lived radionuclides released to the environment.

1.35. 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.

As noted (paragraph 1.34), methods and criteria for assessments continue to evolve and have benefited from IAEA co-ordinated programmes aimed at model development and application (see Appendix A).

Together, ICRP and IAEA thinking in relation to environmental protection has driven further the evolution of assessment approaches and the incorporation of environmental protection requirements within national regulatory regimes, both in terms of existing and planned exposure situations.

3.2 PROTECTION OF THE ENVIRONMENT IN THE CONTEXT OF LONG-TERM ASSESSMENTS

International guidance on the development of safety cases and underpinning safety assessments for the disposal of radioactive waste states that the fundamental safety objective is to protect people and the environment from harmful effects of ionising radiation [IAEA, 2012b]. The key focus of a safety assessment is therefore to evaluate the performance of a disposal system and quantify its potential radiological impact on human health and the environment and to provide assurance to the regulatory body and other interested parties that safety requirements will be met.

Paragraph 4.22 of IAEA [2012b] states that the safety principles adopted should give particular reference to Principle 7 of the IAEA Safety Fundamentals on protection of present and future generations. In this regard, it is stated that regulatory criteria established by a regulatory body should, as a minimum "*address radiation dose and risk constraints for workers and the public (both present and future generations), and protection of the environment*". Whilst protection of the environment is specifically detailed as requiring consideration, no guidance on its incorporation within a safety assessment is provided on the basis that "*an international consensus on approaches and criteria for addressing this issue is still evolving*".

As discussed in Section 2.2, a reference biosphere methodology has been developed [IAEA, 2003] that provides a procedure for meeting the challenge of providing a robust yet reasonable level of assurance with respect to the acceptability of possible future releases from a repository into the biosphere. This

methodology is routinely applied as the basis for undertaking safety assessments for the post-closure phase in support of repository safety cases. However it should be noted that the methodology was developed in the years preceding the incorporation of environmental protection objectives in ICRP's 2007 recommendations [ICRP, 2007] and in the IAEA's 2011 International Basic Safety Standards [IAEA, 2014]. As such, procedures for demonstrating compliance with environmental protection objectives in this context were not specifically considered. Nonetheless, many of the procedures outlined in IAEA [2003] are directly applicable in terms of assessing impacts on plants and animals following possible future releases from a repository into the biosphere. However, it may be appropriate to consider environmental protection endpoints in their own right to ensure appropriate discretisation of the biosphere.

3.2.1 Protection endpoints

Smith et al. [2012a] discussed protection endpoints in relation to protection objectives for long-term assessments. A number of environmental protection goals have been set both nationally and internationally and, whilst differences are evident, a number of recurring themes arise, including protection of rare or endangered species or the protection of communities of organisms or biodiversity. Whilst these protection objectives may pose limited issues for the majority of assessments whereby species or communities of organisms that may be impacted can be identified and risks evaluated, Smith et al. [2012a] noted that difficulties arise in relation to identifying targets that would meet such protection objectives under the timeframes relevant to post-closure safety assessments. A problem with long-term safety assessments is that environmental changes may occur, unrelated to the source of exposure, which dominate over any population changes. Smith et al. [2012a] therefore considered approaches that could be taken to enable various protection objectives to be considered within a safety assessment framework, concluding that, in practice, protection of populations provides the most accessible target. It was however recognised that any assessment of populations must be based on dose-effect relationships expressed at the individual level and therefore care in identifying 'typical' exposures experienced across a population is required. A population endpoint thus necessitates consideration of the spatial scale over which radioactivity is present in relation to the area occupied by that population.

3.2.2 Environmental protection approaches in repository safety assessments

In terms of long-term safety assessments for radioactive waste disposal facilities, assessments of the radiological impact of radioactive releases to the biosphere on plants and animals are increasingly common. However, approaches adopted to demonstrate compliance and the degree to which spatial and temporal scales of assessment have been considered are variable.

In one of the earliest assessments made in the current context [Punt et al., 2003] environmental concentrations in environmental media were determined on spatial scales relevant to human dose assessment, not on the scales relevant to endpoints for environmental protection assessment. No consideration was given to the area required to support populations. More recently, a conservative screening assessment was undertaken to demonstrate that activity concentrations were below regulatory criteria for all possible release scenarios for the Swedish repository for spent nuclear fuel license submission [Torudd, 2010; Jaeschke et al. 2013]. The approach taken was to apply maximum environmental activity concentrations, irrespective of the location at which they occur or the period in which they are released, as the basis for conducting the assessment. In their 2011 safety case assessment, LLW Repository Ltd [2011a] focussed their assessment on individuals with the argument that, if protection of individuals of a species could be demonstrated then by inference, the higher-order systems within which they have a role would also be adequately protected. Activity concentrations applied in biota assessments were those established for the calculation of dose to people and dose rate

to individual plants and animals occupying terrestrial, aquatic (freshwater and marine) and transitional (intertidal) biotopes evaluated. Whilst individuals were the focus of assessment, some consideration was given to the likely impacts of individual effects on populations by reasoned argument. Posiva, however, in their 2012 license submission for the Finnish repository at Olkiluoto [Posiva, 2014a], evaluated radioactivity in environmental media throughout the assessment area through time with activity concentrations being averaged across the different biotopes that could be inhabited by an assessment species. Time-series typical dose rates were therefore calculated rather that dose rate maxima. Spatial scales were considered to the extent that individual species were evaluated on the proportion of time spent in different biotopes. The overall area required to sustain a population was not, however, evaluated.

Irrespective of the approach employed, it is apparent that biosphere discretisation within assessment models is primarily driven by assumptions of the area required to sustain a defined human population (see section 2.2) or through site characterisation activities, whereby distinct biosphere objects are defined taking into account the assessed locations for release of contaminants into the biosphere from a repository. The former approach was the basis for the assessment approach described in Punt et al [2003] with the latter being demonstrated in both SKB [Torudd, 2010; Jaeschke et al. 2013] and Posiva [2014a] approaches. How such areas relate to the areas utilised by distinct populations of plants or animals has not, so far, been specifically addressed. Difficulties may therefore arise in relating, conceptually, the assessment endpoint to the protection objectives.

The application of appropriate temporal and spatial scales in safety assessments would assist in communicating risks in terms of environmental protection objectives to stakeholders and mitigate against situations arising whereby unnecessary effort is expended on environmental protection that is incommensurate with the actual level of risk. The spatial extent of contamination relative to the area utilised by populations is also a key area of consideration in the application of the ICRP framework for environmental protection [ICRP, 2008].

4. PERSPECTIVES FROM INTERNATIONAL PROGRAMMES ON SPATIAL AND TEMPORAL SCALES OF ASSESSMENT

This chapter provides an overview of ICRP, IAEA and other work in the field of NHB assessments and considers the degree to which spatial and temporal scales of assessment have been considered in these programmes to date.

4.1 ICRP ENVIRONMENTAL PROTECTION FRAMEWORK

As noted previously, ICRP [2008] describes a framework for protection of the environment that is broadly consistent with the approach for human protection. The framework is based around the assessment of dose rates to a set of reference animals and plants (RAPs): however, and in line with the evolution of the system of protection for people over time, it has been acknowledged that the framework for protection of the environment will take time to fully embed and will require revision as new information becomes available [ICRP, 2008; Pentreath, 2012]. Notwithstanding that, ICRP [2008] provides a mechanism by which dose rates to the RAPs can be evaluated and provides pointers to key areas of consideration when evaluating the potential for environmental harm from ionising radiation in the environment.

One such consideration is that the assessment framework necessarily requires dose rate calculations to be performed at the level of the individual, both in terms of dosimetric calculations and evaluation of effects, for which data have largely been derived for individuals rather than higher levels of organisation. Whilst assessing dose rates and effects for individuals can ultimately be considered to offer protection to populations, it is acknowledged that effects in individuals may be of little, if any, significance in an ecological context unless a proportion of individuals in a population are affected: an effect in one individual does not necessarily imply effects at the level of the population. Nonetheless, since population effects are mediated via effects on individuals, the development of the assessment framework was focussed around radiation effects on the individual with the intention that the implications of individual effects to higher levels of biological organisation be considered in assessments, consistent with relevant protection objectives. Indeed, it is specifically noted in ICRP [2008] that an assessment should necessarily consider the different fractions of a population exposed to different dose rates [ICRP, 2008].

In order to support the consideration of population effects in assessments, population characteristics for each of the ICRP RAPs are provided. This information is reproduced in Table 4-1 and includes data relating to sustainable population sizes for each RAP and their reproductive strategies. The need to consider the geographic area required to support populations of these sizes is also noted although data specifically relating population size to geographical areas are not provided. Information on the proportion of a population receiving effects that would constitute impact in terms of population sustainability is also lacking. In the case of the former, this may, in part, result from the fact that the RAPs are representative of families of organisms for which large variations in spatial ranges may be evident. The temporal scale of exposure is also noted as a relevant consideration for assessments. Timescale considerations are relevant both to the exposure of different life stages of a species, but also the length of time over which continued exposure could manifest in, for example, reproductive impacts affecting population dynamics.

Table 4-1. General population characteristics for ICRP RAPs (reproduced from Table 2.1 in ICRP [2008]).

Reference Animal / Plant	Population characteristics
Deer	Iteroparous, distinct cohorts, high female:male ratio, low fecundity, population number <500
Rat	Iteroparous, equal gender ratio, high fecundity, population number <1000
Duck	Iteroparous, distinct cohorts, equal gender ratio, low fecundity, population number <500
Frog	Iteroparous, distinct cohorts, equal gender ratio, high fecundity, population number <500
Trout	Iteroparous, distinct cohorts, equal gender ratio, high fecundity, population number <500
Flatfish	Iteroparous, distinct cohorts, equal gender ratio, high fecundity, population number >10,000
Bee	Semelparous (for males), high male:female ratio, high fecundity, population number <10,000
Crab	Iteroparous, distinct cohorts, equal gender ratio, high fecundity, population number >500
Earthworm	Iteroparous, hermaphrodite, high fecundity, population number >10,000
Pine tree	Iteroparous, canopy forming, high fecundity, population size >1000
Wild grass	Iteroparous, high fecundity, perennial with yearly regrowth, population size >1000
Brown seaweed	Iteroparous, low recruitment to adult population, population size >1000

4.1.1 Protection of the environment under planned exposure situations

Since the publication of ICPR [2008], guidance on the protection of the environment under different exposure situations has been developed [ICRP, 2014]. The approach involves the use of dose rate bands first described in ICRP [2008], each spanning an order of magnitude, at which different radiation effects may be observed for each of the 12 RAPs.

For planned exposure situations, which encompasses future disposal of radioactive waste, it is suggested that the use of the lower banding of what is termed the Derived Consideration Reference Level (DCRL) may be appropriate as a consideration level whereas the DCRL band itself may be appropriate for existing exposure situations. In this respect it is noteworthy that application of the lower dose band may result in more stringent assessment criteria that the ERICA / PROTECT value of 10 µGy/h that is widely applied throughout Europe as a screening value, although similar values were derived by PROTECT for vertebrates using the species sensitivity distribution (SSD) statistical approach for the derivation of screening criteria [Andersson et al., 2009].

Whilst the outlined approach considers effects in terms of dose bands under different exposure situations, no reference is made as to the need to consider the spatial or temporal scales of assessment at which the dose band ‘triggers’ should be considered in terms of regulatory compliance demonstration, but this is something that ICRP is known to be considering [Copplestone, personal communication]. Such considerations are necessarily complex due to the differences in behaviour of animals in different environmental situations. Nonetheless some reference to the need to consider such scales of assessment would be beneficial to avoid management decisions being applied to individual effects assessments rather than wider population considerations. In this respect, Copplestone [2012] notes that protection at the level of the individual should provide a level of protection adequate for even the most sensitive species, but that such a defined and strict level of assessment may be prohibitive in

terms of cost and may be difficult to evaluate for individuals in different environments. Assessments should therefore take into account factors such as the area over which exposure occurs and the time spent in different parts of the exposed area by different species [Copplestone, 2012].

4.2 IAEA PROGRAMMES

Since 1988, the IAEA have organised a series of co-ordinated programmes on the development and application of models for use in environmental assessments [IAEA, 2012a]. These programmes have included VAMP (Validation of Model Predictions, 1988-1996); BIOMASS (BIOsphere Modelling and ASSessment, 1996-2001); EMRAS (Environmental Modelling for Radiation Safety, 2003-2007); EMRAS II (2009-2011); and the current MODARIA (Modelling and Data for Radiological Impact Assessments) programme, which commenced in November 2012. In parallel with the increasing international consideration of the impacts of ionising radiation on wildlife, EMRAS and subsequent programmes have included specific Working Groups focussing on NHB modelling.

Extensive testing and intercomparison of models has been undertaken within the IAEA EMRAS and EMRAS II programmes, utilising two terrestrial scenarios (Chernobyl & Little Forest Burial Ground), two freshwater scenarios (Perch Lake & Beaverlodge Lake) and a wetland scenario (see Appendix A). This model testing has included models which are under on-going development by individual modellers and those which are freely available for download. Spatial and temporal averaging and the associated considerations for predicting exposure and, ultimately, effects at the wildlife population level are not specifically addressed within the models that have been applied or the scenarios that have been used for model application.

The IAEA MODARIA (Modelling and Data for Radiological Impact Assessments)[6] programme is more specifically addressing considerations of spatial scales through the activities of Working Group 8. Through two case study scenarios (moose in Sweden and reindeer in Norway, as described in Appendix A) the application of animal-environment interaction models will be compared to conventional NHB dose assessment modelling approaches. However, there is a lack of radioecological data, especially direct measurements of external exposure of the case study animals, against which model predictions can be compared. It should also be noted that the considerations of spatial scales in the context of MODARIA are currently focussed towards how specific and/or representative individuals interact with heterogeneous contamination. The focus is not at the population level *per se*.

4.3 TRANSFER-EXPOSURE-EFFECTS (TREE)

Beyond the major international bodies that are helping to advance the field of NHB dose assessment, there are also various international research programmes that are continuing to make significant contributions in this area. One programme of particular relevance to discussions of spatial and temporal scales is the TRansfer – Exposure – Effects (TREE) project [Wood et al., 2014]. This 5-year research programme has been funded by four organisations in the United Kingdom, namely the Natural Environment Research Council (NERC), Radioactive Waste Management Ltd (RWM), the Environment Agency (EA) and the Science & Technology Facilities Council (STFC). The programme runs from October 2013 to September 2018 and involves research in Japan (Fukushima), Norway, Spain, Ukraine (Chernobyl) and the UK.

[6] The IAEA has a dedicated website that provides details on the MODARIA programme: http://www-ns.iaea.org/projects/modaria/default.asp?l=116

Of particular relevance for consideration here is the research being undertaken on medium-large mammals within the Chernobyl Exclusion Zone (CEZ). The prevailing climatic conditions at the time of the Chernobyl accident in 1986, in particular wind direction and rainfall, resulted in a highly heterogeneous spatial distribution of radionuclide deposition [Smith and Clark, 1986]. A network of motion-activated trail cameras, known as camera traps, has been established in three study locations within the CEZ that can broadly be classified as covering areas of high, medium and low contamination. These camera traps are providing information on the utilisation of different areas of the CEZ by a range of mammal species[7]. In the next phase of the research programme, the research team will use state-of-the-art satellite navigation technology to track the movement of selected large mammals within the CEZ over a period of 12 months. Preliminary data suggest that this will most likely be a canid species (*Nyctereutes procyonoides* or *Canis lupus*). GPS collars with integrated dosimetry technology will be fitted to the target species and provide accurate movement tracking coupled with georeferenced dose rate measurements. The data obtained within TREE will provide an ideal case study for the further evaluating the uncertainties associated with applying current simplistic modelling in comparison with the use of more advance animal-environment interaction models.

4.4 SUMMARY OF THE SPATIAL AND TEMPORAL SCALES PERSPECTIVES FROM INTERNATIONAL PROGRAMMES

There has been some consideration of spatial and temporal scales within ICRP and IAEA activities and also in the work being undertaken by ongoing international research programmes, such as TREE. Within the context of the ICRP the focus to date has been on the development of the RAP framework and relatively little attention has been given to the application of this at a population level. The IAEA EMRAS programmes have included some consideration of temporal and spatial scales, but this has largely been limited to exposure scenarios the have been explored during model intercomparison exercises. More recently, the IAEA MODARIA programme has initiated studies that consider the spatial interactions between animals and contamination within a heterogeneously contaminated environment, but the focus is on the movement of specific/representative individuals. The work within the IAEA MODARIA programme lacks direct exposure measurement data against which results can be compared. However, international research programmes such as TREE, which are undertaking these types of direct measurements, are expected to provide data that will enable more comprehensive evaluation of animal-environment interaction models in the future. In all cases (ICRP, IAEA and TREE), the focus of spatial and temporal scale considerations is not the population *per se*. There is an ongoing need for robust research that addresses the challenging question of how spatial and temporal scale considerations can be applied at the population level and what the influence of different assumptions is on the results obtained.

[7] Further information on the camera trap network and other aspects of the TREE research programme can be accessed via the project website: www.ceh.ac.uk/tree

5. BARRIERS AND KEY AREAS OF UNCERTAINTY

Radiation dose and risk assessments for humans over the long-term periods required according to international recommendations and national regulatory requirements, are recognised variously as being indicators of safety or measuring tools rather than absolute predictions of the impacts [NEA, 2012] and the same is true for long-term NHB assessments.

The radiation dose and risk assessment approach for long-term safety assessments has been developing for many years and has benefited from international collaborations (e.g. BIOMASS [IAEA, 2003] and various projects within BIOPROTA, among others). Whilst international collaboration has been a useful aspect in the development of assessment approaches for environmental protection, relatively little consideration has, as yet, been given to the application of methods in the context of long-term safety assessments. As such, those undertaking assessments have largely derived environmental input data for biota dose assessments from models established for the calculation of doses to people. The appropriateness of such model discretisation in terms of demonstrating compliance with environmental protection objectives would appear to be questionable, on the basis that spatial and temporal averaging relevant to exposure of a representative person or member of a critical group cannot automatically be expected to be the same as that for one of more populations of other biota.

The ICRP framework for environmental protection from ionising radiation makes it clear that consideration should be given to the risk of impacts to populations of plants and animals rather than focussing upon individuals alone. To achieve this, consideration must be given to the area required by a population in relation to the area of contamination. Thought should also be given to the reproductive strategies of the plants and animals since this will, in part, determine the temporal scale over which population effects may become apparent.

One of the primary issues associated with the incorporation of spatial scale considerations for plants and animals within long-term safety assessments is the diversity of species: it is not feasible to consider different scales of assessment for each species that may be representative of a site. Some form of aggregation may therefore be appropriate, potentially in terms of both species ranging habits and reproductive strategies; although, even with aggregation it would not be feasible to consider all ‘groups’ that may be relevant to repository safety cases.

A further potential barrier is that safety assessments are variable in the approach taken to address GBI release scenarios. There are two main categories of GBI scenarios in long-term safety assessments; well extraction and natural groundwater discharge to the biosphere [BIOPROTA, 2014]. Gas release may also be included in some cases as well as erosive release.

In the case of well extraction, time development of the surface environment may not need to be taken into account and the spatial area of assessment may be limited to the area directly affected by extracted water, for example an agricultural field subject to irrigation practices. Such managed systems are seldom the focus of NHB dose assessments: the protection of people is considered protective also of the animal living in what is essentially a human environment [ICRP, 2008]. Small animals (mammals, reptiles, amphibians and insects etc.) may be occasional visitors to agricultural areas but, with the exception of soil dwelling invertebrates such as earthworms, it is assumed to be unlikely that sustainable populations of wildlife would be inhabitants. A broader spatial scale would therefore require consideration in terms of environmental protection of wildlife, the scales of which would not normally be encompassed within assessment models designed to address protection of humans. Hence some degree of model development may be required to allow the spatial scales relevant to NHB populations to be considered and for radionuclide transport between these areas to be evaluated. Multiple biosphere assessment models may therefore be required to allow discretisation relevant to both people and NHB

for such scenarios and to allow the migration of radionuclides from agricultural soils to be evaluated. It should nonetheless be noted that simple assumptions of 100% occupancy at the site of discharge may be fit for purpose for wildlife dose assessments given these constraints.

In the case of groundwater discharge, the timescales of landscape development may be comparable with the timescales over which radionuclides move through the GBI. An explicit representation of that landscape development may therefore be required with consideration given to how radionuclides move from one component of the environment to another as the landscape evolves [BIOPROTA, 2014]. Such consideration of radionuclide migration though the biosphere is more conducive to the inclusion of spatial scales of assessment for biota, since radionuclides outwith managed systems are more relevant to wildlife populations. Dose rates to individuals throughout an area relevant to a population could therefore be modelled explicitly, based on assumptions around individual ranges, which would allow variation in individual dose rates to be considered. Alternatively, radionuclide activity concentrations in soils throughout the area of relevance to a population could be determined and the typical dose rate to individuals within that population evaluated.

6. IDENTIFICATION OF REPRESENTATIVE SPECIES AND ASSOCIATED PARAMETERS FOR EVALUATING TEMPORAL AND SPATIAL SCALES

In the preceding chapters, the rationale for considering spatial and temporal scales within NHB dose assessments has been discussed and the activities of major international organisations and of a relevant international research programme have been reviewed. In Chapter 5, various barriers and key areas of uncertainty have been highlighted. Drawing on these findings, an approach has been developed that will allow an evaluation to be undertaken of the influence of spatial and temporal scale assumptions when assessing doses to NHB. This approach has four main steps that are outlined below. Further information is presented in Appendix B.

