Modelling Radiation Exposure and Radionuclide Transfer for Non-human Species

Report of the Biota Working Group of EMRAS Theme 3

Environmental Modelling for RAdiation Safety (EMRAS) Programme

FOREWORD

Environmental assessment models are used for evaluating the radiological impact of actual and potential releases of radionuclides to the environment. They are essential tools for use in the regulatory control of routine discharges to the environment and also in planning measures to be taken in the event of accidental releases; they are also used for predicting the impact of releases which may occur far into the future, for example, from underground radioactive waste repositories. It is important to check, to the extent possible, the reliability of the predictions of such models by comparison with measured values in the environment or by comparing with the predictions of other models.

The International Atomic Energy Agency (IAEA) has been organizing programmes of international model testing since the 1980s. The programmes have contributed to a general improvement in models, in transfer data and in the capabilities of modellers in Member States. The documents published by the IAEA on this subject in the last two decades demonstrate the comprehensive nature of the programmes and record the associated advances which have been made.

From 2002 to 2007, the IAEA organised a programme titled “Environmental Modelling for RAdiation Safety” (EMRAS). The programme comprised three themes:

Theme 1: Radioactive Release Assessment

Working Group on the revision of IAEA Handbook of parameter values for the prediction of radionuclide transfer in temperate environments (Technical Reports Series (TRS) 364).

Working Group on model testing related to countermeasures applied to the intake of iodine-131 from the Chernobyl accident.

Working Group on testing of models for the environmental behaviour of tritium and carbon-14 following routine and accidental releases.

Working Group on testing of models for predicting the behaviour of radionuclides in freshwater systems and coastal areas.

Theme 2: Remediation Assessment

Working Group on testing of models for the remediation of the urban environment.

Working Group on modelling the transfer of radionuclides from naturally occurring radioactive material (NORM).

Theme 3: Protection of the Environment

Working Group on the review of data and testing of models for predicting the transfer of radionuclides to non-human biological species.

This report describes the work of the Biota Working Group under Theme 3. The IAEA wishes to acknowledge the contributions of the Working Group Leaders, B. Howard and N. Beresford (both of the United Kingdom), as well as J. Vives i Batlle (United Kingdom) and T. Yankovich (Canada) to the editorial preparation of this report. The IAEA Scientific Secretary for this publication was initially M. Balonov and subsequently D.M. Telleria both of the Division of Radiation, Transport and Waste Safety.

CONTENTS

SUMMARY

Internationally, the ICRP, IAEA and European Commission (EC) are addressing environmental protection as an element of their revision of Recommendations and Basic Safety Standards. Some countries already have requirements and guidelines for the protection of non-human biota. For instance, in England and Wales, the requirement to assess impacts affecting Natura 2000 sites has been interpreted to include ionising radiation. In the USA, biota protection guidelines and dose rates are contained in USDOE Orders 5400.5 and 450.1.

In response to these developments, a number of models and approaches have been developed specifically to estimate the exposure of non-human biota to ionising radiations. Some countries (e.g. Canada, Finland, England and Wales, and the USA) are now using these within their national regulatory frameworks for (existing and proposed) nuclear and other sites that may release radioactivity to the environment. Software and/or documentation for some of these approaches are readily available and hence third parties are able to use them when conducting assessments.

The *Biota Working Group* (BWG) of the IAEA Environmental Modelling for Radiation Safety programme was formed in 2004 to address the relative lack of validation and intercomparison of the different models and approaches. The primary objective of the BWG, was: '*to improve Member State's capabilities for protection of the environment by comparing and validating models being used, or developed, for biota dose assessment (that may be used) as part of regulatory process of licensing and compliance monitoring of authorised releases of radionuclides'*. Group members included modellers, regulators, industry and researchers.

In total, 15 models and approaches were applied to one or more of the four exercises conducted by the BWG. The models/approaches applied encompass those being developed, and in some instances, used in a regulatory context, in Belgium, Canada, France, Lithuania, Russia, the UK and the USA, as well as the outputs of recent EC EURATOM programmes. The participating models included those freely available to any interested users.

The four intercomparison exercises included evaluations of the basic components of the models assuming 1 Bq per unit media or 1 Bq kg^{-1} in the organism, and two scenario (one freshwater and one terrestrial) applications in which model predictions were compared to available field measurements.

The work of the BWG has clearly demonstrated that the largest contribution to variability between model predictions, and comparison with available data, is the parameterisation of the models transfer components. The methods used to determine absorbed dose rate contribute relatively little to variability between model outputs.

The report concludes with recommendations for future activities within the EMRAS II programme.

CHAPTER 1. INTRODUCTION

Over the past decade, the need for a system to protect the environment from ionising radiation has been recognised internationally and the International Commission on Radiological Protection [1] is addressing environmental protection as an element of its ongoing revision of recommendations. Some countries already have requirements and guidelines for the protection of non-human biota. In the USA, the United States Department of Energy (USDOE) has published biota protection guidelines and dose rates [2, 3]. In England and Wales, the requirement to assess impacts affecting Natura 2000 sites (under the Conservation (Natural Habitats) Regulations 1994 the UK implementation of the EU Birds and Habitats Directives[1]) has been interpreted to include ionising radiation. In Canada, the responsibility is on the licensee to demonstrate that the environment is adequately protected.