6.1 SELECTION OF 'SPACE' REPRESENTATIVE SPECIES FOR EVALUATION

As stated in Chapter 1, due to the wide range of plants and animals present in terrestrial and aquatic ecosystems globally, the scope of the present project was necessarily constrained. To enable evaluation of spatial scales in NHB dose assessment within long-term safety assessments, whilst still providing a useful database for those undertaking biosphere assessments associated with radioactive waste disposal facilities, this project focussed on terrestrial environments within temperate climate regions.

6.1.1 Approach to species selection

A multi-stage approach was adopted for the development of the SPACE representative species list. Initially, generic types of organisms relevant to long-term safety assessments were identified based on assessment approaches such as ICRP [2008] and ERICA. These generic organism categories were then mapped onto species included to date within post-closure safety assessments and associated studies [e.g. Posiva, 2014a; Jaeschke et al. 2013; Torudd, 2010; Sheppard, 2002]. This mapping process produced an initial species list that was subsequently refined through consultation with SPACE project sponsors.

The SPACE representative species list is illustrative of the types of plant and animal that are likely to be of interest in assessments and covers all major terrestrial vertebrate groups (mammals, birds, reptiles and amphibians) and a selection of invertebrates, plants, mosses and fungi. The purpose of the species selection process was to provide a focus for targeted data collection to support the on-going development of the SPACE methodology. Selection therefore focussed on those representative species for which assessment parameter data were most likely to be available. Agricultural animals have not been considered specifically. This is consistent with the ICRP [2008] view that such animals, living essentially in a managed human environment, would be covered by the human animal itself.

The SPACE representative species that have been used for the purposes of SPACE data collation and modelling during subsequent phases of the SPACE project are presented below.

Vertebrates

Vertebrates are considered in terms of mammals, birds, reptiles and amphibians.

Mammals

- **ICRP Deer**. Reference deer [ICRP, 2008] is representative of a large woodland deer. In addition to information presented in ICRP [2008], the following representative species were selected as the basis for deriving data on spatial and temporal scales:
    - Roe deer (*Capreolus capreolus*)
    - White-tailed deer (*Odocoileus virginianus*)
    - Moose (*Alces alces*)
- **ICRP Rat (and other small burrowing mammals)**. Reference rat belongs to the family Muridae to which mice, hamsters, lemmings and voles also belong. In collating data, it was proposed that the scope be extended beyond the Muridae family to encompass other small mammals with burrowing habits. The following representative species were therefore selected:
    - Brown rat (*Rattus norvegicus*)
    - Bank vole (*Myodes glareolus*)
    - Water vole (*Arvicola amphibious*)
    - Rabbit (*Oryctolagus cuniculus*)
- **Medium and large ground-dwelling mammals**. Not specifically included within ICRP RAPs or ERICA, but with potential for exposure due to ground-dwelling habits. The following species were selected for consideration:
    - Badger (*Meles meles*)
    - Red fox (*Vulpes vulpes*) and arctic fox (*Vulpes lagopus*))
    - American mink (*Neovison vison*)
    - Gray Wolf (*Canis lupus*)

Birds

Due to the nature of assessed releases to the biosphere, exposure of plants and animals will be largely associated, in the terrestrial environment, with soil occupancy. As such, consideration to representative species for the purposes of this study was restricted to those terrestrial (and semi-aquatic birds) that primarily reside on the soil surface.

- **ICRP duck.** Ducks are semi-aquatic in habits and both the adult and egg life stages are of interest. The representative species selected, including other non-duck ground-nesting species, include:
    - Mallard duck (*Anas platyrhynchos*)
    - Willow Ptarmigan (*Lagopus lagopus*)
    - Pheasant (*Phasianus colchicus*)

- Greylag goose (*Anser anser*)

Feral pigeon (*Columba livia*)

Reptiles

It was proposed that considerations around reptiles be restricted to the order Squamata. This is the largest order of reptiles and comprises all lizards and snakes. The following representative species were selected:

- Adder (*Vipera berus*)
- Common lizard (*Zootoca vivipara*)

Amphibians

Amphibians commonly require water for breeding purposes with eggs and juveniles being aquatic inhabitants and adults being primarily terrestrial inhabitants. Different life stages may therefore be exposed to varying extents. The ICRP RAPs approach includes Reference Frog with each life stage being specifically represented.

- ICRP Frog. All three life stages (egg, tadpole, adult) are specifically represented. The selected representative species were:
    - Common frog (*Rana temporaria*)
    - Common toad (*Bufo bufo*)
    - Great crested newt (*Triturus cristatus*)

Invertebrates

The invertebrates are a diverse group of organisms. In terms of biota assessments, consideration to terrestrial organisms largely includes flying insects and earthworms. Suggested representatives for inclusion in the study were:

- **ICRP bee**. Bees play a vital role in the ecology of terrestrial ecosystems, many being vital to plant pollination. Reference bee is assumed to be a typical social bee. Selected representative species:
    - Common honey bee (*Apis mellifera*)
- Ants (social insects often inhabiting the soil environment). Selected representative species:
    - Black garden ant (*Lasius niger*)
- Earthworm.
    - Common earthworm (*Lumbricus terrestris*)

Plants

The range in spatial scales to represent plant populations is likely to be less diverse than for animals due to their sessile nature. As such it was proposed that the study be limited to two broad categories of

plants that are covered by the ICRP RAPs: 'Pine Tree' and 'Grasses and herbs'. From the perspective of SPACE, these two plant categories cover populations that exhibit contrasting spatial and temporal scale requirements. Proposed representative species:

- Scots pine (*Pinus sylvestris*)
- Fescue grasses (*Festuca sp.*)

Given their propensity to accumulate high levels of some radionuclides and their differing life-histories, the following representatives of the mosses and fungi were also included within the SPACE representative species list:

- Glittering wood-moss (*Hylocomium splendens*)
- Fragile brittlegill fungus (*Russula fragilis*)

6.2 COMPILATION OF A DATABASE OF TEMPORAL AND SPATIAL SCALE PARAMETERS FOR THE SPACE REPRESENTATIVE SPECIES

For the selected SPACE representative species, a literature review strategy was developed to collate data on lifespan, home range, territoriality, global distribution and whether or not the species is migratory. The literature review strategy was based on a 'Systematic Review' methodology, details of which are presented in Appendix B.

One of the complicating factors when collating temporal and spatial parameters for wildlife is the variability in those parameters for individual species as a result of other environmental factors [e.g. Schradin et al. 2010; Torres et al. 2012; van Moorter et al. 2013]. For example, small mammals inhabiting sand dunes have large home ranges compared with those inhabiting woodlands (kilometres versus 10s of metres) as a result of the lower food availability per unit area in sand dune habitats [Akbar and Gorman, 1993]. Home ranges, also known as forage areas [Hope 2005], can be defined as the areas occupied by individuals or in which they spend 95% of their time [Minta 1992; USEPA 1993; Waser 1987]. Another way to define home range that has been proposed more recently is "*that part of an animal's cognitive map of its environment that it chooses to keep updated*" [Powell and Mitchell 2012]. Essentially the concept of home range reflects the extent of an individual's spatial interaction within a landscape.

Prevailing environmental conditions can have a pronounced impact on the spatial parameter values for each SPACE representative species and the representativeness of data from individual studies may therefore be questioned. Temporal parameter variability, as a result of seasonality for example, may also be observed [Wood et al. 2009]. A systematic review approach was specifically adopted to minimise issues of data bias by drawing on a wide range of both published and unpublished sources.

Previous research on the approaches to assessing impacts on wildlife, due to radionuclide releases from radioactive waste disposal facilities to the biosphere, has identified that protecting biodiversity (a specified target of protection at both the national and international level [ICRP 2003; STUK 2010]) is synonymous with protection of populations [Smith et al. 2012a; Jackson et al. 2014]. Therefore, SPACE representative species data were required to be representative of the population rather than the individual. However, the spatial extent of populations is rarely defined in the literature because there are many factors that may significantly modify the population range. These include the habitat structures present and extent of connectivity to other areas supporting representatives of that species. Acknowledging that population range data is not readily obtainable, the spatial data compilation focussed on home ranges and data were converted to SI units (km^2).

The extent of territoriality was recorded where this information was available, recognising that the extent of territoriality is often a function of age (territoriality increasing with sexual maturity) [Maublanc et al., 2012]. This provides an indication of the degree to which home ranges may be expected to overlap. An approach that has been adopted in other contaminant exposure modelling studies for estimating the spatial scale of the population is to adopt a scaling factor. Hope [2005] uses a scaling factor of 40, so the assessment population area is assumed to be 40 times the home range area. The information on the degree of territoriality, and hence the extent to which home ranges may be expected to overlap, may be used to inform the selection of an appropriate scaling factor to estimate population size from home range size.

6.2.1 Parameters for SPACE representative species

A common criticism of many previous data compilations used within wildlife risk assessment modelling is the lack of clarity over data provenance. Therefore, all SPACE representative data collated through the systematic review process were recorded in a database along with details of the data source. As a result, all values used in the parameterisation of the SPACE representative species can be traced back to the original data source(s). The summarised dataset is presented in Tables 6-1 to 6-3.

6.3 CATEGORISING SPACE REPRESENTATIVE SPECIES

In any modelling process, it is advantageous to be able to minimise the number of times that a model must be run or the range of modelling parameters that need to be entered. The purpose of modelling is to provide a 'sufficiently accurate' representation of reality and, from the perspective of SPACE, there is a focus on both the spatial and temporal averaging of radionuclide activity concentration data for use within dose assessment models for wildlife. Given that the appropriate scale of spatial (e.g. metres to kilometres) and temporal (e.g. month to years) averaging of relevance from ecological risk assessment perspective may vary depending on the specific species under consideration, categorising SPACE species based on their spatial and temporal scale attributes should increase modelling efficiency and ensure models are fit for purpose.

To facilitate the implementation of scale considerations within NHB dose assessments, a categorisation strategy was therefore developed to group SPACE representative species, based on their spatial and temporal scale characteristics. The parameters for SPACE representative species presented in Section 6.2 were reviewed to assess the potential for categorisation into 'SPACE reference groups'.

Given that radionuclide releases to the biosphere from a radioactive waste repository are unlikely to be pulsed, unless there are winter periods when the ground is frozen, the intra-annual temporal variability in wildlife activity is unlikely to be of relevance. More relevant from the perspective of temporal averaging is the life span of organisms. When deciding on temporal averaging time steps, consideration should be given to the number of generations that a selected time step would cover. Radiation damage to germ cells may result in impacts in subsequent generations so, if peak activity concentrations in the biosphere lasting one or a few generations are 'lost' within temporal averaging over many generations, the resultant dose assessment may fail to identify potentially significant impacts. Therefore, one potential categorisation approach for SPACE representative species is to group species by life span. The life span of an organism appears to be fairly consistent across different environmental settings, so adopting this categorisation approach should allow the SPACE representative species to be grouped in a manner that would be useful for a wide range of assessment scenarios. A proposed grouping based on life span is presented in Table 6-4.

From the spatial perspective, the focus is on the relationship between the assessment area (area of contamination) and the spatial range of the population. Animals with large home ranges in comparison

with the assessment area may be deemed to be spatially irrelevant from the perspective of risk assessment due to the limited amount of time that an individual is exposed within the assessment area and the small proportion of the population that is exposed [Tannenbaum et al. 2013].

Table 6-1. SPACE representative species data: Mammals.

Common Name	Latin name	Home range (km^2)	Life span (y)	Distribution	Territoriality	Migratory?
Roe deer	*Capreolus capreolus*	6.0E-1 – 1.0E+1[a,b,c,d]	15[e]	Whole of Europe except Ireland, Iceland, northern Scandinavia and the Mediterranean Islands, northern Asia as far east as China.[f]	Groups of 3 - 30 individuals[f]	N[g] (Some may migrate seasonally)
White-tailed deer	*Odocoileus virginianus*	2.0E-1 – 1.0E+1[h,i,j,k]	10[l]	Americas, NZ & Europe[m]	Some territoriality[k]	N[m]
Moose	*Alces alces*	1.3E+1 – 2.6E+1[n,o]	12[p]	Scandinavia, northeast Europe, northern Asia, Canada, Alaska[f]	Solitary[f]	N[f]
Brown rat	*Rattus norvegicus*	3.3E-4 – 9.0E-3[q]	2[r]	Global (except Antarctica)[f]	Large family groups of up to 60 individuals[f]	N[f]
Bank vole	*Myodes glareolus*	4.0E-4 – 7.0E-4[s]	2[t]	Europe (except most of Iberian Peninsula and Scandinavia), northern Asia to central Siberia. Absent from Iceland and most of Ireland.[f]	Can be territorial, but up to 100 voles per ha have been recorded[u]	N[f]
Water vole	*Arvicola amphibius*	2.0E-1 – 2.6E+0[v]	2[w]	Europe (except Iceland, Ireland, much of France and most of Spain and Portugal), Asia[f]	Territorial, but home ranges overlap[x]	N[f]
Rabbit	*Oryctolagus cuniculus*	7.0E-3 – 2.1E-2[y]	9[l]	Western, central and southern Europe (except Balkan Peninsula), northwest Africa[f]	Social with large family groups[f]	N[f]
Badger	*Meles meles*	5.4E+0 – 9.5E+0[z,aa]	14[ab]	Europe (except northern Scandinavia, Iceland, Corsica, Sardinia, Sicily & Cyprus), temperate parts of Asia, east to China[f]	Setts of 5 - 10 individuals[f]	N[f]
Red fox	*Vulpes vulpes*	1.4E+0 – 5.0E+1[f,ac]	3[ad]	Europe (except Iceland), Asia, North Africa, North America[f]	Territorial groups of one male & several females[ae]	N[f]
American mink	*Neovison vison*	3.0E-1[f]	10[e]	North America, Europe & South America[af]	Territorial, but with overlap of male & female home range sizes[ag]	N[ah]
Gray wolf	*Canis lupus*	9.8E+1 – 2.3E+2[ai]	10[f]	Eastern Europe, Scandinavia, southeast Europe, Spain, Portugal, Sardinia, Italy, Asia and North America[f]	Lives in packs[f]	N[f]

[a]Maillard et al. 2002; [b]Mysterud 1999; [c]Zejda and Bauerova 1985; [d]Guillet et al. 1996; [e]Grzimek 1990; [f]Hofmann 1995; [g]Ramanzin et al. 2007; [h]Webb et al. 2010; [i]Tannenbaum 2013; [j]Tamara Yankovich, pers. Comm.; [k]Gavin et al. 1984; [l]Nowak 1999; [m]Nelson and Mech 2006; [n]Cederlund and Sand 1994; [o]van Beest et al. 2011; [p]Wilson and Ruff 1999; [q]Lambert et al. 2008; [r]Nowak and Paradiso 1983; [s]Lindblom 2008; [t]Macdonald 2001; [u]Bujalska and Grum 2013, Hansson and Henttonen 1985; [v]Pita et al. 2011; [w]Nazarova 2013; [x]Moorhouse and Macdonald 2008; [y]Devillard et al. 2008; [z]Myslajek et al. 2012; [aa]Kauhala and Holmala 2011; [ab]Delahay et al. 2008; [ac]Janko et al. 2012; [ad]Ables 1975; [ae]Lindstrom 1989; [af]Macdonald and Harrington, 2003; [ag] Zschille et al. 2012; [ah]Zschille et al. 2010; [ai]Nowak et al. 2008.

Table 6-2. SPACE representative species data: Birds, Reptiles & Amphibians.

Common Name	Latin name	Home range (km^2)	Life span (y)	Distribution	Territoriality	Migratory?
Mallard duck	*Anas platyrhynchos*	1.0E+0 – 2.8E+1[a,b]	23[c]	Temperate and subtropical Americas, Europe, Asia, and North Africa, and has been introduced to New Zealand and Australia[d]	Ranges overlap considerably[d]	Y[d]
Willow Ptarmigan	*Lagopus lagopus*	2.6E-2[e]	15[f]	Widespread and common throughout the Northern hemisphere[g]	Some territoriality, but up to 207 birds per square mile have been recorded[h]	Females may migrate up to 100 miles to overwinter[e]
Pheasant	*Phasianus colchicus*	1.1E+1 – 3.2E+0[i,j]	2[k]	Eurasia, Europe, North America, New Zealand, Australia[j]	Not very territorial, form breeding harems[j]	N[j]
Greylag goose	*Anser anser*	1.0E+1[l]	20[m]	Widespread global distribution[n]	Flock forming[n]	Y[o]
Feral pigeon	*Columba livia*	2.25E-2[p]	6[q]	Widespread global distribution[r]	Flock forming[p]	N[s]
Adder	*Vipera berus*	5.2E-2[t]	20[u]	Widespread through Europe to the Pacific coast of Asia[u]	Solitary except when hibernating[u]	N[v]
Common lizard	*Zootoca vivipara*	5.6E-4 – 1.7E-3[w,x,y]	10[z]	Widespread across Eurasia[z]	Often encountered in groups[v]	N[v]
Common frog	*Rana temporaria*	5.0E-4[aa]	14[ab]	Throughout Europe[ac]	Solitary[ac]	N[v]
Common toad	*Bufo bufo*	1.0E+0[aa]	12[ab]	Widespread throughout Europe, west Asia and north Africa[ad]	Solitary[ae]	N[v]
Great crested newt	*Triturus cristatus*	5.0E-4[af]	10[aa]	Europe & parts of Asia[ag]	Solitary, but up to 1250 adults per ha[aa]	N[v]

[a]Legagneux et al. 2009; [b]Mack 2003; [c]Clapp et al. 1982; [d] Pöysä et al.1998; [e]Bowman 2003; [f]Braun et al. 1993 (assumed that data on *Lagopus leucurus* life span would be a suitable surrogate); [g]http://www.birdlife.org/datazone/speciesfactsheet.php?id=290; [h]Johnsgard 2008; [i]Applegate et al. 2002; [j]Blackburn et al. 2001, Göransson et al. 1990, Williams et al. 2003; [k]Weigand et al. 1976, Schantz et al. 1997; [l]Kear 2005; [m]de Magalhaes and Costa 2009; [n]Fox and Kahlert 2005; [o]Newton 2006; [p] Sol and Senar 1995 (data for urban pigeons); [q]Johnston 1992; [r] Hahn et al. 2009; [s]Caula et al. 2008; [t]Neumeyer 1987; [u]Forsman and Lindell 1991; Ursenbacher et al. 2006, Herczeg et al. 2007; [v]Arnold 2004; [w]Massot 1992; [x]Ortega-Rubio et al. 1990; [y]Verwaijen and Darmne 2008; [z]Massot et al. 2011; Bleu et al. 2013; Martin et al. 2013; [aa]Langton and Beckett 1995; [ab]Smirina 1994; [ac] Teacher et al. 2009; [ad] Garcia-Porta et al. 2012; [ae] Cunningham et al. 2007; [af]http://www.naturalengland.org.uk/Images/GreatCrestedNewts_tcm6-21705.pdf; [ag]Arntzen and Wallis 1999.

Table 6-3. SPACE representative species data: Invertebrates, Plants, Mosses & Fungi.

Common Name	Latin name	Home range (km^2)	Life span (y)	Distribution	Territoriality	Migratory?
Common honey bee	*Apis mellifera*	8.0E-04[a]	<1 (workers), 8 (queen)[b]	Near global distribution[c]	Social, live in hives[d]	N[b]
Black garden ant	*Lasius niger*	1.6E-06[e]	<2 (male), 28 (queen)[f]	Europe, North America and Asia[f]	Social, colony forming[g]	N[f]
Common earthworm	*Lumbricus terrestris*	3.0E-06[h]	6[h]	Europe and North America[i]	Solitary, but 30-200 individuals/m^2[j]	N[h]
Scots pine	*Pinus sylvestris*	n.d.	<500[k]	Europe, western Asia, North America and New Zealand[k]	N/A	N/A
Fescue grass	*Festuca sp.*	2.0E-08[l]	0.55[m]	Europe, Asia and North America[n]	N/A	N/A
Glittering Wood-moss	*Hylocomium splendens*	n.d.	20[o]	Europe, North America, Japan, North Africa, Australia, New Zealand[p]	N/A	N/A
Fragile brittlegill fungus	*Russula fragilis*	n.d.	n.d.	Europe, Asia and North America[q]	N/A	N/A

n.d. = no data located during review; N/A = information not applicable; [a]Hagler et al. 2011; [b]Page and Peng 2001; [c]Abrol 2012; [d]Farina et al. 2005; [e]Doncaster 1981; [f]Brian 1964, Keller 1998, Niculita et al. 2007; [g]Depickere et al. 2008; [h]Velavan et al. 2009; [i] Hale et al 2005; [j]Sims and Gerard 1985; [k]Norin and Winell 1972, Angelstam 1997, Rehfeldt et al. 2002, Prus-Glowacki and Stephan 1994; [l]Fairey and Lefkovitch 1996; [m]Janisova 2007; [n]http://www.brc.ac.uk/plantatlas/index.php?q=node/1958; [o]Callaghan et al. 1997; [p]Sainsbury 1942; [q]Pala et al. 2011

Table 6-4. SPACE representative species temporal range categorisation

SPACE reference group	Common name	Latin name
< 5 y	Fescue grass	*Festuca sp.*
	Fragile brittlegill fungus[a]	*Russula fragilis*
	Brown rat	*Rattus norvegicus*
	Bank vole	*Myodes glareolus*
	Water vole	*Arvicola amphibius*
	Red fox	*Vulpes vulpes*
	Pheasant	*Phasianus colchicus*
5 - 10 y	Common honey bee	*Apis mellifera*
	Feral pigeon	*Columba livia*
	Common lizard	*Zootoca vivipara*
	Common earthworm	*Lumbricus terrestris*
	Glittering Wood-moss	*Hylocomium splendens*
	Rabbit	*Oryctolagus cuniculus*
	White-tailed deer	*Odocoileus virginianus*
	American mink	*Neovison vison*
	Gray wolf	*Canis lupus*
	Great crested newt	*Triturus cristatus*
11 - 25 y	Greylag goose	*Anser anser*
	Moose	*Alces alces*
	Common toad	*Bufo bufo*
	Badger	*Meles meles*
	Common frog	*Rana temporaria*
	Roe deer	*Capreolus capreolus*
	Willow Ptarmigan	*Lagopus lagopus*
	Mallard duck	*Anas platyrhynchos*
	Adder	*Vipera berus*
>25 y	Black garden ant[b]	*Lasius niger*
	Scots pine	*Pinus sylvestris*

[a]In the absence of data for this species, the life span for Fescue has been used; [b]Black garden ant queen.