In response to these developments, a number of models and approaches have been developed specifically to estimate the exposure of non-human biota to ionising radiations (e.g. [4–7]. Some countries (e.g. Canada, England and Wales, Finland and the USA) are now using these within their national regulatory frameworks for (existing and proposed) nuclear and other sites that may be releasing radioactivity to the environment (e.g. [8, 9]). Software and/or documentation for some of these approaches are readily available and hence third parties are able to use these (see Appendix I).

Previously, there has been only limited validation of these approaches (e.g. [10, 11]) and there has been virtually no attempt to compare the outputs of the different models used. To address this gap, the *Biota Working Group* (BWG: see http://www-ns.iaea.org/projects/emras/emras-biota-wg.htm) was formed by the IAEA as part of the EMRAS (Environmental Modelling for Radiation Safety) programme in November 2004. The primary objective of the BWG, as defined by its members, was: '*to improve Member State's capabilities for protection of the environment by comparing and validating models being used, or developed, for biota dose assessment (that may be used) as part of regulatory process of licensing and compliance monitoring of authorised releases of radionuclides'*.

In total, 15 models and approaches, which are described in Chapter 2, have participated in the BWG. These encompass those being developed and, in some instances applied in a regulatory context, in the USA, Canada, France, Belgium, Russia, Lithuania, and England and Wales, as well as the outputs of international programmes.

This report describes the work and findings of the BWG. The group first conducted two model-model comparisons to evaluate the basic components of the various participating models. These were comparisons of predicted: (i) unweighted absorbed dose rates (Chapter 3); and (ii) biota whole body activity concentrations (Chapter 4). Both exercises assumed unity media activity concentrations. Subsequently, the models were applied to two scenarios to allow the comparison of predictions to available data for a freshwater (Chapter 5) and a terrestrial (Chapter 6) site. A scenario for marine or coastal ecosystems was not included because insufficient of the participating approaches consider these ecosystems. The final chapter of this report presents conclusions and recommendations for the future based upon the findings of the BWG.

[1]Council Directives 79/409/EEC and 92/43/EEC.

CHAPTER 2. MODEL DESCRIPTIONS

The participating approaches included environmental assessment tools (considering at least transfer and dosimetry), transfer models, dosimetry tools and approaches being used or developed by individual organisations. Descriptions are provided below together with key references which describe the method in more detail for those approaches which are already well documented. Less well documented approaches are more fully described in the text below. Appendix I provides a tabulated summary of each participating approach.

This chapter provides overall description of the approaches, specifics associated with thier application to the four exercises conducted by the BWG are provided in the appropriate chapters. In most instances, the models have been applied by organisations involved in their development. However, in the latter stages of the work of the BWG, organisations other than the originator applied one of the approaches to the freshwater ecosystem scenario.

2.1. Common elements

Whilst the derivation and parameterisation of the models described below differs, most have some commonality in their rationale and elements which are described in this section.

2.1.1. Dosimetry

Radionuclides in the environment lead to both external and internal exposure of plants and animals to ionising radiation. Internal exposure arises following the uptake of radionuclides by organisms via pathways such as ingestion or root uptake. External radiation exposure depends on various factors, including contamination levels in the environment, the geometrical relationship between the radiation source and the organism, organism size, shielding properties of the medium, and the physical properties of the radionuclides present.

Dosimetry for biota, therefore, represents a wide range of exposure conditions, as well as the inevitable variability of species and habitats. Consequently, a number of extreme simplifications are made. One commonly used simplification is the reduction of the whole organism to simple shapes, such as ellipsoids and cylinders [12, 13]. Although radionuclide concentrations in animals and plants display variations amongst tissues and organs as observed in humans, radionuclide kinetics in the organism and organ distribution are generally not taken into account. The endpoint considered is the average absorbed dose rate for the whole body per unit activity concentration in the organism or the surrounding media. This is estimated by the use of dose conversion coefficients (DCCs), which relate unweighted absorbed dose rate to the activity concentration in an organism or media.

Although current practice is to consider the absorbed dose rate averaged over the whole organism, there are instances where the non-uniform distribution of radionuclides in tissue (e.g. see [14]) can be important. One such instance is exposure of radiosensitive tissues to incorporated α-emitting radionuclides, resulting in significantly higher doses than obtained with a uniform distribution [4].

Absorbed dose, in all of the existing approaches to estimate non-human exposure, is defined as the amount of energy absorbed per unit mass of tissue of an organ or organism, given in units of Gray (Gy) [15]. Application of multiplicative radiation weighting factors, based on experimental data for relative biological effectiveness to derive an equivalent dose, is used for biota in some of the existing approaches. However, as there is currently no general agreement

as to the appropriate radiation weighting factors that should be applied for alpha and low-energy beta radiation, we have restricted our comparison to unweighted absorbed doses.

A fundamental quantity for estimating internal exposure is the absorbed fraction (AF), which is defined as the fraction of energy emitted by a decaying atom that is absorbed within the organism [12, 13, 16, 17]. In the simplest case, the organism is contained in an infinite homogeneous medium, activity is uniformly distributed throughout and the densities of both the medium and the organism are assumed to be equal. Under such conditions, both internal (D_{int}) and external (D_{ext}) radiation dose rates for mono-energetic α-, β-, and γ-radiation (in units of Gy s^{-1}) can be expressed as function of the absorbed fraction:

$$\begin{aligned} D_{\text{int}} &= 1.6 \cdot 10^{-13} \cdot q \cdot E_i \cdot AF\ (E_i) \\ D_{ext} &= 1.6 \cdot 10^{-13} \cdot q \cdot E_i \cdot (1 - AF\ (E_i)) \end{aligned} \tag{2.1}$$

Where 1.6×10^{-13} is a conversion factor (J MeV^{-1}), i denotes the radiation type (α-, β-, γ-radiation or spontaneous fission fragments), q is the radionuclide activity concentration in the organism or in the surrounding media (Bq kg^{-1}), and E_i is the energy (MeV)[2].

The absorbed dose rate can never exceed that in a uniform infinite media (D_∞), which for any given mono-energetic particle is:

$$D_\infty = 1.6 \cdot 10^{-13} \cdot q \cdot E_i \tag{2.2}$$

If the organism's dimensions are much smaller than the radiation range in the medium, especially for longer-range radiation (high-energy electrons and photons), then AF → (tends towards) 0, $D_{int} \rightarrow 0$ and $D_{ext} \rightarrow D_\infty$. Conversely, when the size of the organism is much larger than the radiation range in the medium (especially for α-particles and low-energy electrons with a range of less than 50–100 μm), then AF → 1, $D_{int} \rightarrow D_\infty$ and $D_{ext} \rightarrow 0$.

2.1.2. Transfer

The fresh weight (fw) activity concentrations of radionuclides in biota are predicted from media activity concentrations using equilibrium concentration ratios (CRs) for at least some organisms by all of the participating approaches which include a transfer component. Other approaches used are detailed as appropriate in the model descriptions below (see also Chapters 3–5 for specifics relating to application in the BWG exercises). The definitions of CR for terrestrial and aquatic ecosystems as used by most approaches within the work described here are:

2.1.2.1. Terrestrial

$$CR = \frac{Activity\ concentration\ in\ biota\ whole\ body\ (Bq\ kg^{-1}\ fresh\ weight)}{Activity\ concentration\ in\ soil\ (Bq\ kg^{-1}\ dry\ weight\ (dw))} \tag{2.3}$$

Exceptions, for those radionuclides considered in subsequent chapters, are for chronic atmospheric releases of ^{3}H and ^{14}C where:

[2]The equation for D_{ext} is an approximation that only holds if the organism and the surrounding medium are of the same density and elemental composition.

$$CR = \frac{Activity\ concentration\ in\ biota\ whole\ body\ (Bq\ kg^{-1}\ fresh\ weight)}{Activity\ concentration\ in\ air\ (Bq\ m^{-3})} \quad (2.4)$$

2.1.2.2. Aquatic

$$CR = \frac{Activity\ concentration\ in\ biota\ whole\ body\ (Bq\ kg^{-1}\ fresh\ weight)}{Activity\ concentration\ in\ filtered\ water\ (Bq\ l^{-1})} \quad (2.5)$$

In aquatic ecosystems most participating approaches also use distribution coefficients (K_d) to describe the relative activity concentrations of sediment and water:

$$K_d\ (l\ kg^{-1}) = \frac{Activity\ concentration\ in\ sediment\ (Bq\ kg^{-1}\ dry\ weight)}{Activity\ concentration\ in\ filtered\ water\ (Bq\ l^{-1})} \quad (2.6)$$

2.1.3. Reference organisms

Different terms are used for the organisms being assessed in the different approaches, including: "reference organism", "representative species", "feature species" and "receptor". In many, although not all of the participating approaches, these terms are used for a set of default organisms. The terminology and definitions used by the different approaches (in terms of 'reference organisms', 'selected species', etc.) are expanded upon within Appendix II together with an overview of selection criteria. The ICRPs proposed reference animals and plants [18] are included within this discussion.

2.1.4. Assessment structure

A number of the approaches described adopt what may be termed a 'tiered' approach when applied for the purposes of regulatory decision-making. Lower tiers are normally highly conservative and require minimal data input. Higher tiers are more realistic and require a more detailed assessment. The purpose of the lower tiers is to rapidly screen out situations where there is no risk of impact on non-human biota with a high degree of confidence. Assessment effort can thus be focussed on those situations which may result in impacts (or which require detailed assessment for other reasons, such as stakeholder interest).

The exercises conducted by the BWG have concentrated on comparing the underlying model parameters and has not compared application of the tiered assessments in scenarios.

2.2. Atomic Energy Canada Limited (AECL) approach

AECL has typically adopted a multi-tiered approach ranging from very conservative Tier 1 to more realistic Tier 3 (based on [19, 20]). Site-specific transfer parameters are preferred, with (preferably Canadian) values from the scientific literature being taken when site-specific data are not available. To determine dose, DCC values and methods to estimate them are taken from various published sources. For the purposes of model-model comparisons, in some instances the opportunity was taken to apply the RESRAD-BIOTA tool (Section 2.4), the FASSET parameters (Section 2.6), and the methodology to estimate DCCs that has been developed by Blaylock et al. [17], to gain more familiarity with available international approaches. The approach used by AECL is fully described in Appendix III.