As noted previously, individual home ranges, population density and hence the spatial range of population are highly dependent on the prevailing conditions (e.g. food availability, distribution of suitable habitat within the landscape etc.). Although there is some evidence of behavioural plasticity in ranging activity of selected species in relation to certain environmental characteristics [Morellet et al. 2011], it is clear from the literature reviewed during the data collation for the SPACE representative species that factors such as habitat characteristics, climate and food availability can all influence spatial extent of populations [e.g. Maillard et al. 2002; Mysterud 1999; van Beest et al. 2013]. For example, snow cover has been shown to produce a ten-fold increase in the home range of *Capreolus capreolus* [Zejda and Bauerova 1985]. Different sexes can also have different home range size requirements. For example, Lambert et al. [2008] measured home range sizes for female and male *Rattus norvegicus* living in agricultural field margins and found the home range size for males to be an order of magnitude higher than that of females (163 m^2 and 8992 m^2 respectively). A further consideration is the diversity of habitat present, with differential habitat use enabling otherwise competing species to co-exist within heavily overlapping spatial ranges [Medina-Vogel et al. 2013].

Given the range of influences on home range size, categorising SPACE representative species on the basis of assumed population ranges is unlikely to justifiable beyond the very coarse classification.

One approach would be to categorise on the basis of migratory or non-migratory species. This coarse distinction may be of value because the populations of migratory species will only spend a fraction of their time within the assessment area, whereas populations of non-migratory species may be partially or wholly resident within the assessment area. Effectively the migratory species may be viewed as spatially irrelevant [*sensu* Tannenbaum et al. 2013].

An alternative approach is to broadly classify the organisms into immobile, small, medium and large home range sizes (Table 6-5). This SPACE reference group categorisation may be useful for a general wildlife dose assessment. However, if there are specific organisms of concern within an assessment, once the likely environmental conditions and landscape characteristics for the specific assessment time period have been identified, the SPACE representative species data presented in this report, coupled with the additional information available within the source references, could be used to provide a more location-specific estimation of likely home ranges (and hence population ranges) for the prevailing environmental conditions.

6.4 SUMMARY

Collation of temporal and spatial scale data for the SPACE representative species has enabled the evaluation of opportunities to categorise organisms into SPACE reference groupings.

The data evaluation has identified a series of temporal groupings, namely <5, 5-10, 10-25 and >25 years. Undertaking long term assessments of radioactive waste repositories using these short time steps is unlikely to be realistic given the uncertainty in models of future releases. However, it should also be noted that, while landscape and other modelling from disposal until the time of assessed radionuclide release to the biosphere may inform the dose assessment, the dose assessment modelling only begins at the time when the releases to the biosphere begin. With the potential exception of the Scots Pine (lifespan <500 years), SPACE representative species are likely to have multiple generations within the averaging time steps that are appropriate for long-term modelling. Attempts to model shorter time steps for long-term assessments (e.g. decadal time steps) are likely to entail significant extra resource and cost, whilst at the same time yielding results that would be difficult to defend on a robust scientific basis. That said, if there was concern over the potential impact on wildlife from exposure to ionising radiation then a focused assessment around the time steps typically applied to human dose assessments using the identified wildlife temporal groupings may be appropriate.

For grouping organisms on the basis of spatial scale, a general classification of organisms has been proposed (immobile, small, medium and large ranges). The focus of environmental radiation protection is generally protection of populations, so population ranges need to be estimated from individual home range sizes (e.g. using the multiplier proposed by Hope [2005]). If the spatial scale of an organism population is smaller than the spatial averaging range used for the human assessment, there may be justification in undertaking assessments for those organisms using smaller spatial averaging areas that include peak concentrations.

In the next chapter we use a spatial assessment scenario to evaluate the influence of different spatial averaging assumptions on the calculation of doses for different SPACE reference groupings. For the reasons described above, an equivalent analysis of temporal scales has not been undertaken.

Table 6-5. SPACE representative species spatial range categorisation

SPACE reference group[a]	Common name	Latin name
Immobile	Fescue grass	*Festuca sp.*
	Fragile brittlegill fungus	*Russula fragilis*
	Glittering Wood-moss	*Hylocomium splendens*
	Scots pine	*Pinus sylvestris*
Small	Black garden ant	*Lasius niger*
(<0.5km^2)	Common earthworm	*Lumbricus terrestris*
	Great crested newt	*Triturus cristatus*
	Common frog	*Rana temporaria*
	Bank vole	*Myodes glareolus*
	Common honey bee	*Apis mellifera*
	Common lizard	*Zootoca vivipara*
	Brown rat	*Rattus norvegicus*
	Rabbit	*Oryctolagus cuniculus*
	Adder	*Vipera berus*
	Feral pigeon	*Columba livia*
	Willow Ptarmigan	*Lagopus lagopus*
Medium	American mink	*Neovison vison*
(0.5 - 10 km^2)	Common toad	*Bufo bufo*
	Water vole	*Arvicola amphibious*
	Pheasant	*Phasianus colchicus*
	White-tailed deer	*Odocoileus virginianus*
	Roe deer	*Capreolus capreolus*
	Badger	*Meles meles*
Large	Greylag goose	*Anser anser*
(>10km^2)	Mallard duck	*Anas platyrhynchos*
	Red fox	*Vulpes vulpes*
	Moose	*Alces alces*
	Gray wolf	*Canis lupus*

[a]The spatial classification is based on the assumed average spatial range for the SPACE representative species.

7. EVALUATING THE INFLUENCE OF SPATIAL SCALE ON DOSE ASSESSMENT RESULTS

Data collated on individual home range size and life span for each of the SPACE representative species (Chapter 6; Tables 6-1 to 6-3) were used to define SPACE reference groups in two ways: (i) temporal range categorisation (Chapter 6; Table 6-4); and (ii) spatial range categorisation (Chapter 6; Table 6-5). This chapter uses a Geographical Information System (GIS) based analysis of a hypothetical disposal facility release scenario to critically evaluate the applicability of the proposed reference groupings and to investigate the variability in total dose estimates that results from making different spatial averaging assumptions for the determination of the media activity concentrations to be used in NHB dose assessments. Since a natural groundwater discharge GBI scenario is more relevant to wildlife population exposures in terrestrial ecosystems, this formed the focus for evaluating the implications of scale assumptions within repository safety assessments.

It is recognised that a key uncertainty in biota dose assessments is associated with assessment parameters, particularly concentration ratios for which many data gaps are evident [Howard et al., 2013]. To avoid additional assessment complexities, beyond those associated specifically with scales of assessment evaluation, the radionuclides assessed were those for which assessment parameters were broadly available (e.g. CRs available within the Wildlife Transfer Database [Copplestone et al., 2013]) to minimise the need to derive parameters. However, in the case of large terrestrial mammals for which transfer parameters are not readily available, analogue application from alternative terrestrial mammals was employed. Whilst there may be uncertainty around the values applied within the assessment tool, this is considered beyond the scope of the current project. The remainder of this chapter describes the approach taken to critically evaluate different spatial scale assumptions on dose rate calculations for NHB, using the ERICA Tool.

7.1 DEVELOPMENT OF AN EVALUATION SCENARIO

To enable evaluation of a broad range of radionuclides that may need to be considered within biosphere assessments for radioactive waste disposal facilities and to provide data at a resolution appropriate for evaluating across different spatial scales, a hypothetical release scenario was developed. Whilst some spatial data were available from biosphere assessments that have already been undertaken (e.g. in Finland and Sweden), the spatial resolution of these data was not sufficient to allow investigation of the averaging assumptions being considered across the range of SPACE representative species. The development and application of the hypothetical release scenario and the critical evaluation of different spatial scale assumptions are presented below.

7.1.1 Radionuclides

A range of radionuclides relevant for disposal facility biosphere assessments was considered in the evaluation of spatial averaging approaches. The radionuclides are largely those considered routinely within the BIOPROTA forum. These radionuclides were grouped according to their assumed environmental mobility (Table 7-1).

Table 7-1. Radionuclides considered within SPACE

Environmental mobility category	Radionuclide
High	C-14
	Cl-36
	I-129
	Tc-99
Medium	Cs-137
	Np-237
	Pb-210
	Po-210
	Ra-226
	Se-79
Low*	Th-230
	U-238

* Additional thorium and uranium isotopes were also analysed, with results presented in Appendix D.

7.1.2 Application of GIS for the SPACE hypothetical release scenario

To analyse the potential exposure of 24 SPACE representative species to radionuclides released from a hypothetical radioactive waste disposal facility, it was necessary to create a spatial model within a GIS. The ESRI ArcGIS 10.2 package was used for this purpose.

Firstly, a modelling grid was established to define the spatial extent of the assessment area. Consideration was given to establishing a modelling grid of a size that would have been larger than the maximum spatial extent of any SPACE representative species population. For the purposes of this analysis, the spatial scale of each SPACE representative species population was defined using Hope's [2005] scaling factor of 40 as a home range multiplier. The SPACE representative species with the largest maximium home range was the Gray wolf (230 km^2), so the estimated population range for this species would be 9200 km^2. However, the subsequent modelling to be performed within GIS would be computer processor-intensive and, given the need to subsequently define a grid cell resolution that was small enough to allow SPACE representative species with small population ranges to be evaluated, it was preferable to define a less processor-intensive assessment area for evaluation. It was also thought unlikely that radionuclide releases to be considered within a biosphere assessment would need to be assessed over such a large spatial extent. It was decided that, as a minimum, the dataset should cover an estimated population range that would be predicted for the lower limit of the largest of the SPACE categories (>10 km^2). Therefore, the maximum spatial extent of the grid was set to 400 km^2.

The next stage of the GIS model development was to define the spatial resolution of the modelling grid. Again, there was a need to consider the influence of grid resolution on subsequent computer processing requirements whilst still ensuring that an appropriate evaluation of spatial scales could be performed. To maximise the efficiency of model runs two models were developed, one with a 50 m^2 resolution and the other with a 100 m^2 resolution. In the subsequent analyses, the appropriate modelling resolution was selected for the SPACE representative species under evaluation; species with large home ranges being modelled using the 100 m^2 grid resolution, whereas small and medium species were modelled using the 50 m^2 grid resolution.

7.1.3 Modelling radionuclide activity concentrations within the assessment area

It was assumed that the disposal facility was situated in the centre of the 400 km^2 grid. Three randomly selected release points were digitized within 3 km of the grid's centre. It was assumed that 1 Bq kg^{-1} of each radionuclide was present at each release point. Each of these release points was then used to model three hypothetical circular plumes, one for low mobility radionuclides, one for medium mobility radionuclides and one for high mobility radionuclides). The plume radius was altered to reflect the environmental mobility of the radionuclides (low = 1 km, medium = 5 km and high = 10 km).

Distances (in metres) were calculated from each plume centre up to a maximum plume radius of 10 km. Beyond 10km from the plume centre, the activity concentrations were assumed to be zero. A dispersion coefficient was then calculated, assuming that radionuclide activity concentrations would be highest at the point where the plume reaches the soil surface and then decrease with increasing distance from the plume release point. Fuzzy logic was used to reclassify the distance in metres from each release point to obtain a dispersion coefficient. The dispersion coefficient ranged from 1 to 0, one being the closest to the release point and 0 the point where there was zero concentration. Given that the release points were randomly selected within 3 km of the grid's centre, plumes of the medium and high mobility radionuclides overlapped in certain areas, resulting in elevated soil activity concentrations in these areas (Figures 7-1 to 7-3).

The next section describes the use of different spatial ranges to average concentrations. In some instances, the averaging area extended beyond the boundaries of the modelling grid. For any part of the averaging area located outside of the modelling grid, the activity concentrations of all radionuclides were assumed to be zero.

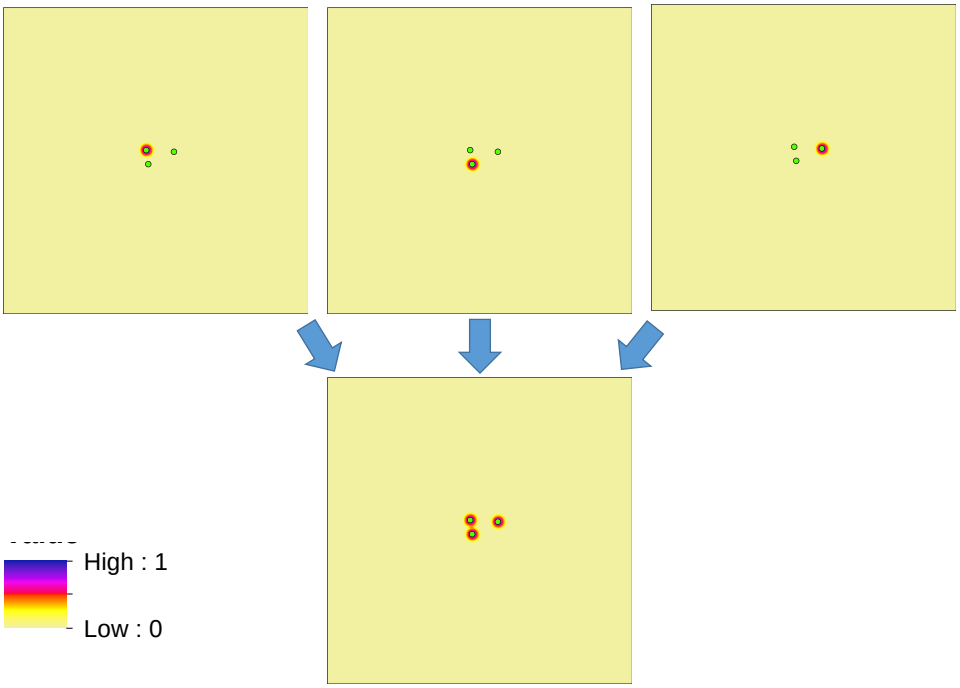


Figure 7-1. Dispersion of low mobility radionuclides (1km maximum dispersal radius) from each of the three randomly selected plume centres. The scale describes the estimated soil activity concentration (scaled from 0 to 1 Bq kg^{-1}).

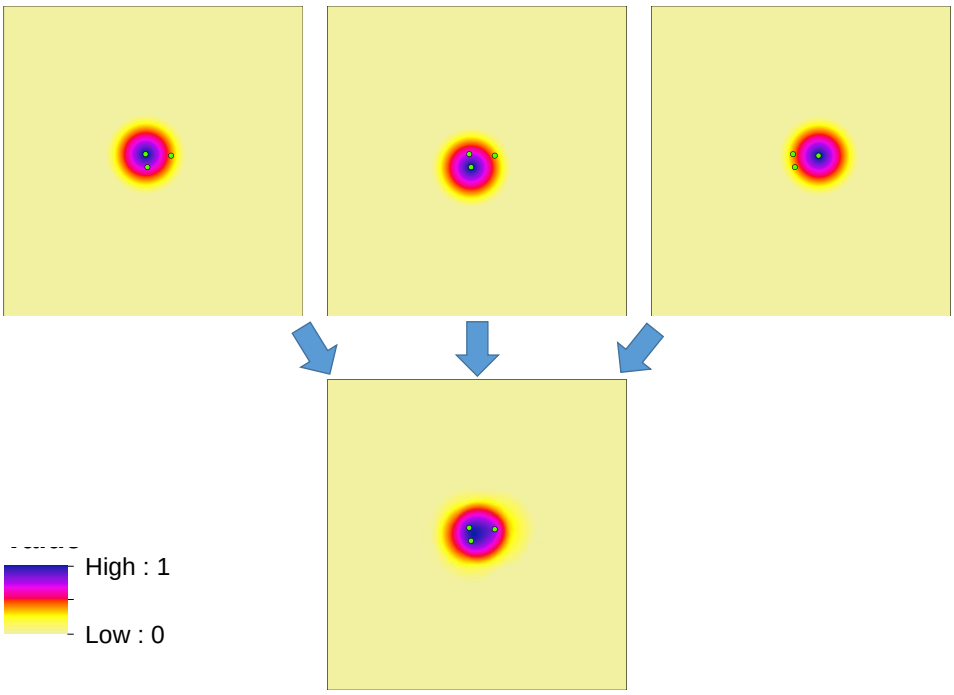


Figure 7-2. Dispersion of medium mobility radionuclides (5 km maximum dispersal radius) from each of the three randomly selected plume centres. The scale describes the estimated soil activity concentration (scaled from 0 to 1 Bq kg^{-1})

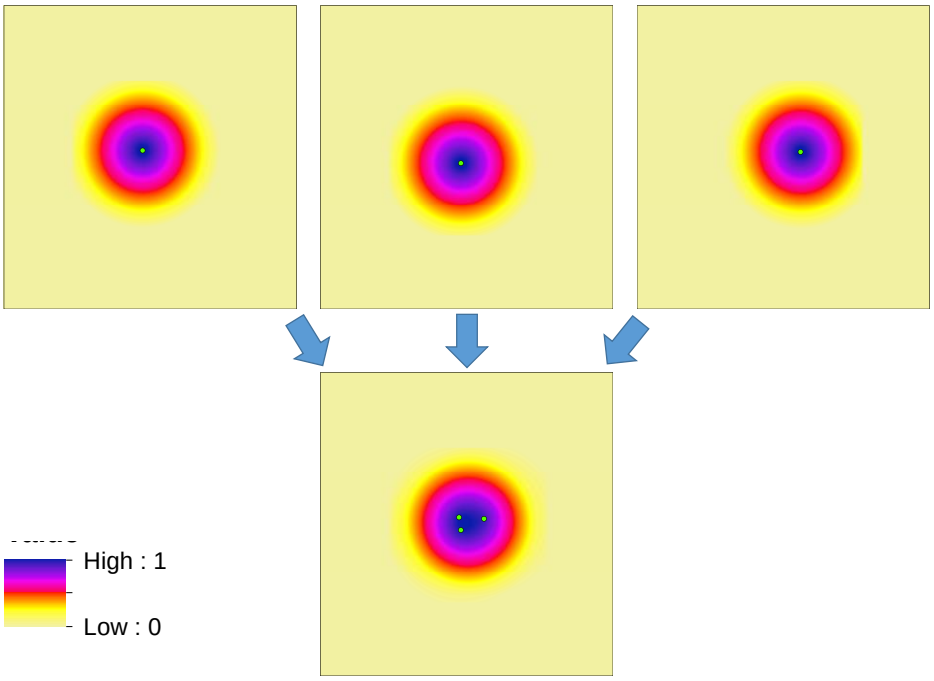


Figure 7-3. Dispersion of high mobility radionuclides (10 km maximum dispersal radius) from each of the three randomly selected plume centres. The scale describes the estimated soil activity concentration (scaled from 0 to 1 Bq kg^{-1}).

7.1.4 Derivation of a representative human scale of assessment

Two basic issues emerge from the discussion in Section 2 which drive the selection of a representative human scale area of assessment. These are the habits of humans living in that area, and the nature of the GBI. For the commonly considered release of radionuclides in groundwater, the GBI divides into release via abstraction from a well and natural discharge to a surface area. A range of factors affects how habits and the GBI need to be considered together in selecting a relevant area for human assessment. These include, in particular: the level of knowledge of the local groundwater flow system carrying the radionuclides from the geosphere; the time before such release occurs and its temporal extent; and other components of the assessment context [IAEA, 2003], notably any regulatory or other prescriptions on assumptions for human behaviour.

The purpose of the current evaluation scenario is to illustrate approaches for NHB averaging and investigate the implications. The assessment context is very generic. Therefore, the methodological approach to evaluation of the factors mentioned above, for determining an appropriate basis for human assessment (discussed in IAEA [2003] and further in Section 2.2) provides little opportunity to constrain the range of possibilities. Accordingly, for the purposes of the current illustrative calculations, a value of 2.00E04 m^2 has been selected from the range of examples mentioned in Section 2.2. It may be noted that a smaller area would in many environments be insufficient to meet the needs of a self-sustaining exposure group, and that if the exposure group was assumed to be dependent on extensive agriculture methods, then a much larger area would be needed. The selected area may therefore be described as cautious but realistic.

7.1.5 Calculation of spatially averaged activity concentrations

To assess the extent of population exposure to radionuclides within the assessment area, the following four sets of spatial averaging were considered, in order to investigate the implications of the alternatives:

- A human averaging assumption (2.00E04 m^2), selected from discussion in Section 7.1.4
- The minimum home range for each representative species
- The maximum home range for each representative species
- The home range sizes appropriate for each of the SPACE reference groups presented in Chapter 6 (small - 0.5 km^2, medium - 5 km^2 and large - 10 km^2).

Using GIS, 1000 points were randomly selected within the assessment area for each of the spatial averaging approaches to be considered. For each averaging approach, a spatial average activity concentration was determined by creating a buffer (radius 0.5 to 25000 m depending on the spatial averaging approach being used) centred on each randomly selected point and calculating the arithmetic mean activity concentration across the spatial averaging area covered by the buffer (see Figures 7-4 to 7-6 for examples of this).

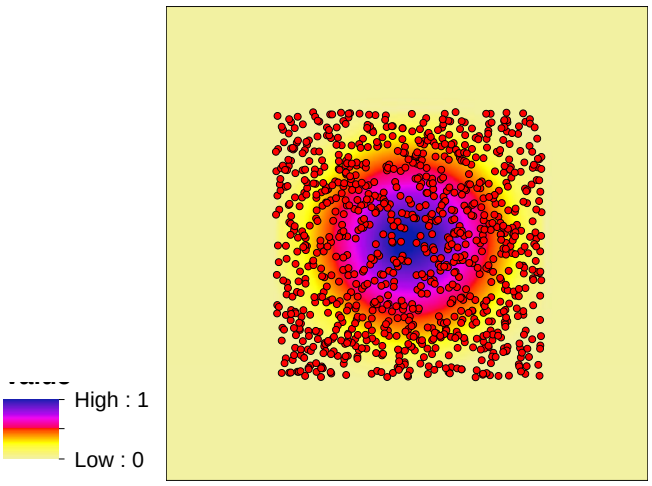


Figure 7-4. One thousand random sampling points for quantifying average activity concentrations for rabbit populations in relation to radionuclides with a dispersion plume of 10 km radius.

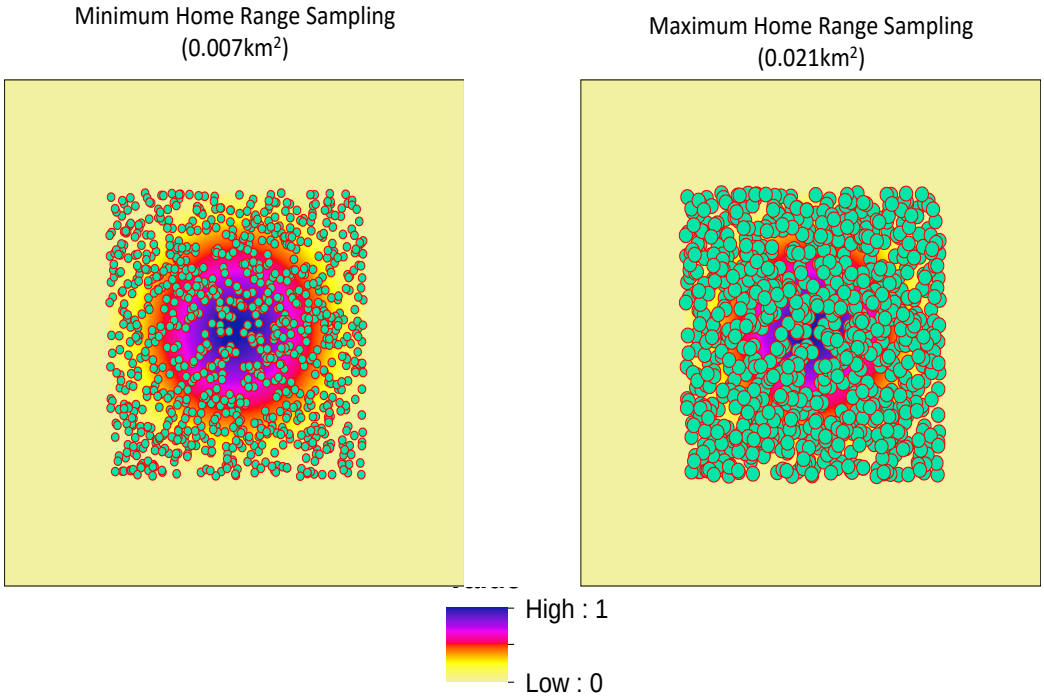


Figure 7-5. Application of minimum and maximum population range buffers to the one thousand random sampling points for rabbit populations in relation to radionuclides with a dispersion plume of 10 km radius.

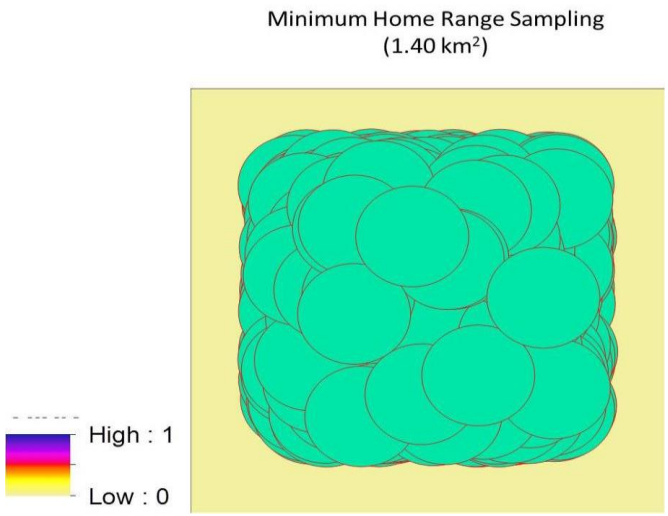


Figure 7-6. Application of red fox minimum population range buffers to the one thousand random sampling points in relation to radionuclides with a dispersion plume of 10 km radius. This resulted in coverage of the entire exposure assessment area.

Each of the randomly selected points can be viewed as representing a sampling replicate for the population under assessment. Home ranges were multiplied by Hope's [2005] scaling factor of 40 to give an estimate of population range. Therefore, the overall arithmetic mean activity concentration for a particular spatial averaging scenario was calculated from the results for each of the 1000 sampling points. The results are presented in Tables 7-1 and 7-2. The gray wolf was excluded because the estimated population range size was much larger than the assessment area that was used within the GIS. Also, only one static species was considered (Fescue grass) because no 'home range' data were located for the other static species during the literature review process (see Chapter 6).

Table 7-1. Spatially averaged soil activity concentrations (Bq kg^{-1}) for each radionuclide mobility category and each of the other averaging approaches (SPACE reference groups and human spatial scales).

	Radionuclide mobility		
Spatial averaging approach	**Low**	**Medium**	**High**
'Small' SPACE group	0.006	0.291	0.778
'Medium' SPACE group	0.006	0.292	0.680
'Large' SPACE group	0.005	0.232	0.542
Human[a]	0.009	0.322	0.769

[a] Results based on an averaging scale of 2.00E04 m^2 (see section 7.1.4).

Within Table 7-2, there are some instances where the activity concentrations for the minimum and maximum population range estimates appear to be the same (e.g. common lizard). In these cases, only a single home range size estimate was present in the data collated on SPACE representative species (see section 6.2.1) so this has been used as both the minimum and the maximum estimate.

7.2 DOSE ASSESSMENT

The dose assessment work was conducted using version 1.0 of the ERICA assessment tool, which was developed to facilitate application of the ERICA integrated approach for assessing radiological risk to terrestrial, freshwater and marine biota [Brown et al., 2008]. Further information on the ERICA assessment approach is provided in Appendix A.

ERICA can be used to calculate both internal and external doses, as well as whole body dose. Media concentrations can be entered into the tool and concentration ratios are used to calculate the concentration of specific radionuclides within the body of an organism. Dose conversion coefficients (DCCs) for external exposure are calculated from media concentrations and consider the amount of time an organism spends in different media, for example soil, air, and water. Internal DCCs calculate an absorbed dose rate based on the internal concentration of radionuclides derived from the concentration ratios.

Table 7-2. Spatially averaged soil activity concentrations (Bq kg^{-1}) for each radionuclide mobility category and SPACE representative species combination.

Population range estimate	Minimum			Maximum		
Radionuclide mobility	**Low**	**Medium**	**High**	**Low**	**Medium**	**High**
Adder	0.008	0.339	0.793	0.009	0.346	0.799
American mink	0.009	0.329	0.780	0.009	0.329	0.779
Badger	0.006	0.283	0.663	0.006	0.242	0.565
Bank vole	0.010	0.357	0.814	0.010	0.357	0.814
Black garden ant	0.007	0.309	0.760	0.007	0.309	0.754
Brown rat	0.008	0.317	0.771	0.008	0.318	0.766
Common earthworm	0.010	0.355	0.801	0.010	0.355	0.796
Common frog	0.010	0.357	0.814	0.010	0.357	0.814
Common honey bee	0.007	0.324	0.775	0.007	0.324	0.775
Common lizard	0.012	0.329	0.789	0.012	0.329	0.789
Common toad	0.009	0.344	0.789	0.009	0.344	0.788
Feral pigeon	0.006	0.353	0.801	0.006	0.323	0.798
Fescue grass	0.006	0.323	0.773	0.006	0.353	0.767
Great crested newt	0.012	0.316	0.782	0.012	0.316	0.782
greylag goose	0.005	0.206	0.749	0.005	0.235	0.548
Mallard duck	0.006	0.206	0.749	0.002	0.107	0.277
Moose	0.005	0.206	0.489	0.002	0.115	0.296
Pheasant	0.009	0.315	0.739	0.009	0.315	0.680
Rabbit	0.007	0.314	0.770	0.007	0.314	0.769
red fox	0.006	0.289	0.738	0.001	0.060	0.156
Roe deer	0.008	0.334	0.813	0.005	0.237	0.557
Water vole	0.008	0.335	0.805	0.009	0.335	0.743
White-tailed deer	0.006	0.304	0.780	0.006	0.237	0.552
Willow Ptarmigan	0.006	0.353	0.801	0.006	0.353	0.801

7.2.1 Input parameters

The input parameters used in the critical evaluation of spatial scale assumptions on biota dose rates for a hypothetical release scenario are described below. Data in support of the assessment are presented in Appendix C.

Organism dimensions

Organisms used in the ERICA tool are represented as an ellipsoid which allows for the calculation of DCCs. Data on body length, width and height of each organism, or ellipsoid, had to be sourced. The Wood et al. [2009] paper and Posiva report [2014b] provided organism geometries and other ecological data for approximately half of the species from the SPACE species list. An additional literature review was undertaken to establish geometry dimensions for those organisms not covered by these two sources. Certain dimensions were difficult to source for some species. For example, the height of an organism often included the legs. Where this was the case, dimensions were estimated from photographs when other dimensions were known. Whole body mass was likely to include the legs and head of each organism. Body masses sourced from Posiva [2014b] were determined using the formula for the volume of an ellipsoid:

$$V = (4\pi \times a \times b \times c)/3$$

ERICA tool default organism geometries were used in the absence of suitable data. The collated organism dimension data are presented in Appendix C (Table C-1).

Occupancy factors

Occupancy factors account for the fraction of time that an organism spends in different parts of the vertical profile of the environment. For example, in the terrestrial environment, some burrowing mammals can spend 50% of their time underground and 50% on the soil surface. As with the organism dimensions, occupancy factors were derived from radioecological literature [Wood et al., 2009; Posiva, 2014b] or estimated using expert judgement from each individual species' ecology. Airborne insects (Common honey bee) were classified as ground living organisms spending 100% of their time on the ground so the assessment results for these were conservative. Burrowing organisms such as rabbits were classified as spending 100% of their time in soil so the results obtained for these organisms were also conservative. The occupancy factors used (Appendix C, Table C-1) summed to a value of 1, as organisms were assumed spend their entire time in the study area.

Concentration ratios

Concentration ratios are transfer parameters that relate the concentration of a radionuclide in an environmental media to whole body concentrations within an organism [Howard et al., 2013]. They are calculated using the following equation:

$$CR = \frac{activity\ concentration\ in\ biota\ whole\ body\ (Bq\ kg - 1\ fresh\ weight)}{activity\ concentration\ in\ soil\ (Bq\ kg - 1\ dry\ weight)}$$

For the SPACE study, appropriate concentrations ratios were sourced primarily from the Wildlife Transfer Database [Copplestone et al., 2013]. The concentration ratios used are presented in Appendix C (Table C-2).

7.2.2 ERICA tool application

The organism dimensions were used to create 'new' organisms within the ERICA tool. Organisms were entered into the organism wizard using the most appropriate classification. For example, the amphibian reference organism was used to represent the common frog. All species were classified as 'ground living organisms' in order to provide the most conservative assessments. For some species such as the black garden ant, their masses were below the weights of the default ERICA tool reference organism geometries and as a result are outwith the extrapolation capability of the Dose Conversion Coefficient (DCC) calculator within ERICA. Where this occurred the smallest default mass was used to allow the DCC calculator to extrapolate the DCCs for the new organism geometry. This led to, for example, the ant mass increasing by an order of magnitude. However, previous work has demonstrated that, except for very small unicellular organisms or very large organisms, changing the mass/size within the ERICA tool has only a small effect on the internal dose received by the organism.

Changing the density of an organism was tested to establish how this would affect the DCCs for the SPACE representative organisms. Three masses were selected for a hypothetical organism (5x5x5 cm): 0.5 kg, 5kg and 50 kg. The DCCs for four selected radionuclides are reported in Tables 7-3 to 7-5. The % increase is reported for the difference between the highest and lowest value. Changing the organism's density has the greatest effect for gamma emitters (approximately a 60% difference in DCC across the mass range evaluated). However, for the case of the ant, changing from 8E-5 to 8E-4 is likely to have very little effect on internal DCCs and therefore the overall dose rate to the organism.

Table 7-3. Internal DCCs for Beta/gamma emitters

Radionuclide	0.5kg	5kg	50kg	% increase
Am-241	3.15E-05	3.47E-05	3.47E-05	10.16%
Cs-137	1.79E-04	2.20E-04	2.89E-04	61.45%
Cl-36	1.56E-04	1.57E-04	1.57E-04	0.00%
Pu-239	3.08E-06	1.18E-06	3.14E-06	1.95%

Table 7-4. Internal DCCs for low energy beta emitters

Radionuclide	0.5kg	5kg	50kg	% increase
Am-241	5.73E-06	5.73E-06	5.73E-06	0.00%
Cs-137	3.71E-07	3.71E-07	3.71E-07	0.00%
Cl-36	6.91E-08	6.91E-08	6.91E-08	0.00%
Pu-239	1.18E-06	1.18E-06	1.18E-06	0.00%

Table 7-5. Internal DCCs for alpha emitters

Radionuclide	0.5kg	5kg	50kg	% increase
Am-241	3.16E-03	3.16E-03	3.16E-03	0.00%
Cs-137	0	0	0	0.00%
Cl-36	0	0	0	0.00%
Pu-239	2.97E-03	2.97E-03	2.97E-03	0.00%

An ERICA Tier 2 terrestrial assessment was run to calculate doses for all SPACE representative species, assuming an activity concentration input for each radionuclide of 1 Bg kg^{-1}. The doses calculated by ERICA were therefore equivalent to a dose rate per unit of activity (µGy h^{-1} / Bq kg^{-1}). The results were then multiplied by the appropriate spatially averaged soil activity concentrations presented in Table 7-1.

7.3 RESULTS

The results of the dose assessment calculations are presented in Figures 7-7 to 7-15. The figures have been grouped according to the radionuclide environmental mobility classification used in the GIS simulation (High: C-14, Cl-36, I-129, Tc-99 (Figures 7-7 and 7-8); Medium: Cs-137, Np-237, Pb-210, Po-210, Ra-226, Se-79 (Figures 7-9 to 7-11); and Low: Th-229, Th-230, Th-232, U-233, U-234, U-235, U-236, U238 (Figures 7-12 to 7-15). Within these groupings, the figures are ordered alphabetically by element and then by isotope number when results for more than one isotope of a given element are presented. It should be noted that the data presented are the weighted[h] dose rates (µGy h^{-1}) for 1 Bq kg^{-1} released to surface soil. Therefore the results provide a relative scaling of doses rather than absolute doses for a genuine source term.

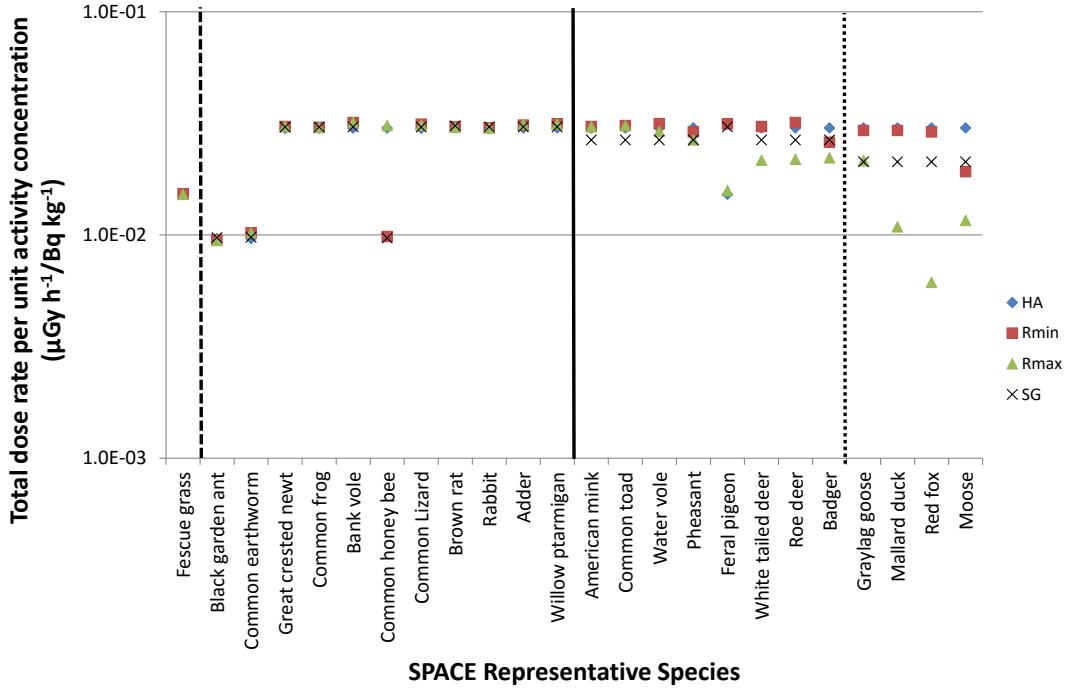
Each figure is presented in a standardised format to facilitate comparison between figures, although the Y-axis has been scaled as required to ensure that the data are displayed clearly. Four data series are presented on each figure:

- Human Averaging (HA) – The dose rate calculated using a typical human averaging assumption.
- Minimum home range (Rmin) – The dose rate calculated using the minimum home range for each representative species as the basis for spatial averaging.
- Maximum home range (Rmax) – The dose rate calculated using the maximum home range for each representative species as the basis for spatial averaging.
- SPACE Reference Group (SG) – The dose rate calculated using the proposed SPACE group dose rate ranges for representative species using the appropriate home range category (small - 0.5 km^2, medium - 5 km^2 and large - 10 km^2) scale as the basis for spatial averaging.

Three vertical lines can be seen on each of the figures; these represent the boundaries between each of the SPACE reference groups. To the left of the dashed line is Fescue grass, which is the one representative of the immobile SPACE reference group that was modelled. Between the dashed line and the solid black line are the organisms within the 'small' home range category. Between the solid black line and the dotted line are the 'medium' home range organisms. Organisms to the right of the dotted line are in the 'large' home range category.

[h] ERICA can be used to calculate weighted or unweighted dose rates. For weighted dose rates, the default values representing Relative Biological Effectiveness are 10 for alpha radiation, 3 for low energy beta and 1 for high energy beta and gamma radiation.