2.3. RESRAD-BIOTA

RESRAD-BIOTA is a computer code that implements the U.S. Department of Energy's (USDOE's) Graded Approach methodology described in DOE Technical Standard DOE-STD-1153-2002, *A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota* [7]. The Graded Approach methodology was developed through the Department's Biota Dose Assessment Committee (BDAC). The code, developed by Argonne National Laboratory, was sponsored by DOE's Office of Environment, Safety and Health, and the Office of Environmental Management, with support from the U.S. Environmental Protection Agency and the U.S. Nuclear Regulatory Commission. The code is freely available from http://www.evs.anl.gov/resrad.

RESRAD-BIOTA considers radiation exposure to biota in terrestrial and aquatic (freshwater) ecosystems; there are 46 radionuclides currently in the database. A range of organisms were evaluated to develop default exposure parameter values. These reference organisms are categorised into terrestrial animals and terrestrial plants for the terrestrial ecosystem, and aquatic animals and riparian animals for the aquatic ecosystem. RESRAD-BIOTA also has the capability of evaluating radiation exposures for specific organisms, providing their exposure parameters are input by the users.

Potential radiation exposures of biota are evaluated following a graded approach that consists of three tiers of analysis. At Level 1, conservative assumptions are made through provision of a general screening process employing DOE's Biota Concentration Guides (radionuclide concentrations in environmental media that would not exceed recommended dose rate guidelines) and few inputs are required. As the user progresses to Levels 2 and 3, fewer assumptions are made but more site- or receptor-specific input data are required; greater user flexibility is offered at Levels 2 and 3. Analysis can start at the general screening level (Level 1) and proceed through Level 2 to Level 3 if screening values are exceeded or receptor-specific evaluations are desired. The code is based on an independently peer-reviewed and user-tested methodology. The code includes a kinetic-allometric approach [21] to estimate the transfer of radionuclides to animals (and which has been used in the BWG exercises). The internal and external dose conversion coefficients (DCCs), relating unweighted absorbed dose to media or biota activity concentrations, are estimated using a Monte-Carlo transport code.

Whilst in the work described in Chapters 3, 4 and 6 the model developers applied RESRAD-BIOTA to the exercises, for the freshwater scenario (Chapter 5) it was applied by two groups previously unfamiliar with the approach. A beta (development) version (1.22) of the RESRAD-BIOTA software was used in the BWG activities.

2.4. ERICA (Environmental Risk from Ionising Contaminants – Assessment and Management)

ERICA was a European Commission (EC) 6th Framework project to provide an integrated approach to scientific, managerial and societal issues concerned with the environmental effects of ionising radiation [4]. The ERICA Tool is a software programme for implementing the ERICA Integrated Approach [22]. Transfer from contaminated media to a range of terrestrial, freshwater and marine reference organisms is estimated using concentration ratios (CRs), predominantly derived from the literature (see [23, 24]. If CR values were not available from the available literature, default values were estimated using an approach ('guidance') developed from that originally described by Copplestone et al. [6]. This included adopting CR values for: taxonomically similar reference organisms in the same ecosystem

type (e.g. for a given radionuclide a CR for flying insects may have been used as the default CR for other terrestrial invertebrate reference organisms); biogeochemically similar radionuclides for a given reference organism (e.g. CRs for transuranic and lanthanide elements were assumed if CRs were not available for a member of these series). This approach is fully documented in Beresford et al. [23]. The ERICA methodology is a tiered approach allowing site- specific, or user-defined, CR values to be used (if sufficient biota activity concentrations are not available) at higher tiers.

For dosimetry, reference organisms are defined as simple three dimensional phantoms (i.e. ellipsoids and cylinders). These provide geometric equivalents of reference organisms according to average characteristics of mass and size (see [25]). The default reference organism geometries available within the ERICA Tool include all of those for the adult stages of the ICRPs proposed reference animals and plants (RAPs) [18]. The approach assumes that a layer of non-active tissue, i.e. the outer layers of the skin and/or fur, provide a degree of shielding for the living organism. Monte-Carlo techniques are applied that include all relevant radiation transport processes. Daughter products with a physical half-life of less than 10 days are assumed to be in secular equilibrium with the parent radionuclide (and hence contribute to the DCC estimated for the parent nuclide). The user is also able to define their own geometries for considering additional organisms.

The ERICA Tool and associated documentation are freely available from: http://www.ceh.ac.uk/PROTECT/ERICAdeliverables.html.

2.5. FASSET (Framework for the Assessment of Environmental Impact)

The FASSET Assessment framework was developed under an EC 5th Framework project [26–28]. The framework considered 31 reference organisms and radionuclides of 20 elements in freshwater, marine and terrestrial systems.

Look-up tables of CR and DCC values are provided in Brown et al. [26]. Transfer from contaminated media is estimated using concentration ratios (CRs), predominantly derived from the literature, which are presented as look-up tables. For some radionuclide-organism combinations, if data were lacking, CR values for some organisms were derived using an atmospheric deposition-soil-plant model coupled with allometric expressions for animal parameters including radionuclide biological half-life [29]. Other missing CR values were derived by adapting the guidance of Copplestone et al. [6], developed for the England and Wales Environment Agency approach described in Section 2.7. Dose conversion coefficients are presented for a range of terrestrial and aquatic reference organisms, which are defined by ellipsoid geometries representative of species corresponding to the reference organisms. For terrestrial systems, DCCs were estimated using a Monte-Carlo approach. For aquatic systems, the approach, as used in the Environment Agency R&D128 methodology, is applied.