(a) C-14



(b) Cl-36

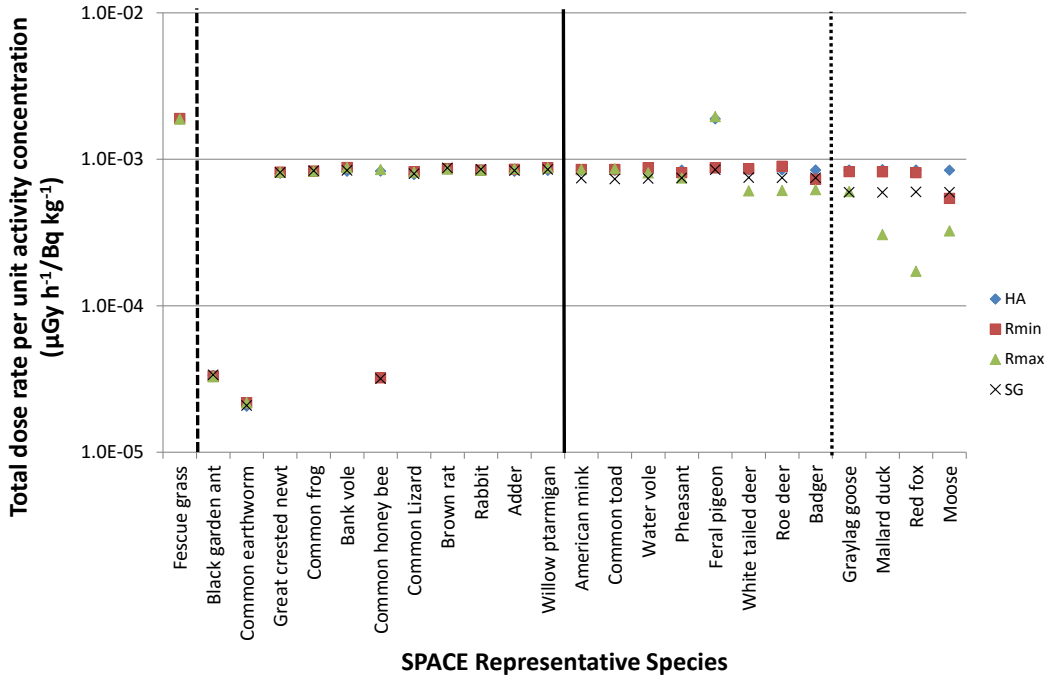
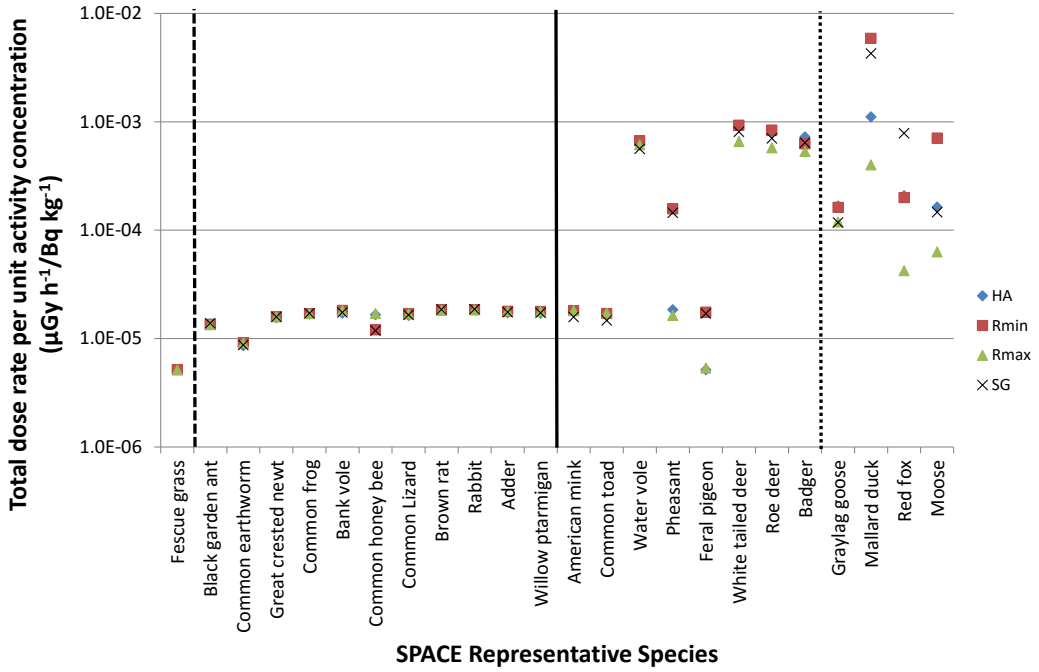


Figure 7-7. Total dose rates from (a) C-14 and (b) Cl-36 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}). Data for four spatial averaging approaches are presented: Human Averaging (HA), Minimum home range (Rmin), Maximum home range (Rmax), SPACE Reference Group (SG) (see text for details).

(a) I-129



(b) Tc-99

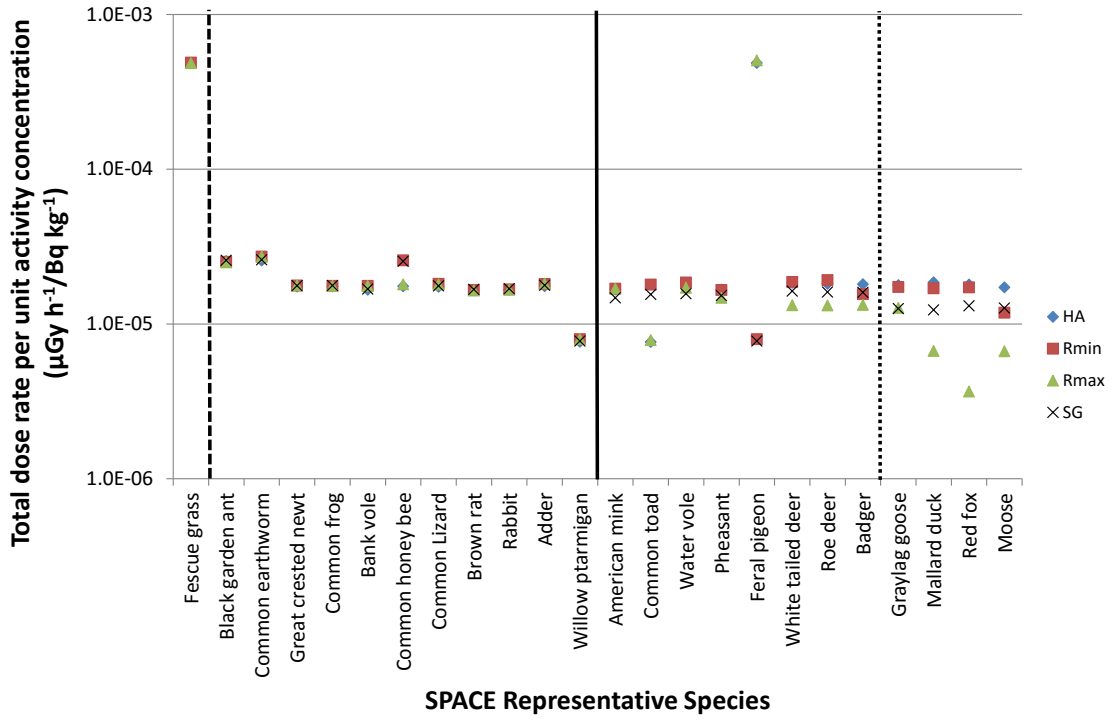
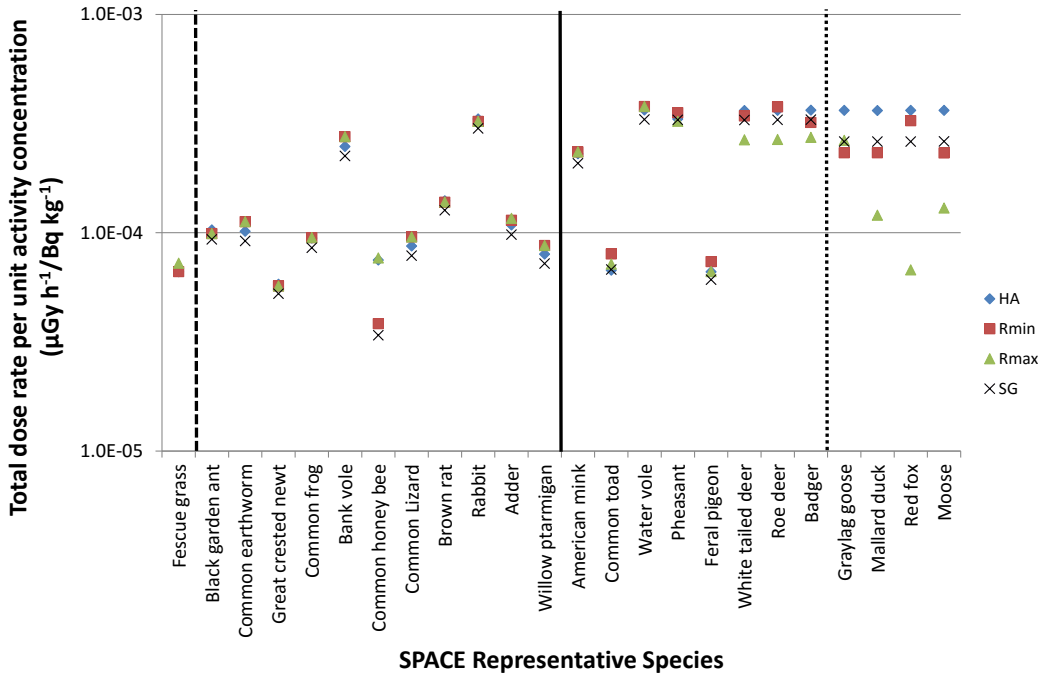


Figure 7-8. Total dose rates from (a) I-129 and (b) Tc-99 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}). Data for four spatial averaging approaches are presented: Human Averaging (HA), Minimum home range (Rmin), Maximum home range (Rmax), SPACE Reference Group (SG) (see text for details).

(a) Cs-137



(b) Np-237

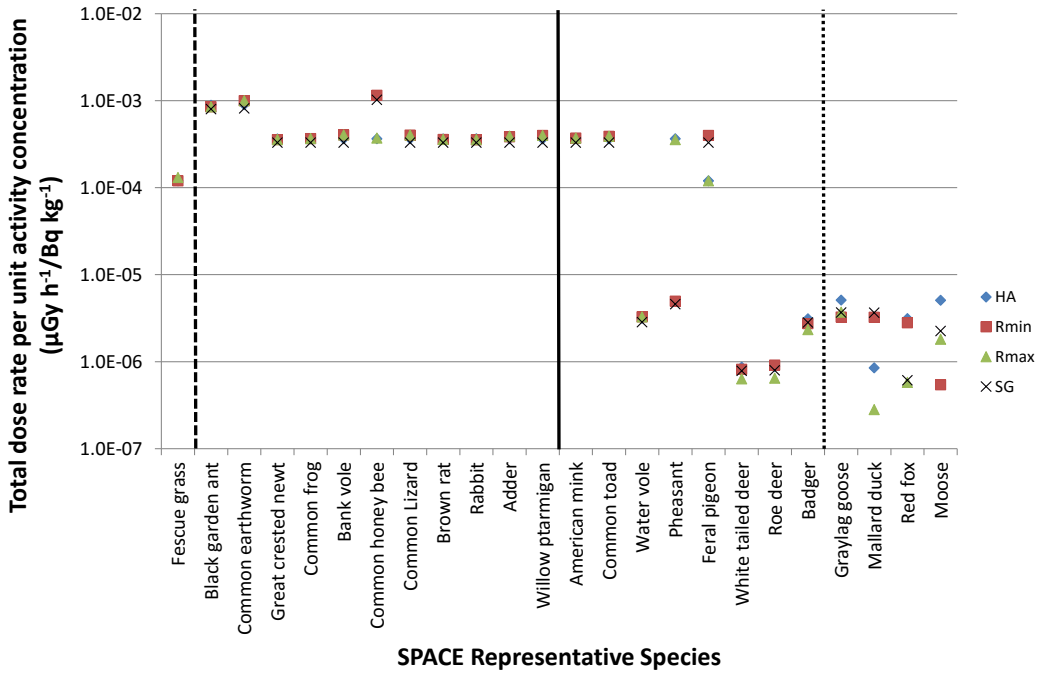
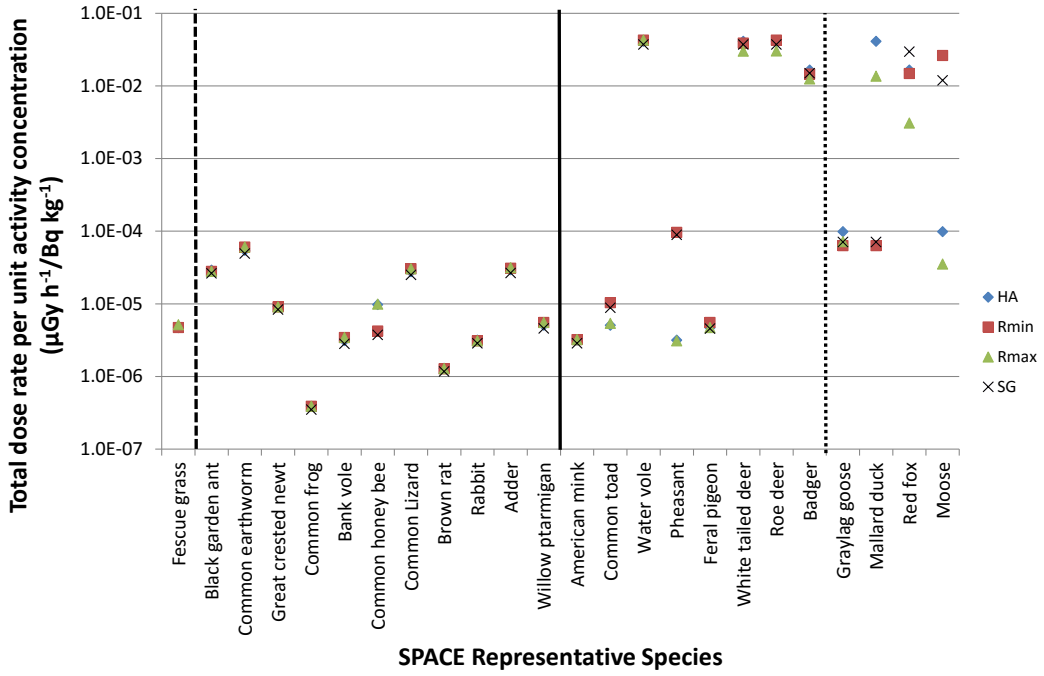


Figure 7-9. Total dose rates from (a) Cs-137 and (b) Np-237 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}). Data for four spatial averaging approaches are presented: Human Averaging (HA), Minimum home range (Rmin), Maximum home range (Rmax), SPACE Reference Group (SG) (see text for details).

(a) Pb-210



(b) Po-210

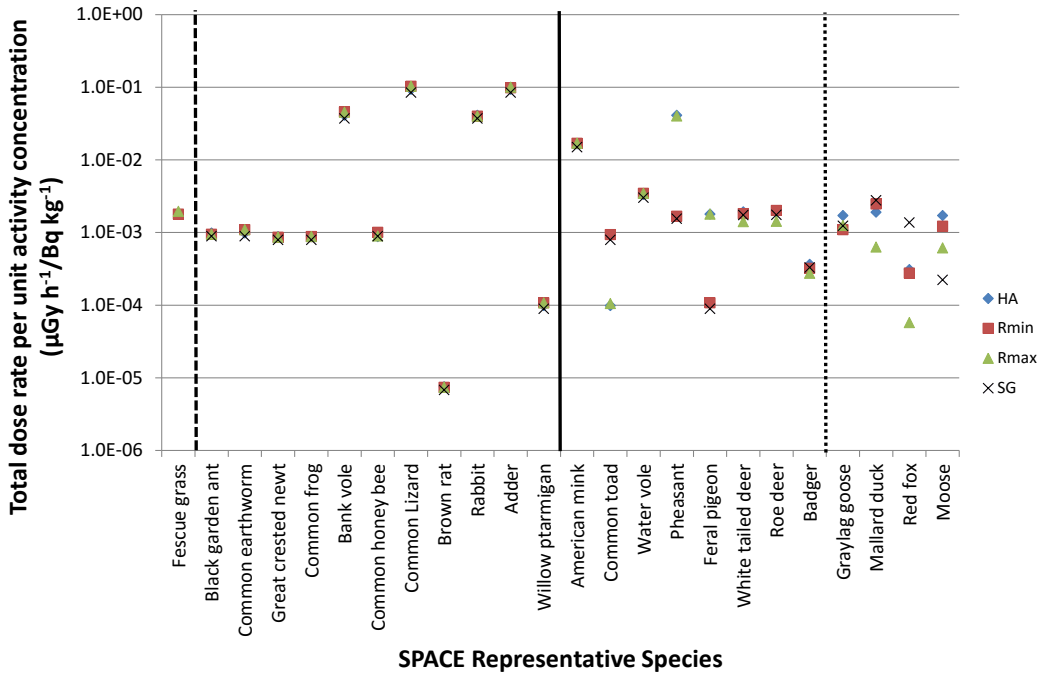
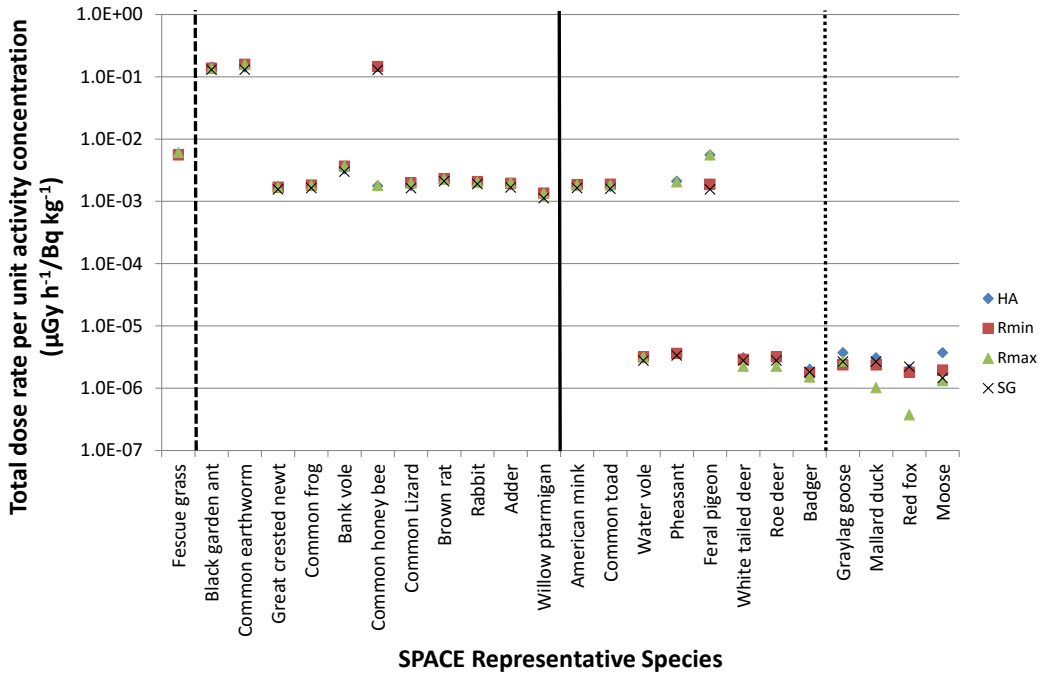


Figure 7-10. Total dose rates from (a) Pb-210 and (b) Po-210 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}). Data for four spatial averaging approaches are presented: Human Averaging (HA), Minimum home range (Rmin), Maximum home range (Rmax), SPACE Reference Group (SG) (see text for details).

(a) Ra-226



(b) Se-79

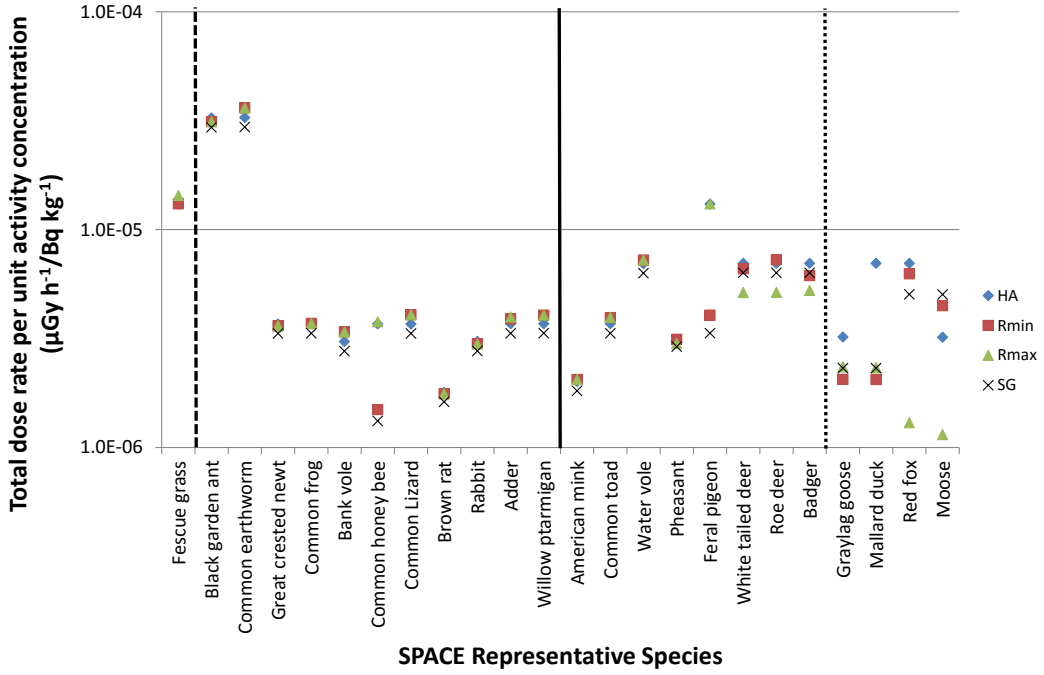
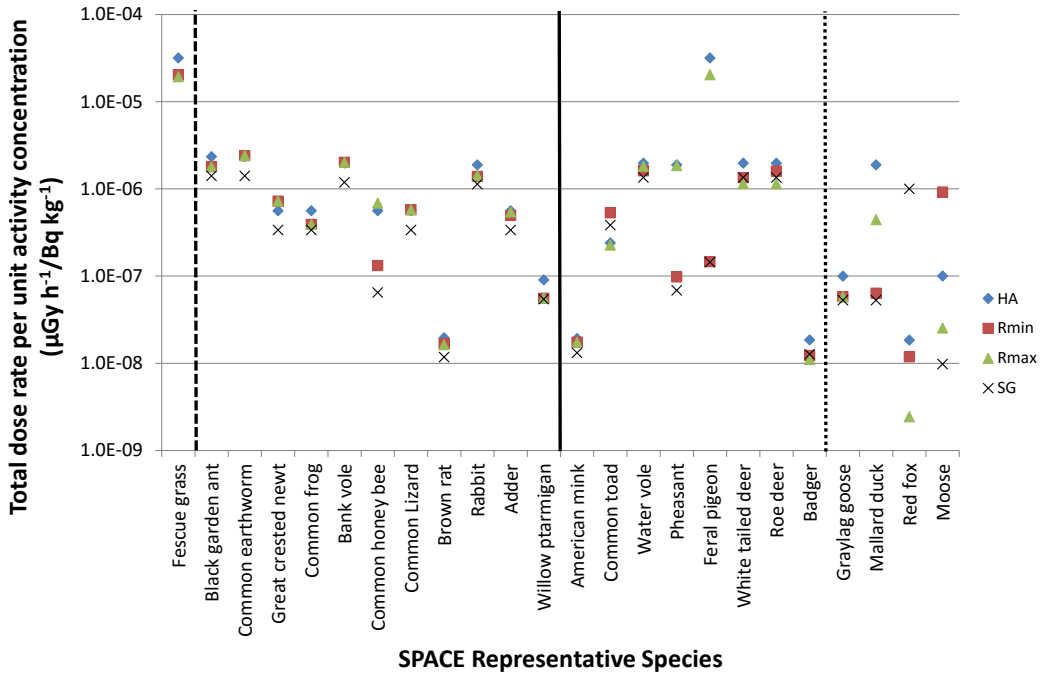


Figure 7-11. Total dose rates from (a) Ra-226 and (b) Se-79 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}). Data for four spatial averaging approaches are presented: Human Averaging (HA), Minimum home range (Rmin), Maximum home range (Rmax), SPACE Reference Group (SG) (see text for details).

(a) Th-230



(b) U-238

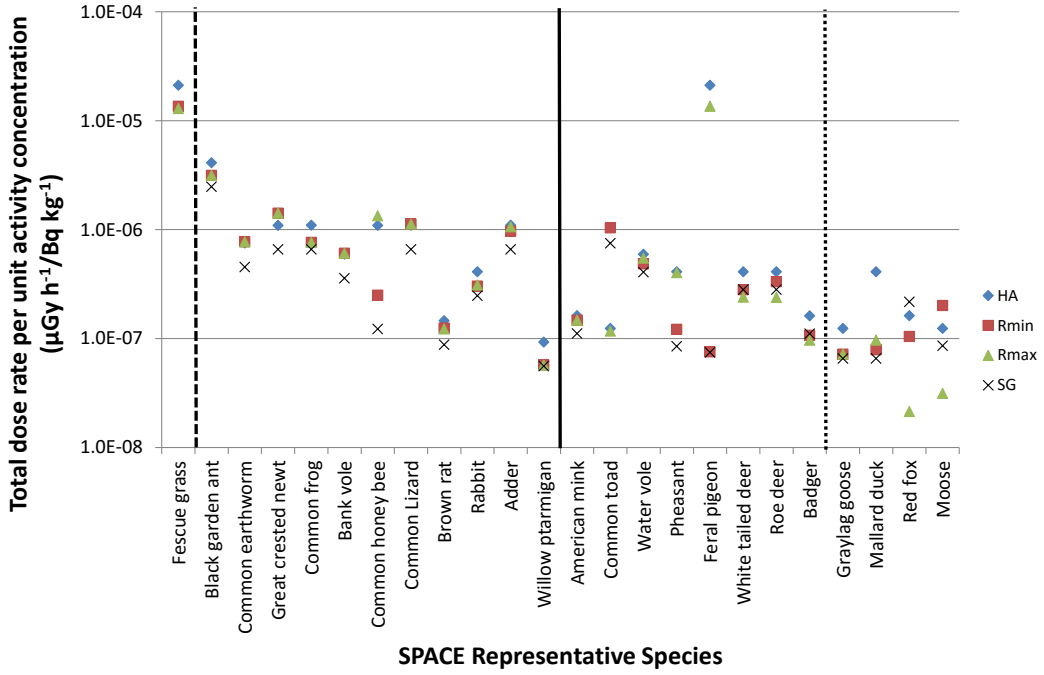


Figure 7-12. Total dose rates from (a) Th-230 and (b) U-238 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}). Data for four spatial averaging approaches are presented: Human Averaging (HA), Minimum home range (Rmin), Maximum home range (Rmax), SPACE Reference Group (SG) (see text for details).

7.4 DISCUSSION

Considering each of the SPACE reference groups, there is no apparent trend in variability in the dose rate predictions for individual SPACE representative species based on radionuclide environmental mobility categories. This may be because the assessment area was relatively large in comparison with the contaminated region.

Results for each radionuclide are discussed below, in relation to radionuclide mobility categories.

7.4.1 High mobility radionuclides

Radionuclides classified as high mobility for the purposes of the assessment were C-14, Cl-36, I-129 and Tc-99.

Carbon-14

For all reference groups, use of the human averaging approach would be considered conservative (Figure 7-7a). For the majority of the SPACE representative species, dose rates calculated are similar in magnitude among all spatial scale assumptions with differences only being observed for medium- and large-ranging species. Dose rates calculated for such species tended to be lower when using maximum species ranges and SPACE reference group ranges as compared with human averaging assumptions. All dose rates were nonetheless within one order of magnitude.

Chlorine-36

With the exception of common honey bee for which use of SPACE reference group and minimum species ranges resulted in lower dose rates than maximum species range and human averaging assumptions, dose rates for representative species with low spatial ranges were similar for all spatial scale assumptions (Figure 7-7b). As species spatial ranges increase, variability in dose rate also increases, but dose rates are all within one order of magnitude. Use of a human averaging approach was conservative in all cases. Little variation in dose rates was observed between representative species with a low spatial range.

Iodine-129

Variation in dose rates for representative species was more pronounced for the different spatial scale assumptions for I-129 than for C-14 or Cl-36. Little variation was observed for species with small spatial ranges, but was highly variable for the more mobile species with dose rates varying by more than an order of magnitude for species with large spatial ranges when different spatial scale assumptions were applied (Figure 7-8a). The greatest variability was observed for mallard duck, red fox and moose. Use of a human averaging approach was not shown to be conservative for these representative species; the use of a SPACE reference group spatial scale was more conservative for these species and was no less conservative than the human averaging scale for all other representative species.

Technetium-99

Variation in dose rates calculated using the different spatial scale assumptions was less pronounced for Tc-99 than for I-129 (Figure 7-8b). The use of the human averaging approach was again conservative, with the exception of one representative species, common toad, for which use of the minimum species and SPACE reference group scales were found to be more conservative. Differences across all spatial scales were within 1 order of magnitude however for all representative species. The greater variation in calculated dose rates was again for the more widely ranging species will little variation being observed for species with a small spatial range. For species with a larger spatial range,

use of the maximum spatial range for each species resulted in a lower dose rate than use of the human averaging approach whereas minimum range for each representative species or use of the SPACE reference group scale resulted in dose rates of similar magnitude to those from the human averaging approach.

7.4.2 Medium mobility radionuclides

Radionuclides classified as having a medium level of mobility in the environment were Cs-137, Np-237, Pb-210, Po-210, Ra-226 and Se-79. Results are discussed for each in the following sections.

Caesium-137

Variation in dose rates was more pronounced for representative species with low spatial scales than was observed for radionuclides classified as having a high environmental mobility, and variation in dose rates between representative species was evident across all spatial range categories (Figure 7-9a). One of the drivers behind the variability observed may be the database of concentration ratios, which is extensive for a wide range of organisms for Cs-137 as opposed to other radionuclides, largely as a result of research following the Chernobyl accident. Whilst variation was observed in dose rates between representative species, the spatial scale assumptions applied for each species resulted in little observed difference with the exception of those species with large ranging habits. In these instances, human averaging assumptions were found to be more conservative than other species or group specific scale assumptions with maximum range assumptions being the least conservative. Dose rates were nonetheless all within an order of magnitude, irrespective of the spatial scale assumptions applied.

Neptunium-237

No difference was observed in dose rates for representative species with the various spatial scale assumptions applied for those species with small spatial scales with the exception of common honey bee for which use of the maximum species range or SPACE reference group scale was found to be more conservative (Figure 7-9b). This was also the case for the majority of medium spatial scale ranging species although for pheasant and feral pigeon some differences were observed. This was most pronounced for pheasant where the species maximum spatial scale and human averaging approach results were around two orders of magnitude greater than those for the minimum species and representative group spatial scales. Overall there were three instances where use of a human averaging approach was not conservative: common honey bee, feral pigeon and mallard duck. For representative species with a large spatial range, no trend is evident with regard to the influence of spatial scale assumptions on dose rates.

Lead-210

As with Cs-137, dose rates for representative species were highly variable across all spatial scale groupings (Figure 7-10a), which is again likely due to variation in concentration ratios between different species. Dose rates for each representative species calculated using the different spatial scale assumptions were largely similar however, with the exception of pheasant, common toad, common honey bee, mallard duck, red fox and moose. The degree to which the human averaging approach was conservative was also variable; for pheasant, red fox and moose, alternative spatial scale assumptions resulted in the calculation of higher dose rates. In the case of moose, differences in dose rates were more than two orders of magnitude higher using the minimum representative spatial scale and SPACE reference group scale as compared with the use of the human averaging approach.

Polonium-210

Dose rates from exposure to Po-210 were also found to be variable (Figure 7-10b). However, for representative species with a small spatial range, no variation in dose rate was observed with the different spatial scale assumptions. Similarly, little variation was observed for those representative species with a medium spatial range with some exceptions (pheasant, common toad and feral pigeon). The degree to which the different spatial scale assumptions affected calculated dose rates was again variable. In the case of pheasant and feral pigeon, use of human average or representative species maximum spatial scales gave rise to the highest dose rates whereas for common toad, use of either the minimum representative spatial scale or SPACE reference group scale gave rise to higher dose rates. For these representative species, differences in dose rates of around an order of magnitude are observed between the different spatial scale assumptions. Dose rates from the different spatial scale assumptions were also variable for representative species with a large spatial range. The use of a SPACE reference group scale was similar or slightly more conservative than a human averaging approach for three of the four species within this category whereas the human averaging approach was more conservative for the moose representative species.

Radium-226

Dose rates for representative species as a result of exposure to Ra-226 fall into two distinct categories with low ranging species having higher dose rates than those with larger ranges (Figure 7-11a). Little difference in dose rate is evident for most species with the different spatial scale assumptions, with two exceptions – common honey bee and pheasant, which each display different responses to changing spatial scale assumptions. The former, common honey bee, has a greater dose rate associated with the use of the SPACE reference group and minimum species scales of assessment as compared with the use of the maximum species or human averaging scales with differences of around two orders of magnitude arising. The opposite trend is however observed for pheasant with more than two orders of magnitude difference in dose rates being observed using the human averaging and maximum representative species scales as compared against the minimum representative species and SPACE reference group spatial scales.

Selenium-79

Dose rates for representative species as a result of exposure to Se-79 were all within around one order of magnitude (Figure 7-11b), irrespective of the spatial scale assumptions employed. The largest variation in dose rate from the different spatial scale assumptions was observed for those representative species with the greatest spatial scales – red fox and moose. In both cases, use of a maximum species spatial scale resulted in the lowest dose rates. Little variation was observed between the other spatial scales employed.

7.4.3 Low mobility radionuclides

Results of the two low-mobility radionuclides, Th-230 and U-238, are discussed below. Results for thorium and uranium isotopes are presented in Appendix C. Isotopes of the same element (e.g. U) are all assigned the same CR within international data compilations such as the Wildlife Transfer Database[i]. Therefore, for U and Th isotopes, the dose rate variation observed between different averaging assumptions for a given SPACE representative species were comparable for each isotope. However,

[i] www.wildlifetransferdatabase.org

the absolute value of the dose rates reported was seen to vary between isotopes. This was due to differences in the dose conversion coefficients (DCCs) for each isotope; the DCC differences reflecting differences in the radioactive emissions of each isotope.

Thorium-230

Dose rates resulting from the exposure of representative species to Th-230 are highly variable (Figure 7-12a) and no clear trend is observed with regard to spatial scale assumptions, spatial ranges of the representative species and resultant dose rates. However, use of the human averaging approach was found to be conservative, or consistent, in most instances with regard to spatial scale assumptions, the exceptions being red fox and moose for which the SPACE reference group and representative species minimum scales were the more conservative, respectively.

Uranium-238

Dose rate results for U-238 were similar to Th-230 with the human averaging approach being conservative or consistent with respect to SPACE reference group or representative species scales of assessment. Red fox and moose were again exceptions although the difference in dose rates was less pronounced than observed for Th-230. A greater variation in dose rate under the different spatial scales was observed however for common toad with use of the representative species minimum and SPACE reference group scales being more conservative than use of the human averaging or maximum representative species scales.

7.5 RESULTS IN THE CONTEXT OF SPATIAL SCALES FOR THE PROTECTION OF POPULATIONS IN LONG-TERM DOSE ASSESSMENTS

In general, the application of the human averaging assumption resulted in a calculated dose rate that was within 10% of the maximum dose rate reported by the different averaging methods for a given radionuclide. Out of the 432 radionuclide-organism combinations evaluated, less than 15% showed a greater than 10% difference between the results from human averaging assumptions and those from other averaging methods. There were only 4 instances in which the human averaging method was found to result in a calculated dose rate that was more than an order of magnitude below one of the other averaging methods for the key assessed radionuclides. This greater than one order of magnitude difference was observed for Pb-210 (Pheasant and Moose), Ra-226 (Common honey bee) and Th-230 (Red fox).

Although these results are only being considered in the context of one simulated case study scenario, they do appear to lend confidence to the use of human averaging scales within biosphere assessments for radioactive waste disposal facilities where the aim is to demonstrate protection of populations. However, the human averaging approach presented here may differ from that used in specific biosphere assessments (see Chapter 2). For assessments where the biosphere around discharge zones is characterised, it may be useful to consider the information presented herein on population scales when considering model discretisation such that both people and biota utilising the area can be appropriately represented in assessment models.

As different approaches for human spatial averaging are adopted within particular biosphere assessments, it is recommended that the influence of these (and any evolving approaches for establishing NHB averaging scales) be further explored through application in specific biosphere assessments. Building on the findings of the SPACE study and drawing on the methodological approaches outlined within this report, the further evaluation of appropriate spatial scales for assessing NHB will increase stakeholder confidence in the assessment process.

8. CONCLUSIONS

The research presented in this report addresses the issue of averaging scales within long-term NHB dose assessments for radioactive waste disposal facilities. A critical review of international programmes and associated literature has allowed the rationale for addressing spatial and temporal scales within both human and NHB dose assessments to be evaluated (Chapters 2 – 4). In addition to information on spatial and temporal scales from biosphere assessments that have recently been undertaken (e.g. in Sweden and Finland), the review included consideration of relevant activities within the context of the ICRP (especially ICRP Committee 5) and within the Working Groups of the IAEA programmes (EMRAS, EMRAS II & MODARIA). MODARIA, in particular, has begun to consider spatial aspects of assessments through an evaluation of animal-environment interaction modelling and this is being extended within an international research programme (TREE). However, there is a general lack of guidance on how best to incorporate scale considerations into long-term assessments for NHB. This is particularly notable for population scales. Given that the protection goal for most NHB assessments is protection at the population level, the lack of guidance on how to address issues such as appropriate spatial averaging is surprising.

From an evaluation of long-term assessment approaches and critical review of life-history parameters for the range of 'SPACE representative species', it was concluded that, over the timescales for which long-term biosphere assessments are being undertaken, the temporal averaging resolution is unlikely to be significant when assessing doses to NHB. Therefore, although both spatial and temporal parameter data were reviewed for the selected 'SPACE representative species' and appropriate 'SPACE reference groups' established (Chapter 6), only spatial averaging considerations were included in the modelling work undertaken to evaluate the influence of scales (Chapter 7). Within the modelling work, the commensurability of NHB and human spatial scales was evaluated using a typical averaging scale for humans, which reflects assumed human utilisation of agriculturally managed ecosystems and their resources, which in turn have been based on actual typical human behaviour today in those ecosystems.

The results of the evaluation presented in this report suggest that the human averaging assumptions will provide conservative assessments of NHB doses to populations being considered within biosphere assessments. However, it should be noted that the human averaging scale utilised in the present assessment represented the lower range of spatial averaging applied in assessments to date (see Section 2.2) and overall conclusions may therefore vary according to the human averaging scale applied. It is recognised therefore that the initial evaluation presented here, whilst providing some confidence in the use of human averaging scales, could be further developed. For example, within future biosphere assessments, account could be taken of the land cover predictions (and hence the degree of habitat fragmentation). Linking this with ecological data for each species, a more direct evaluation of the spatial extent of the populations within the assessment area could be undertaken. Furthermore, the implications of using different human averaging assumptions on the conclusions drawn from the current assessment could be evaluated.

For the purposes of the SPACE analysis, population scales were estimated for all of the representative organisms using a generic scaling value of 40 [Hope, 2005]. Whilst this was appropriate for allowing the analysis to be performed, and also has some provenance within ecological risk assessment, it is unrealistic to think that all populations will scale in the same way. As observed in Chapter 6, a complex mix of environmental and ecological factors determine population scales. For the purposes of an initial evaluation of spatial scale influences, it was not appropriate to consider these in detail within the analysis presented here (beyond the evaluation of minimum and maximum home range sizes). Therefore, whilst the SPACE study provides a useful indication of the influence of spatial scale

assumptions within NHB dose assessments, there could be value in extending this evaluation to a range of real assessment situations in the future, and to consider variability in spatial scales under the different climate conditions that may arise over long-term assessment timeframes. To further develop the SPACE analysis, information on territoriality could be used, alongside to predictions of landscape change and habitat cover, to evaluate the extent of functional connectivity of habitats of relevance to SPACE representative organisms and hence determine the potential spatial ranges of populations that may be expected to be present. Such work would most effectively be focussed and carried out on a site-specific basis, rather than being based on generic considerations as in the current study.

A further future modelling study that could be undertaken would be to constrain the selection of the 1000 randomly sampled points used for spatial averaging to within the contaminated zone. This would reduce the uncontaminated area covered by the random sampling approach.

International guidance is provided (see Chapter 2), that human exposure groups should be characterised in terms relevant to the biosphere system that they live in, but no current similar recommendation is made for biota. With international and national legislation increasingly giving specific consideration to the protection of the environment from ionising radiation, there may be merit in giving due consideration to the utilisation of the biosphere by populations of plant and animal that may be exposed due to their potential occupancy in areas that may be affected by discharge zones concurrently to the consideration of human utilisation of the system. Consideration of the biosphere in terms of both people and biota at an early stage in assessments may help alleviate any concern that NHB assessments are undertaken as something of an afterthought to human dose assessments and ensure that model discretisation is appropriate to both human and biota dose evaluations.

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APPENDIX A: MODELS FOR BIOTA DOSE ASSESSMENT AND THEIR APPLICATION

Models that have been applied internationally for evaluating biota dose rates from exposure to ionising radiation are discussed below and their application in international test scenarios presented in relation to spatial and temporal scale considerations. The focus is on those models that are publicly available.

A.1 Models

Part of the work undertaken within the IAEA Working Groups has been the identification and intercomparison of available NHB dose assessment models [e.g. Beresford et al., 2009]. This presents the opportunity to evaluate the functionality of these models in terms of the ability to accommodate spatial and temporal scale considerations within the assessment process. Given that many of the models that have been included within the IAEA Working Group activities are under development and are not publicly available, this review focuses on the three that are publicly available for download and use by third parties, which are ERICA, R&D128/SP1a and RESRAD-BIOTA. These are the three models most widely used for the purposes of NHB dose assessment internationally.

ERICA

The ERICA Tool [Brown et al., 2008] was developed during the EC EURATOM funded project 'Environmental Risks from Ionising Contaminants: Assessment and Management' (ERICA) to support the application of the 'ERICA Integrated Approach' to assessing the impact of ionising radiation on ecosystems [Larsson, 2008]. Building on the outputs of the Framework for Assessment of Environmental Impact (FASSET) project, which was funded by the EC 5th Framework Programme [Larsson, 2004], the ERICA Tool was developed specifically to meet the needs of users in EC Member States. This was achieved through direct interaction with the end-user community from the outset of the tool development process and at regular intervals throughout the 3-year project [Zinger et al., 2008].

The Tool allows input of single activity concentrations for individual radionuclides, or probability density functions to represent activity concentration input data within the most advanced part of the Tool (Tier 3). The ERICA Tool uses equilibrium concentration ratios (CRs) to model the transfer of radionuclides to NHB. For terrestrial ecosystems, Beresford et al. [2008a] defined the CR value for a specific radionuclide (R) as:

$$\text{CR}=\frac{\text{Activity concentration of R in biota whole-body (Bq kg}^{-1}\text{ fwt)}}{\text{Activity concentration of R in soil (Bq kg}^{-1}\text{ dwt)}} \quad (1)$$

with the exception of chronic atmospheric releases of ^{3}H, ^{14}C, ^{32,33}P and ^{35}S, for which CR is defined as:

$$\text{CR}=\frac{\text{Activity concentration of } R \text{ in biota whole-body (Bq kg}^{-1}\text{ fwt)}}{\text{Activity concentration of } R \text{ in air (Bq m}^{-3}\text{)}} \quad (2)$$

For aquatic ecosystems, Hosseini et al. (2008) defined the CR as:

$$\text{CR}=\frac{\text{Activity concentration of } R \text{ in biota whole-body (Bq kg}^{-1}\text{ fwt)}}{\text{Activity concentration of } R \text{ in filtered water (Bq l}^{-1}\text{)}} \quad (3)$$

For both terrestrial and aquatic systems, media activity concentrations are the primary input data for the ERICA Tool.