The FASSET framework only participated in the two model-model comparisons (Chapters 3 and 4) as it has been superseded by the ERICA Integrated Approach which was still under development at the beginning of the BWGs activities. However, some participants adopted elements (i.e. CR or DCC values) of the FASSET framework within their own approach (this is noted in model descriptions and applications as appropriate).

2.6. England and Wales Environment Agency R&D128 (EA R&D128)

Developed to assess Natura 2000 sites for compliance with the EC Habitats Directive in England and Wales, this model uses a similar approach to that of ERICA, although a smaller range of organisms and radionuclides is considered [5, 6]. Dose conversion coefficients are estimated using simple functions for energy deposition in a medium of unit density from point isotropic sources to represent the absorption of photons and electrons. Energy absorbed fraction functions are fitted separately for photons and electrons to provide an interpolation between calculated values. These functions are then integrated numerically using a stochastic (Monte-Carlo) algorithm to calculate the absorbed fraction. The model uses CR values from literature reviews to estimate activity concentrations in biota based on media concentrations [5]. Guidance is provided on how to estimate CR values if they are missing for a given radionuclide-organism combination [6]. The nature of this guidance ranges, for example, from using a CR value for an organism of similar ecology through to assuming that the CR value is 1 for a terrestrial organism if no data were available to suggest it may be higher than this. The guidance aims to provide conservative values where data-derived (or site-specific) CR values are lacking as the overall approach to the assessment is to be conservative. This guidance was later adapted for use within the FASSET and subsequent ERICA approaches.

The spreadsheet-based models, provided with this approach, are freely available from http://www.coger.org.uk/R&D128index.html This approach is refereed to as EA R&D128 in the subsequent text.

2.7. LIETDOS-BIO

This is an approach being developed to address contamination issues associated with nuclear power production in Lithuania [30]. The model uses two CR databases: site-specific (used by preference) and generic (mostly based on FASSET and data from the Russian language literature). A Monte-Carlo transport code is used for DCC derivation. A specially derived method for describing phantoms allows DCC values to be calculated for organisms of any size or shape [30].

2.8. DosDiMEco

This model is under development by SCK·CEN and an extended description can be found in Appendix IV. The approach consists of a software package, written in MathCad 2001i professional, which is divided into three sub-programs. The first sub-program calculates the energy absorption in a reference organism due to gamma irradiation originating from a certain contaminated volume. The two remaining sub-programs follow a similar approach, but calculations are performed for a volumetric contamination by an α- or a β-emitter. Appropriate mass attenuation data are taken from the literature [31].

DCCs are calculated using a point kernel technique (corrected with a build-up factor) [32]. For β DCCs, an approach using the Bethe-Bloch formula [15] is implemented. For α radiation the same equations are used, except that in this case, no relativistic or Bremsstrahlung effects need to be taken into account. Radionuclide-specific DCCs are determined by linear interpolation, taking into account the nuclide-specific energies emitted and their emission probabilities.

Concentration ratio values for plants, invertebrates, fish, zooplankton and phytoplankton are predominantly derived from review and other publications [33–40]. Terrestrial mammal and bird concentrations are calculated from the intake rate (using an allometric relation between body mass and intake rate described by Nagy [41] and the fractional absorption of the radionuclide from the gastrointestinal tract [42–52]. This is combined with a retention function to calculate whole-body activity concentrations as described by Coughtrey and co-workers [46–48] and ICRP, [50, 51, 53]. If no retention functions were available, the biological half-lives given in ANL [42–44].

2.9. ECOMOD

In general, the ECOMOD methodology [54–58] considers radionuclide accumulation and elimination by aquatic organisms as a dynamic process, which is linked to the processes of organism growth and metabolism, concentrations of stable analogous elements in the organism, organism diet and the environment in which the organism lives. The model is particularly suited to radionuclides that are analogous to, or isotopes of, biologically active chemical elements.

The ECOMOD semi-empirical model uses available literature, recommended values from handbooks (e.g. [34]) or site-specific data to evaluate equilibrium concentration factors (CF_{eq}) in reference aquatic biota species.

Activity concentration (y) of a radionuclide in aquatic plants, zooplankton, macro invertebrates and mollusc is calculated using the simple formula:

$$y = CF_{eq} \cdot C_{water} \tag{2.7}$$

where:

$_{water}$ is activity concentration of a radionuclide in water; and
CF_{eq} is equilibrium concentration factor of a radionuclide in a given reference organism.

The activity concentration (y in Bq kg^{-1} (fw)) of a given radionuclide in fish is calculated using the following equation [59]:

$$\frac{dy}{dt} = -(\lambda_r + \mu + \varepsilon_A \frac{W}{M}) \cdot y + (\mu + \varepsilon_A \frac{W}{M}) \cdot CF_{eq} \cdot C_{water}(t) \tag{2.8}$$

where:

λ_r is radioactive decay constant (in year^{-1});
μ is the annual average rate of increase of the fish mass (in year^{-1});
M is annual average fish mass (in kg);
W is rate of the fish metabolism (in kg year^{-1});
ε_A is the coefficient of proportionality between the rate of bioelimination of radionuclide from fish and rate of metabolism (which is radionuclide-specific parameter);
CF_{eq} is equilibrium concentration factor of radionuclide in fish (l kg^{-1}); and
$C_{water}(t)$ is activity concentration of a given radionuclide in water (in Bq l^{-1}) at time (t) years.

If:

$C_{water}(t)=\text{constant } y \rightarrow CF_{eq} \cdot C_{water}$ (2.9)

then the concentration factor is expected to be stable and equal to CF_{eq}; however, if the activity concentration of a given radionuclide in the water changes over time, then concentration factor will be dynamic and will be influenced by characteristics depending on fish mass, metabolism (and, accordingly, water temperature), as well as radionuclide turnover in fish tissues. In the latter case, the concentration factor can differ from its equilibrium value.

The ECOMOD semi-empirical model can be applied to calculate the dynamics of radionuclides in fish on the basis of the determined equilibrium concentration factors. This model could be useful for screening estimates, evaluation of the dynamic concentration factors for water bodies where equilibrium concentration factors were clearly determined in previous studies and for the calculation of bioaccumulation of isotopes, which have no stable analogues (such as, Ra, Th, U, Pu, Am and others).

Concentration ratios, predominantly sourced from the Russian language literature, may also be used. ECOMOD uses previously published absorbed fractions across a range of ellipsoids to estimate unweighted absorbed doses for a limited number of radionuclides.

2.10. EDEN 2 (Elementary Dose Evaluation for Natural Environment)

The EDEN code is a Monte Carlo tool [60–62] that estimates DCCs based on two main assumptions: (i) any organism is represented by an ellipsoid, defined by its three axes; (ii) all characteristics defining any source of radiation (density, elemental composition, radionuclide concentration) are considered to be homogeneous throughout the whole volume. All the required data are user-defined with the exception of spectroscopic data which are sources from the JEF database [63].

Depending on the size of the target organism, one of two methodologies is implemented within the tool (recognising the lower penetration power of alpha and beta radiations) (Table 2.1). For large organisms, the internal exposure DCCs for α- and β-irradiation may be calculated using a local deposition method, whilst external exposure is assumed to be negligible. For smaller organisms, all DCCs are calculated using a Monte-Carlo approach. There is no absolute size cut-off between the two methodologies and the choice of methodology for a given organism is left to the user.

Using ellipsoids to describe organisms and plane sources to define the geometrical exposure configuration, Monte Carlo calculations are used to estimate mono-energetic DCCs for given energies in the ranges corresponding to alpha, beta, and gamma spectra (1 to 10 MeV, 0.1 to 3 MeV, 0.01 to 5 MeV respectively). The DCCs for other alpha, beta and gamma emission energies can then be obtained mathematically by interpolating between known mono-energetic DCCs. The final DCCs that are applied in the model correspond to a combination of the mono-energetic DCCs assessed for each energy value in the spectrum under consideration.

Table 2.1. Available methodologies for A and B radiation according to the target size.

Organism	Exposure	
	External	Internal
Macroscopic	Negligible when considering whole-body doses	Local deposition
Microscopic	Monte Carlo calculation	

For alpha particles, a continuous slowing down approximation is used. Two calculation methods are implemented in the tool in relation to beta particles, depending on the relative densities of the different media they cross. For similar densities pre-calculated maps of energy deposition are used. For different densities simulation of a mono-directional electron beam is used. Three methods are applied simultaneously for gamma particles: collision (only the energy deposited during the interactions occurring in the organism is collected), chord flux (based on statistical considerations, associating the energy that may be deposited in a volume with the mean pathways -chords- of the particles in this volume) and virtual flux (combination of the energy deposited by virtual particles, defined to reach the target, and their probability to reach it effectively). The results used to generate the gamma DCCs are those with the lowest statistical uncertainty. The energy loss of an alpha or beta particle along its course is determined by integrating the stopping power of each medium crossed. For gamma particles, mass attenuation coefficients are used.

EDEN results are individual α, β and γ DCCs, plus the total DCCs, weighted and non-weighted, with their mathematical uncertainty, for each source of exposure defined by the user. For each processing step, several statistical tests are assessed with the total proportion of tests passed giving an indicator (referred to as the global confidence index) of the quality of the results produced.

The EDEN 2 model was used in combination with the CASTEAUR (Section 2.12) aquatic transfer model for some BWG activities (see Chapter 5).

2.11. CASTEAUR

The CASTEAUR (French acronym for simplified calculation of radionuclides transfers in receiving waterways) calculation code is designed to estimate spatial and temporal variation in the radionuclide concentrations of biotic and abiotic components of a river as a consequence of liquid releases from nuclear installations [64–68].

The river ecosystem is described by three main abiotic components (i.e. dissolved fraction, suspended matter and sediments) (Figure 2.1) that represent the potential sources of contamination for the biotic components. Biota are defined as three major trophic levels, the primary producers (e.g. phytoplankton), the first order consumers (zooplankton and macrobenthos), and fish (whose diet depends on age and species).

The processes considered are dispersion, dilution and transfer which are related to specific local features (nature of solids and radionuclide, physical and chemical properties of the river, structure and functioning of the trophic web, etc.). In practice, they are combined with physically, biologically and radioecologically homogeneous zones of the river, the ‘reaches’. Their succession constitutes a ‘hydrographic net’, which responds to the functioning of the corresponding river basin.

For one run of CASTEAUR, average values for several physical parameters are used to characterise each reach: length; width; flow rate; diffusion coefficient; and nature (mineral or phytoplankton), load and critical deposition tension of suspended materials. The biological and radioecological parameters are defined constants for each reach: feeding and growth rates; diet by species for fish; and distribution coefficients (K_d), accumulation and depuration kinetics for each radionuclide (data for ^{54}Mn, ^{58,60}Co, ^{103,106}Ru, ^{110m}Ag, ^{134,137}Cs, ^{241}Am are available from experimental data obtained by IRSN).