The ERICA Tool has been revised on four occasions since its first release[j]. The latest version contains a number of significant developments. These include (i) the addition of eight radionuclides (Ba-140, Ca-45, Cr-51, Cf-252, Ir-192, La-140, Pa-231 & Zn-65) to make the Tool consistent with the developing ICRP RAP framework [ICRP, 2008; 2009]; and (ii) the incorporation of concentration ratio values from Wildlife Transfer Database [Copplestone et al., 2013; www.wildlifetransferdatabase.org/], which was compiled within the IAEA EMRAS programme to support the preparation of the IAEA Handbook of parameter values for predicting transfer to NHB [Howard et al., 2013; IAEA, 2014].

Even though the original release of the Tool and the subsequent revisions have been informed through extensive engagement with the end-user community, the needs of end-users who wish to consider spatial and temporal scales of assessment are not currently well catered for by the ERICA Tool[k]. There is also limited guidance within the help file on how to handle spatial or temporal data sets within the different tiers of the ERICA Tool. For the purposes of an assessment that properly addresses the endpoints of relevance, which have to take account of temporal or spatial scale issues, the Tool would either need to be run multiple times and the result averaged, as appropriate, to reflect the spatial and/or temporal averaging for each species under assessment or the input data would need to be manipulated in advance to identify 'appropriately averaged' media activity concentrations.

R&D128/Sp1a

The R&D128/Sp1a assessment tool [Copplestone et al., 2001; Copplestone et al., 2003] was developed for the Environment Agency of England and Wales to assist them in fulfilling their regulatory obligations under European and national legislation [EC, 1979, 1992; UK Parliament, 1981, 1994]. The development time for the model was less than 6 months and the scope of the model (in terms of the radionuclides and organisms for which parameters are provided) is therefore more limited than ERICA. It should also be noted that, although the model is publicly available, it was developed for a specific user (the Environment Agency) rather than the broad range of intended users that ERICA was developed for. The use of R&D128 has largely been replaced by ERICA, although it remains the only one of the freely available models that allows modelling of Ar and Kr isotopes and the 'version 2' release now incorporates the functionality to assess doses from Ar-41,Kr-85, Kr-88, Xe-131m and Xe-133 [Vives i Batlle et al., 2015].

As with ERICA, R&D128/SP1a also lacks the functionality to support temporal and spatial data input. For the purposes of an assessment that addresses temporal or spatial scale issues, the R&D128 spreadsheets would either need to be run multiple times and the result averaged as appropriate to reflect the spatial and/or temporal averaging for each species under assessment or the input data would need to be manipulated in advance to identify 'appropriately averaged' media activity concentrations.

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[j] The latest release (version 1.2 update) was released in November 2014 and is available to download from http://www.erica-tool.com/

[k] The possibility of adding spatial and temporal scale functionality to the ERICA Tool has been discussed at various international meetings, including within the IAEA EMRAS/EMRAS II/MODARIA programme, so it is possible that this functionality will be included in future release.

RESRAD-BIOTA

The RESRAD family of model codes developed by Argonne National Laboratory in the United States includes a software tool specifically developed for the assessment of radiation impact on NHB. This tool, RESRAD-BIOTA [USDoE, 2004; https://web.evs.anl.gov/resrad/home2/biota.cfm], implements the United States Department of Energy (USDoE) graded approach for evaluating radiation doses to aquatic and terrestrial biota [USDoE, 2002]. It is an evolution of the BGC (Biological Concentration Guide) calculator that was originally developed to support the 'graded approach'. One of the main differences between RESRAD-BIOTA and the other two models considered above is the extent to which foodchain transfer can be modelled; RESRAD-BIOTA enables the application of a kinetic-allometric approach to modelling transfer [Higley et al., 2003], with associated modelling of transfer from the diet of animals rather than relying solely on the application of equilibrium transfer parameters. As with R&D128/SP1a, the model was developed for a specific user (the USDoE) and has subsequently been made publicly available, with training courses being offered to those wishing to use the model for undertaking assessments.

There have been four versions of the RESRAD-BIOTA to date. The latest version (RESRAD-BIOTA 1.5) incorporates probabilistic functionality, enabling sensitivity analysis to be performed on most of the modelling parameters. However, with regard to spatial and temporal assessment functionality, RESRAD-BIOTA has similar constraints to those observed for ERICA and R&D128/SP1a, namely that the tool is not specifically designed to receive spatial and/or temporal data sets as inputs to the modelling process. There is some consideration of spatial utilisation by organisms under assessment, which is reflected in the use of an 'area factor' (AF) within the assessment. Effectively, this defines the proportion of time that the organism spends within the assessment area. If AF<1, to obtain a comprehensive estimate of organisms exposure, it would be necessary to undertake assessments for each area that an organism utilises and sum the results.

A.2 Assessment scenarios

As indicated above, the various models that have been considered within the IAEA programmes have mainly been evaluated through the use of intercomparison exercises. The starting point for each of these exercises was the development of a modelling scenario. Each model was then used, by both model developers and 'informed users' [Wood et al. 2009], to make predictions (normally both activity concentrations and dose estimates) based on the scenario data and, where possible, the results were compared with measured data.

Chernobyl

The Chernobyl scenario that was used within the IAEA EMRAS programme [IAEA, 2012b] was developed from the scenario that had previously been used to test the application of the ERICA Tool within the context of the ERICA project [Beresford et al., 2008b]. The model input data available were soil activity concentrations derived from Geographical Information System (GIS) maps of soil contamination within the zone based. For small mammals, the derivation of the soil activity concentration values was based on assumptions of animal home range, thereby incorporating an element of spatial scaling. The small mammal soil activity concentration data were paired with dose measurements from thermoluminescent dosimeters (TLDs), which were attached to the small mammals using collars. These dose measurements provided one point of reference against which model predictions of external dose could be compared. Other measurement data available for model comparison were activity concentration data for NHB at the site.

Although there was the potential for undertaking a more spatially-focussed modelling approach to the assessment, the scenario only provided mean, minimum and maximum environmental activity concentration data for use by the modellers because the available models can specifically handle spatial dose assessments. Consequently, there was no consideration of spatial scales of assessment within the modelling work undertaken within the context of the IAEA EMRAS programme although this is now being considered within the latest IAEA sponsored programme, MODARIA.

Perch Lake

The second scenario that was used for model intercomparison within EMRAS was Perch Lake [IAEA, 2012b]. For this scenario, no spatially explicit data were available, but the activity concentration data available for use as model inputs spanned the period 1968 – 2004 [Yankovich et al., 2010]. Unfortunately, the radionuclides reported within this time period were not consistent: 1968 & 1971 for ^{90}Sr, ^{60}Co and ^{137}Cs; 1994 & 1998 for ^{90}Sr, ^{60}Co and ^{137}Cs; and 2003 & 2004 ^{3}H. There was also a lack of consistency in the biota data available over this time period. As a result, the model intercomparison focussed on model predictions for different time periods rather than utilising any form of temporal averaging.

Beaverlodge Lake

This scenario was used within the IAEA EMRAS II programme[l]. Natural radionuclide data (^{210}Pb, ^{210}Po, ^{226}Ra, ^{230}Th & ^{238}U) were available for specific bays around Beaverlodge Lake and also from surrounding lakes. Metal contamination data were also available. These sites had been contaminated as a result of U mining activities and the focus of the scenario was to undertake assessment for fish and invertebrate species at these sites. Again, no consideration of temporal or spatial scale was factored into the modelling. However, the scenario did lead to consideration of heterogeneous contaminant distributions within sediment and the influence of such distributions on the exposure of invertebrates.

Little Forest Burial Ground

Utilising this site in Australia as the basis for a scenario within IAEA EMRAS II provided an opportunity for model application in a very different terrestrial setting to that considered within the Chernobyl exclusion zone. Parameterisation of models had not drawn heavily on data from Australia and the Little Forest Burial Ground scenario included a requirement to assess transfer and doses for various Australian wildlife, including marsupials [Johansen et al., 2012]. The scenario did incorporate a consideration of the spatial distribution of the contamination at the site, both vertically and horizontally, and this provoked some interesting debate regarding strategies for modelling trees within the assessment process. However, this consideration was confined to individual trees, without any consideration of spatial averaging across a broader population.

Wetlands scenario

The final scenario undertaken within the IAEA EMRAS II programme was a wetlands scenario [Stark et al., 2015]. The justification for this scenario was that the organisms inhabiting wetlands are in an interface system with characteristics of both aquatic and terrestrial systems. Given that most models are not developed to specifically consider this type of system, the scenario was viewed as presenting some conceptual challenges for modelling. Unfortunately, data for individual wetlands were limited, so

[l] http://www-ns.iaea.org/projects/emras/emras2/working-groups/working-group-four.asp?s=8

a combination of data collected from Canada, the US and Sweden were used. This presented no opportunities for spatial and temporal scale considerations beyond the assumptions that modellers used when considering spatial utilisation of the wetland environment by individual organisms.

A.3 MODARIA

In October 2012, the IAEA started a new programme to improve capabilities in the field of environmental radiation dose assessment called MODARIA (Modelling and Data for Radiological Impact Assessments)[m]. There are a number of working groups within this programme and one of these, Working Group 8, has begun to address spatial considerations within its work programme [Wood et al., 2014]. This work was initiated in recognition of the fact that contamination in the natural environment is invariably spatially heterogeneous and that animal movement in the environment is governed by factors such as habitat preferences, topography and food availability. Such considerations are seldom taken into account when applying current, conventional approaches for NHB dose assessments.

An example of the inclusion of habitat preferences and food availability considerations within the application of current modelling approaches is the modelling of amphibian exposure in a sand dune ecosystem where the species of interest was known to: (i) only use aquatic areas for a short period each year; (ii) forage in terrestrial sites; and (iii) have a known foraging range [Wood et al., 2009]; in this instance the amphibian species under assessment was assumed to spend 100% of its time in the terrestrial parts of the dune system and soil activity concentrations were averaged over a range comparable with the foraging range. However, this is still a simplistic, and purposefully conservative, approach to considering spatial factors within the assessment process. At best, current modelling approaches account for major spatial differences in contaminant distribution by calculating exposure within different areas of contamination, estimating the fraction of time that an organism spends in each area and then summing the fractional doses to obtain an estimate of total dose [e.g. Johansen et al., 2012].

This contrasts with some of the models that have been developed for use in other fields of pollution ecology [e.g. Hope, 2005; Loos et al., 2010]. These models attempt to quantify exposure by modelling animal movement through a heterogeneously contaminated environment and calculating exposure based on the extent of interaction with the contamination present over the assessment period. Such models have been referred to as ‘individual-based movement models’ (e.g. Hope [2005]), ‘object-oriented models’ (e.g. Loos et al. [2010]) and ‘agent-based models’ (e.g. Forbes and Calow [2012]). Hereafter, they will be referred to collectively as ‘animal-environment interaction models’.

The modelling approach used within these different animal-environment interaction models is broadly similar. For application of these animal-environment interaction models in the context of exposure assessment, georeferenced data on the contamination are used to define the spatial extent and spatially heterogeneous distribution of contaminants within the assessment area. A grid of cells is then overlaid onto the assessment area and modelling rules are established to define the movement of an animal within this grid. The spatial resolution of the grid needs to be commensurate with the spatial resolution of the contamination. Given the spatial nature of the contaminant data set, many of these animal-environment interaction models either directly, or indirectly, utilise a Geographical Information System (GIS) [Wood et al., 2014].

[m] The IAEA has a dedicated website that provides details on the MODARIA programme: http://www-ns.iaea.org/projects/modaria/default.asp?l=116

Animal-environment interaction models range from simple unconstrained random walk models, in which there is an equal probability of the animal moving in any direction and the resultant movement pattern is akin to Brownian motion [Codling et al., 2005], to more advanced modelling approaches that aim to produce more realistic simulations of animal movement by incorporating rules relating to spatial features such as habitat and topography (e.g. Loos et al. [2010]). Given that the models generally utilise a GIS, the habitat and topographical features can be readily incorporated within the modelling environment. One method for parameterising the more advanced animal-environment interaction models is to use habitat characteristics of each grid cell to define a Habitat Suitability Index (HSI) [Purucker, 2006]. The HSI describes the 'attractiveness' of each grid cell based on evaluation of the quality of the habitat and the probability of an animal moving from its current grid cell to a specific neighbouring grid cell is influenced by the HSI of each of the neighbouring cells into which the animal could move. Animals will have a higher probability of moving to a grid cell with higher habitat suitability.

There are various freely available software tools that can be used to facilitate animal-environment interaction modelling. These include Adehabitat [Calenge, 2006] and Eco-SpaCE [Loos et al., 2010]. Adehabitat is a package that includes the capability to integrate with open source GIS solutions, such as QGIS (http://www.qgis.org/en/site/). Eco-SpaCE is a spatial modelling tool that is available within the 'Ecopath with Ecosim' modelling suite (http://www.ecopath.org/).

The need to incorporate spatial analysis of contaminant distributions in relation to animal behavioural characteristics, such as foraging, into environmental risk assessments has long been acknowledged [Kooistra et al., 2001] and there are suggestions that animal-environment interaction models are now being applied for assessing the exposure of wildlife to chemical contamination in the environment for the purpose of undertaking chemical risk assessment [e.g. Loos et al., 2010]. There are also limited examples of the application of such approaches in a radiological assessment context, namely the assessment of doses to wild hogs [Gaines et al., 2005] and raccoons [Chow et al., 2005] at the Savannah River Site, USA.

Given that various tools are now available that facilitate animal-environment interaction modelling and that there is some provenance for the application of such approaches within the context of risk assessment, Working Group 8 of the IAEA MODARIA programme has undertaken to evaluate dose estimates obtained from some of these more complex models in comparison with application of conventional NHB dose assessments using simplistic assumptions regarding animal-environment interaction. The basis of this evaluation is to test the hypothesis "*that current simplistic assumptions, which ignore how animals utilise their environment, ensure wildlife is protected by generating a conservative estimate of exposure (for regulatory purposes)*" [Wood et al., 2014]. Two case study scenarios are being used within MODARIA to investigate these different modelling approaches.

The first case study is focussed on moose (*Alces alces*) in Sweden and will use published data from studies on the movement of moose fitted with Global Positioning System (GPS) collars (e.g. Singh et al. [2012]) in combination with habitat, topography and caesium deposition data for Sweden. These data will allow comparison between modelling approaches and the GPS movement data and will be used to estimate the actual external exposure of moose based on the caesium deposition map. However, no direct radioecological validation data are available. In this context, radioecological validation data would be direct measurements of exposure (e.g. using GPS-collars fitted with dosimeters) or, if internal exposure was also being modelled, live-monitoring measurements to determine the whole-body activity concentrations of the moose being tracked using the GPS collars.

The second scenario is based on studies of reindeer (*Rangifer rangifer*) movement in Norway (e.g. Skarin et al. [2010]). Habitat, topography and GPS collar data are available to enable model evaluation, but in this scenario there is also live monitoring data available which could allow internal exposure

estimates based on animal movement predictions to be compared with external exposure estimates. However, there are no direct exposure measurements against which predictions of external exposure can be compared.

The modelling associated with the moose and reindeer case study scenarios is being undertaken during 2015 and will be reported in the final meeting of the IAEA MODARIA programme, which will be held in Vienna in November 2015. It is anticipated that the findings will be published subsequently within IAEA documentation and in the peer-reviewed literature.

APPENDIX B. APPROACH TO DATA COLLATION FOR SPACE REPRESENTATIVE SPECIES

A systematic review approach [CEE, 2010] was adopted for the parameterisation of the SPACE representative species. Search terms were constructed using common and latin species names coupled with key words such as 'home range', 'territory', 'life span' and 'seasonality'. A wildcard symbol (*) was used to expand the range of keyword variants covered by searches (e.g. 'territor*' to return hits with 'territory', 'territories', 'territorial' or 'territoriality') and Boolean operators (AND, OR, NOT) enabled the searches to be targeted to the most relevant information sources. Taking *Capreolus capreolus* as an example, potential search terms included:

"*Capreolus capreolus*" AND "home range*"

"Roe deer" AND "home range*"

"*Capreolus capreolus*" AND "population range*"

"Roe deer" AND "population range*"

"*Capreolus capreolus*" AND "territor*"

"Roe deer" AND "territor*"

"*Capreolus capreolus*" AND "migrat*"

"Roe deer" AND "migrat*"

"*Capreolus capreolus*" AND "season*"

"Roe deer" AND "season*"

"*Capreolus capreolus*" AND "life span*"

"Roe deer" AND "life span*"

Databases and information sources used to provide data for this study included 'Web of Knowledge' (for peer reviewed journal articles), Google Scholar/Google (to cover technical reports and unpublished data sets) and ecological text books.

APPENDIX C. INPUT DATA FOR SPACE REPRESENTATIVE SPECIES IN A HYPOTHETICAL RELEASE SCENARIO FROM A RADIOACTIVE WASTE DISPOSAL FACILITY

Input data for the SPACE hypothetical release scenario, used to evaluate relative dose rates according to spatial scale assumptions are presented below. Table C-1 provides information on the dimensions used to represent SPACE representative species within the ERICA assessment tool and the occupancy assumptions applied. Table C-2 presents concentration ratios applied for each assessment radionuclide and SPACE representative species.

Table C-1. Input parameters for ERICA

SPACE representative species	Length (m)	Width (m)	Height (m)	Mass (Kg)	Occupancy on soil	Occupancy in soil	Occupancy in water	Occupancy in air	Reference
Roe Deer	1.00E+00	1.60E-01	2.10E-01	1.43E+01	100%	0%	0%	0%	Czyżowski et al. (2009); Ward et al. (2004)
Adder	5.80E-01	2.50E-02	2.50E-02	1.00E-01	25%	75%	0%	0%	Posiva (2014b).
American mink	2.00E-01	6.00E-02	6.00E-02	3.80E-01	30%	70%	0%	0%	Posiva (2014b).
Badger	7.00E-01	3.50E-01	2.00E-01	1.00E+01	20%	80%	0%	0%	Murphy et al. (2009); Racheva et al. (2009);
Bank vole	7.00E-02	3.50E-02	3.50E-02	4.00E-02	40%	60%	0%	0%	Posiva (2014b).
Black garden ant	3.00E-03	1.00E-03	1.00E-03	8.00E-04	0%	100%	0%	0%	Fjerdingstad (2005)
Brown rat	2.00E-01	5.00E-02	6.00E-02	3.14E-01	0%	100%	0%	0%	ICRP Rat
Common earthworm	1.00E-01	1.00E-02	1.00E-02	5.00E-03	0%	100%	0%	0%	Posiva (2014b).
Common honey bee	1.20E-02	3.00E-03	2.00E-03	7.30E-05	0%	100%	0%	0%	Feuerbacher et al. (2003); Roubik et al. (2006)
Common lizard	1.40E-01	1.00E-02	7.00E-03	6.00E-03	60%	40%	0%	0%	Wood et al. (2009)
Common toad	1.50E-01	8.50E-02	7.50E-02	4.45E-02	75%	25%	0%	0%	Wood et al. (2009)
Feral pigeon	1.50E-01	5.40E-02	5.60E-02	3.28E-01	100%	0%	0%	0%	Atasoy et al. (2013)
Fescue grass	5.00E+00	1.00E+00	1.00E+00	2.62E-03	100%	0%	0%	0%	ICRP Wild grass; Posiva (2014b)
Gray wolf	8.50E-01	3.00E-01	4.00E-01	3.74E+01	100%	0%	0%	0%	Harestad and Bunnel, (1979); Schmitz and Lavigne (1987)
Graylag goose	3.80E-01	1.30E-01	1.20E-01	3.43E+00	100%	0%	0%	0%	Fox and Kahlert (2005); Mazanowski et al. (2005)
Great crested newt	1.50E-01	1.90E-02	1.40E-02	7.00E-03	100%	0%	0%	0%	Wood et al. (2009)
Mallard duck	4.00E-01	1.30E-01	1.10E-01	1.58E+00	100%	0%	0%	0%	Posiva (2014b).
Moose	1.65E+00	5.00E-01	7.50E-01	3.24E+02	100%	0%	0%	0%	Posiva (2014b).
Pheasant	2.90E-01	8.80E-02	1.00E-01	9.40E-01	100%	0%	0%	0%	Tobalske and Dial (2000); Adamski (2006)
Rabbit	2.30E-01	6.38E-02	6.50E-02	1.52E+00	0%	100%	0%	0%	Swihart (1984); Setiaji et al. (2012)
Red fox	4.50E-01	1.30E-01	1.30E-01	6.00E+00	70%	30%	0%	0%	Posiva (2014b).
Water vole	1.80E-01	5.00E-02	5.00E-02	1.20E-01	20%	80%	0%	0%	Posiva (2014b).
White tailed deer	1.03E+00	2.60E-01	3.77E-01	4.50E+01	100%	0%	0%	0%	Ditchkoff et al. (2001); Bundy et al. (1991)
Willow Ptarmigan	3.00E-01	1.50E-01	1.60E-01	5.30E-01	100%	0%	0%	0%	Erikstad et al. (1985); Chr (1988)

Table C-2: Concentration ratios used for SPACE representative species within the ERICA assessment tool.