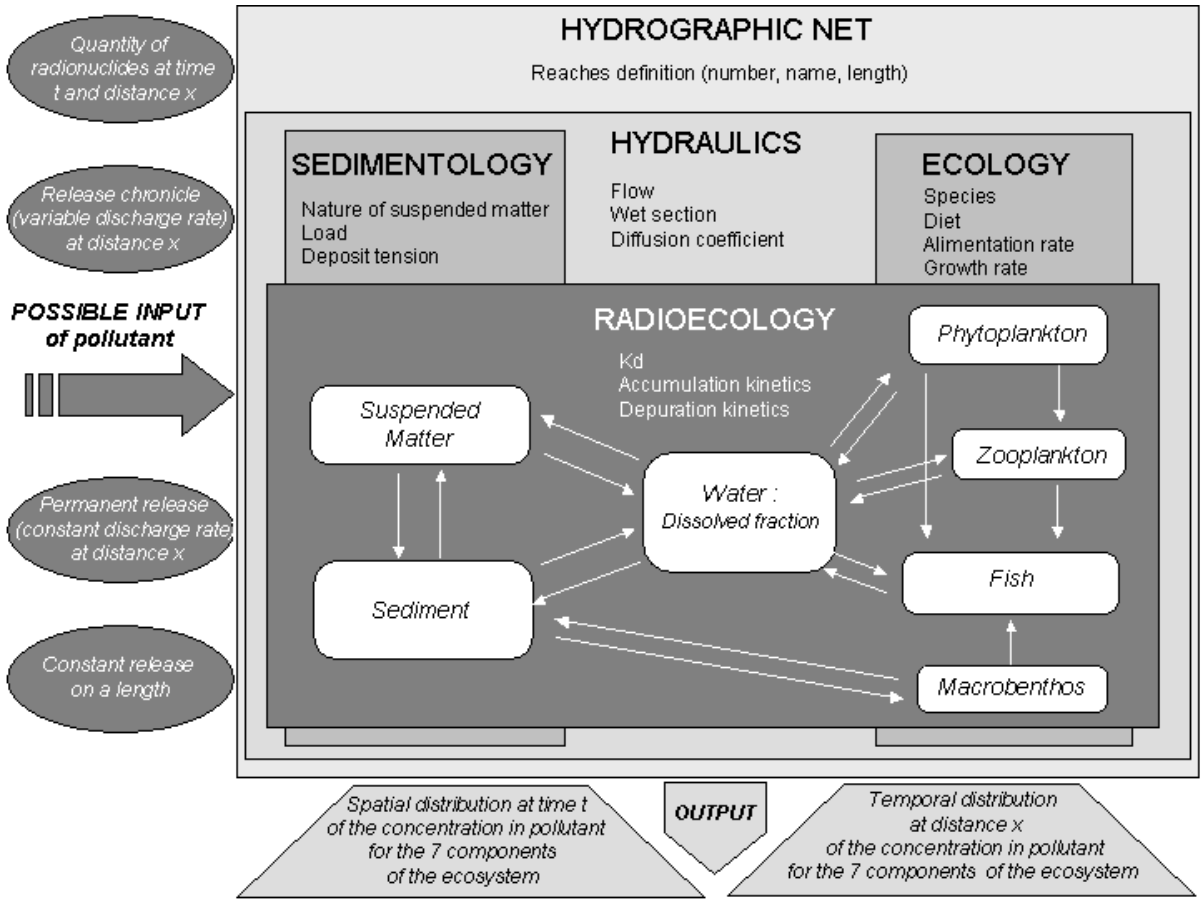

Fig. 2.1. Diagrammatic representation of the CASTEAUR calculation code.

Four pollutant inputs may be considered: point continuous release, point pulse release, point sequential pulse releases and linear fluxes (typically run-off). Inputs can be combined, distributed in space along the river and include several radionuclides.

CASTEAUR outputs are temporal pollutant concentrations in the seven components describing the freshwater ecosystem, distributed along the reaches.

2.12. EPIC DOSES3D

The computer code EPIC DOSES3D [69] is a tool for the calculation of doses to biota developed within the EC Inco-Copernicus Programme's EPIC (Environmental Protection from Ionising Contaminants in the Arctic) project [70].

The software enables the user to define organisms of any size and shape and can be used to derive DCCs for any radionuclide for which transformation data are available. The absorbed fractions for specific geometries are calculated from the chord distribution function that describes numerous possible path lengths within the organism by means of Monte Carlo simulations. The energy deposition along these tracks is quantified by dose attenuation functions; empirical formulae defining dose distribution functions for α and β radiation

around point isotropic sources are used [71–73]. The absorbed fraction is obtained by integration of energy deposited over all tracks within the organisms.

For the case of an organism exposed on the ground surface or at the sediment/water interface, the kinetic energy released in the material (kerma) at a specified location in a given environment is derived [74]. The ratio of the mean absorbed dose in an organism and the kerma in that environment is then calculated for the different energies characteristic of different radionuclides [69]. This ratio allows the absorbed dose-rate to the organism to be derived.