Species	C	Cl	Cs	I	Np	Pb	Po	Ra	Se	Tc	Th	U
Adder	1.34E+03	7.00E+00	5.70E-01	4.00E-01	4.08E-02	3.70E-01	9.50E+00	3.60E-02	3.50E-01	3.90E-01	2.20E-03	4.80E-03
American mink	1.34E+03	7.00E+00	2.76E+00	4.00E-01	4.08E-02	3.70E-02	1.68E+00	3.53E-02	1.90E-01	3.70E-01	7.00E-05	7.09E-04
Badger	1.34E+03	7.00E+00	3.40E+00	4.00E-01	4.08E-02	3.70E-02	1.68E+00	4.00E-03	1.90E-01	3.70E-01	7.00E-05	7.09E-04
Bank vole	1.34E+03	7.00E+00	3.49E+00	4.00E-01	4.08E-02	3.70E-02	4.17E+00	6.91E-02	2.90E-01	3.70E-01	7.74E-03	2.60E-03
Black garden ant	4.30E+02	3.00E-01	1.10E-01	3.00E-01	9.92E-02	4.00E-01	1.00E-01	3.20E+00	3.10E+00	5.75E-01	9.20E-03	1.80E-02
Brown rat	1.34E+03	7.00E+00	9.10E-01	4.00E-01	4.08E-02	1.40E-02	7.50E-04	4.70E-02	1.70E-01	3.70E-01	7.00E-05	6.40E-04
Common earthworm	4.30E+02	1.80E-01	8.10E-02	1.60E-01	1.01E-01	7.20E-01	1.00E-01	3.20E+00	3.10E+00	5.75E-01	9.20E-03	3.30E-03
Common frog	1.34E+03	7.00E+00	5.60E-01	4.00E-01	4.08E-02	3.10E-03	8.90E-02	3.60E-02	3.50E-01	3.90E-01	2.20E-03	4.80E-03
Common honey bee	4.30E+02	3.00E-01	7.00E-03	3.00E-01	1.27E-01	6.09E-02	1.00E-01	3.20E+00	1.40E-01	5.75E-01	4.20E-04	8.90E-04
Common Lizard	1.34E+03	7.00E+00	5.70E-01	4.00E-01	4.08E-02	3.70E-01	9.50E+00	3.60E-02	3.50E-01	3.90E-01	2.20E-03	4.80E-03
Common toad	1.34E+03	7.00E+00	4.60E-01	4.00E-01	4.08E-02	1.20E-01	8.90E-02	3.60E-02	3.50E-01	3.90E-01	2.20E-03	4.80E-03
Feral pigeon	1.34E+03	7.00E+00	5.60E-01	4.00E-01	4.08E-02	6.10E-02	1.00E-02	3.60E-02	3.50E-01	1.70E-01	3.90E-04	5.41E-04
Fescue grass	8.90E+02	2.10E+01	1.10E+00	1.40E-01	1.72E-02	8.00E-02	2.30E-01	1.60E-01	1.60E+00	1.40E+01	1.60E-01	1.20E-01
Gray wolf	1.34E+03	7.00E+00	2.76E+00	4.00E-01	4.08E-02	3.70E-02	1.68E+00	3.53E-02	1.90E-01	3.70E-01	5.52E-03	7.09E-04
Great crested newt	1.34E+03	7.00E+00	4.60E-01	4.00E-01	4.08E-02	1.20E-01	8.90E-02	3.60E-02	3.50E-01	3.90E-01	2.20E-03	4.80E-03
Greylag goose	1.34E+03	7.00E+00	5.60E-01	4.00E-01	4.08E-02	6.10E-02	1.00E-02	3.60E-02	3.50E-01	1.70E-01	3.90E-04	5.41E-04
Mallard duck	1.34E+03	7.00E+00	4.00E+01	4.00E-01	4.08E-02	6.10E-02	1.00E-02	8.40E-02	3.50E-01	1.70E-01	3.90E-04	5.41E-04
Moose	1.34E+03	7.00E+00	4.00E+00	4.00E-01	4.08E-02	1.00E-02	4.17E+00	4.13E-02	2.90E-01	3.70E-01	7.40E-03	1.80E-03
Pheasant	1.34E+03	7.00E+00	5.60E-01	4.00E-01	4.08E-02	6.10E-02	1.00E-02	3.60E-02	3.50E-01	1.70E-01	3.90E-04	5.41E-04
Rabbit	1.34E+03	7.00E+00	4.00E+00	4.00E-01	4.08E-02	3.70E-02	4.17E+00	4.13E-02	2.90E-01	3.70E-01	7.40E-03	1.80E-03
Red fox	1.34E+03	7.00E+00	6.50E-01	4.00E-01	4.08E-02	3.70E-02	1.68E+00	4.00E-03	1.90E-01	3.70E-01	7.00E-05	7.09E-04
Roe deer	1.34E+03	7.00E+00	4.00E+00	4.00E-01	4.08E-02	1.00E-02	4.17E+00	4.13E-02	2.90E-01	3.70E-01	7.74E-03	1.80E-03
Water vole	1.34E+03	7.00E+00	3.49E+00	4.00E-01	4.08E-02	3.70E-02	4.17E+00	6.91E-02	2.90E-01	3.70E-01	7.74E-03	2.60E-03
Water vole	1.34E+03	7.00E+00	3.49E+00	4.00E-01	4.08E-02	3.70E-02	4.17E+00	6.91E-02	2.90E-01	3.70E-01	7.40E-03	2.60E-03
White tailed deer	1.34E+03	7.00E+00	4.00E+00	4.00E-01	4.08E-02	1.00E-02	4.17E+00	4.13E-02	2.90E-01	3.70E-01	7.74E-03	1.80E-03
Willow ptarmigan	1.34E+03	7.00E+00	7.60E-01	4.00E-01	4.08E-02	6.10E-02	1.00E-02	2.53E-02	3.50E-01	1.70E-01	3.52E-04	4.05E-04

APPENDIX D. ADDITIONAL RESULTS

Dose rates for representative species for the thorium and uranium isotopes, excluding Th-230 and U-238, are presented below. Data for four spatial averaging approaches are presented: Human Averaging (HA), Minimum home range (Rmin), Maximum home range (Rmax), SPACE Reference Group (SG) (see text for details).

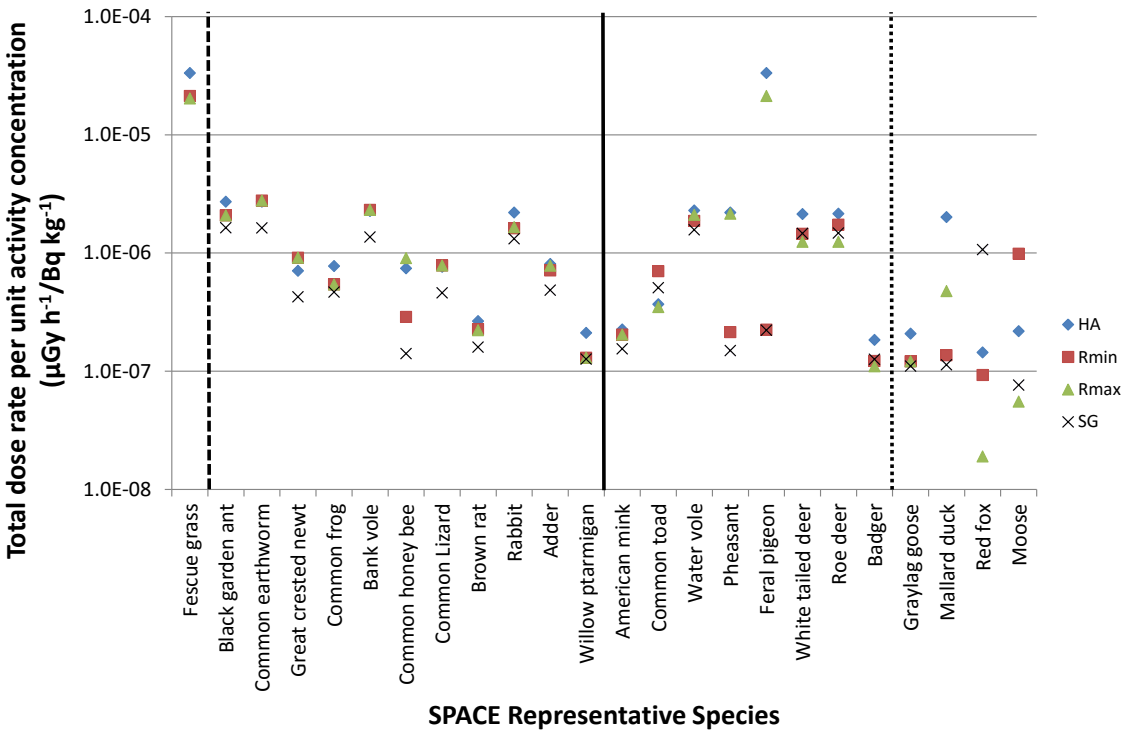

Figure D-1. Total dose rates from Th-229 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}).

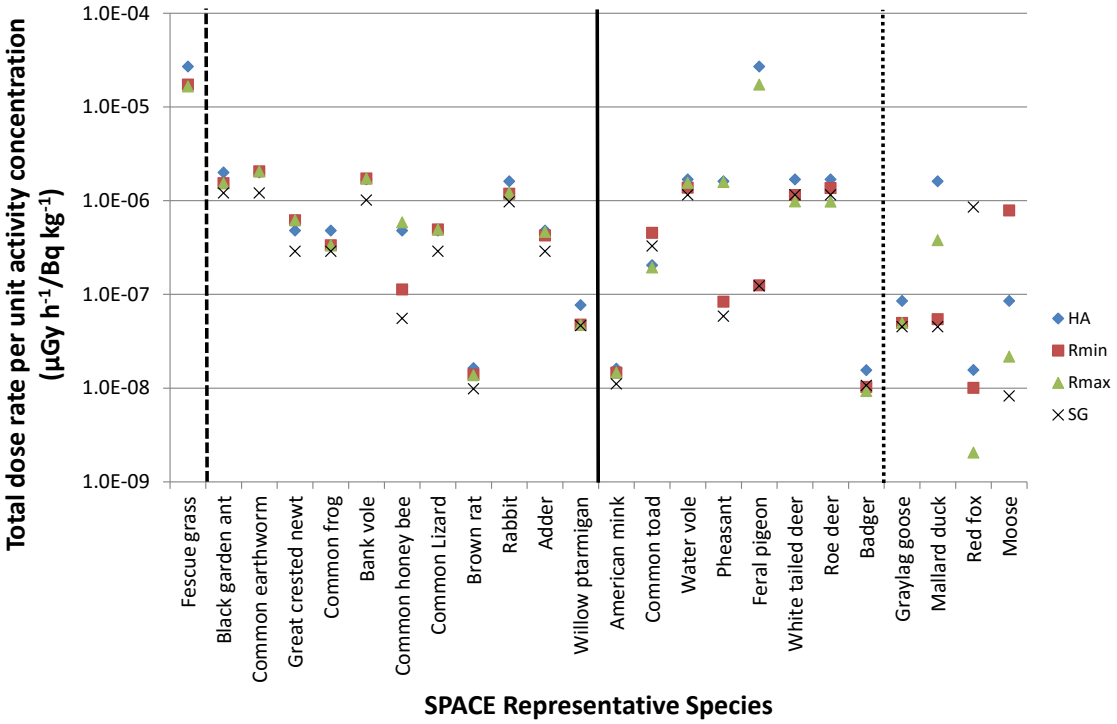

Figure D-2. Total dose rates from Th-232 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}).

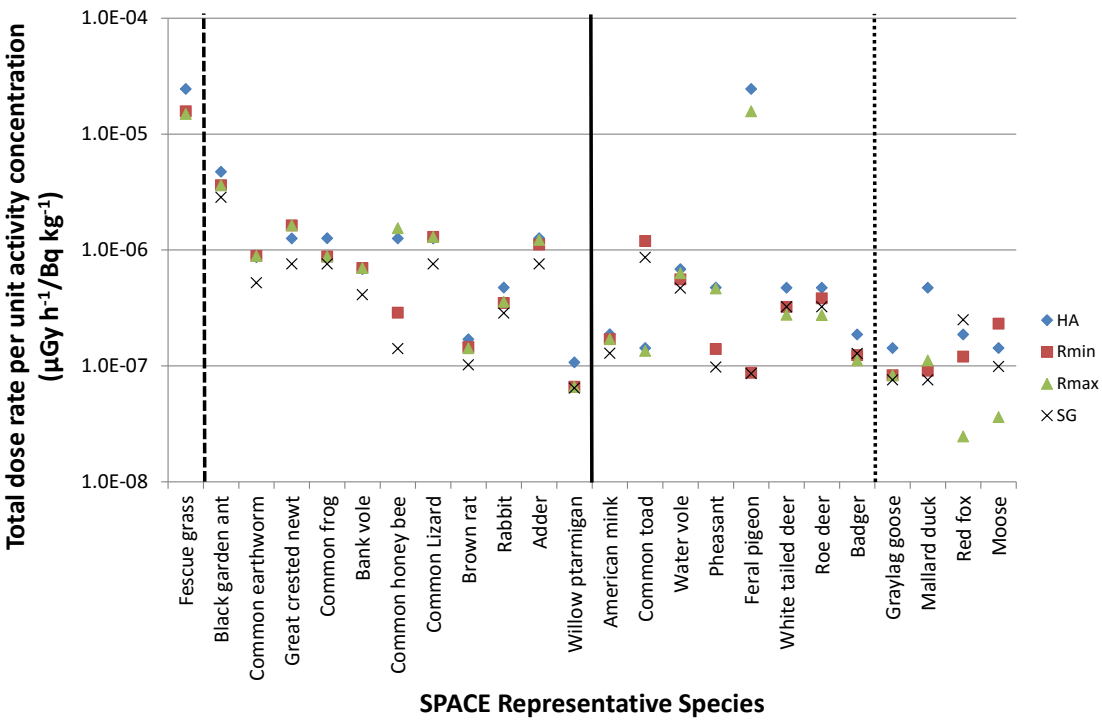

Figure D-3. Total dose rates from U-233 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}).

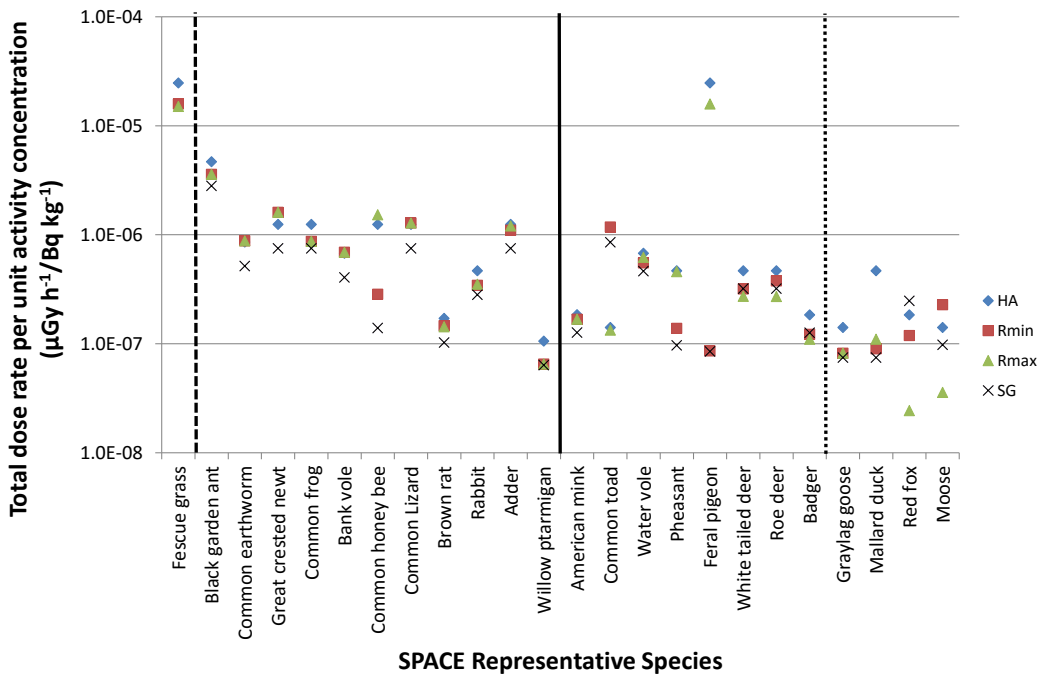

Figure D-4. Total dose rates from U-234 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}).

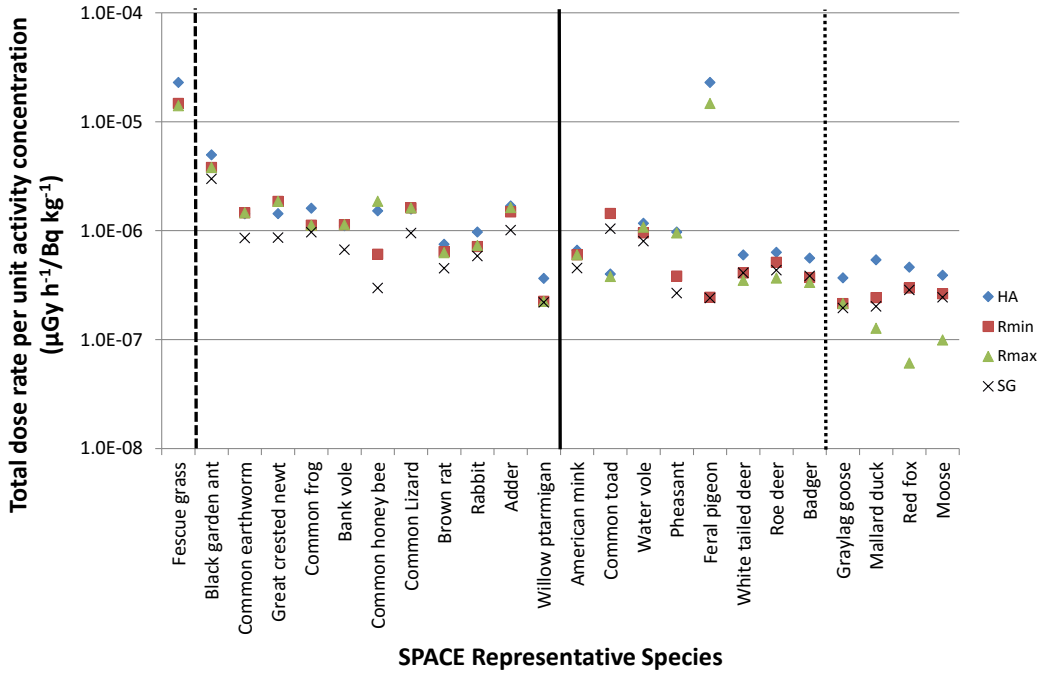

Figure D-5. Total dose rates from U-235 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}).

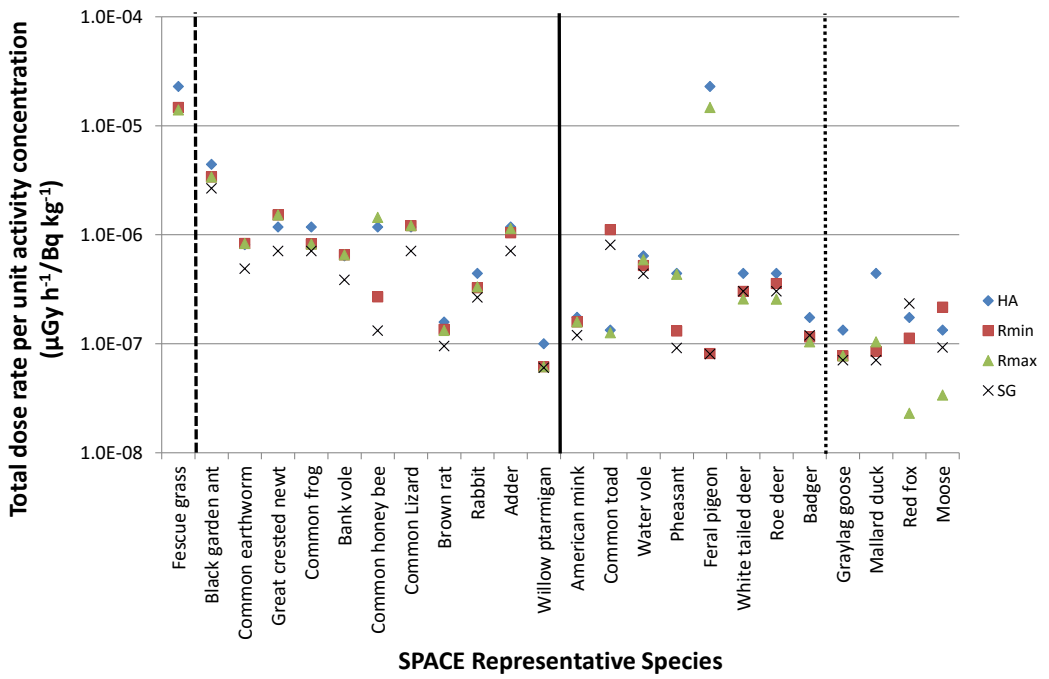

Figure D-6. Total dose rates from U-236 for SPACE representative organisms, assuming an initial soil activity concentration of unity (µGy h^{-1} / Bq kg^{-1}).