The EPIC DOSES3D software has been used in combination with transfer models in terrestrial (the FASTer model; Chapter 6) and freshwater (CR values derived from the ERICA databases; Chapter 5) assessments conducted by the BWG.

2.13. FASTer

The FASTer model was originally configured to consider a simple food-chain consisting of vegetation-herbivore-carnivore in part to provide transfer parameters for organism-radionuclide combinations for which data were lacking within the FASSET project [26]. It provides a few default CRs within the ERICA Tool database.

FASTer is multi-compartmental model that can be used to simulate transfer through a simple terrestrial food-chain. The rate of change of the radionuclide inventory in the compartments is described with a linear differential equation of the form:

$$dA_i / dt = \sum_j k_{ji} * A_j - \sum_i k_{ij} * A_i \qquad (2.10)$$

where:

k_{ji} is the transfer rate from compartment “j” to compartment “i”; and
k_{ij} is the transfer rate from compartment “i” to compartment “j”.

Originally, the model was set up using separate compartments for herbivore (prey) and carnivore (predator). However this configuration was modified owing to some fundamental concerns involving the conceptualisation of the system. If the various compartments represent a pool of individuals making up an arbitrary population it stands to reason that some parameters describing the population, not least its relative size, is required. On the other hand if the model represents an individual it is clear that losses from a prey animal cannot occur by “partial” predation. In the absence of population-related parameters, a simplified structure, renamed FASTer(lite), was applied whereby the equilibrium activity concentration of prey species were used as inputs to the model for carnivorous animals. In practical terms this only concerned the case of roe deer and wolf, the activity at equilibrium derived for roe-deer was used as input into the “wolf” model.

Transfer from soil to vegetation is modelled using a simple concentration ratio approach assuming equilibrium.

2.13.1. Transfer to herbivores and carnivores

Transfer to the herbivore occurs via ingestion of vegetation and ingestion of soil. Uptake from vegetation, UH_{veg}, is defined by:

$$UH_{veg} = \frac{FMI_H}{M_H} * A_{veg} * f1_H \quad (2.11)$$

where:

FMI_H is the daily fresh matter intake by herbivorous mammals [kg d^{-1}];
A_{veg} is the radionuclide activity concentration in vegetation [Bq kg^{-1} (fw)];
$f1_H$ is the fractional gut uptake for herbivorous mammal [relative units]; and
M_H is the live-mass of the herbivorous mammal [kg].

Uptake from soil, UH_{soil}, is defined by:

$$UH_{soil} = f_s * \frac{DMI_H}{M_H} * A_{topsoil} * f1_s \quad (2.12)$$

where:

fs is the fraction of the daily matter intake by herbivorous mammals that is soil [relative units] = 0.1 [140];
DMI_H is the daily dry matter intake by herbivorous mammals [kg d^{-1}]; and
$f1s$ is the fractional gut uptake of radionuclides ingested with soil [relative units] = 0.1

Carnivores are assumed to consume herbivores, with uptake from the herbivore, UC, defined as:

$$UC = \frac{FMI_C}{M_C} * C_H * f_{soft} * f1_C \quad (2.13)$$

where:

FMI_C is the daily fresh matter intake by carnivorous mammals [kg d^{-1}];
M_C is the live-mass of the carnivorous mammal [kg];
C_H is the radionuclide concentration in the herbivorous mammal [Bq kg^{-1}];
f_{soft} is the fraction of total body activity that is in the soft tissues of herbivores [relative units]=1 for Cs and 0.1 for Sr; and
$f1_C$ is the fractional gut uptake for carnivorous mammal [relative units].

Transfer from ingestion of soil is also modelled, uptake from soil, UC_{soil}, is defined by:

$$UC_{soil} = f_{sc} * \frac{DMI_H}{M_H} * A_{topsoil} * f1_s \quad (2.14)$$

where:

fsc is the fraction of the daily matter intake by carnivorous mammals that is soil [relative units] = 0.06 [140]; and
$f1_s$ is the fractional gut uptake of soil for carnivorous mammal [relative units].

Table 2.2 presents the values of f_{1H}, f_{1C} and f_{1S} used within the FASTer model.

Table 2.2. Fractional gut uptake values used in the FASTer model.

Nuclide	$f1_H$	$f1_C$	$f1_S$	Reference
Cs	1	1	0.1	[45, 148]
Sr	0.2	0.2	0.2	[45, 148]
Pu	0.0005	0.0005	0.0001	[45, 148]
Am	0.0005	0.0005	0.0001^{++}	[45, 148]

$^{++}f1_S$ assumed to be the same as values presented for Pu by Beresford et al. [45].

2.13.1.1. Loss rate defined by RH for herbivore and RC for carnivore

The following equation defines the depuration rate from biota:

$$R_H = \frac{\ln(2)}{Tb_H} * A_H, \quad R_C = \frac{\ln(2)}{Tb_C} * A_C \tag{2.15}$$

where:

Tb_H and Tb_C are the biological half-life for herbivorous and carnivorous mammals respectively [d]; and
A_H and A_C are the radionuclide inventories in the body of herbivorous and carnivorous mammals respectively [Bq].

2.13.1.2. Parameters used for allometric relationships

Allometric relationships have been used to derive fresh and dry matter ingestion rates:

where:

*(Fresh or Dry) Matter Intake (kg d^{-1}) = a *(X)b*

where:

a (kg d^{-1}) and b are constants;
X = dimensionless variable equal to the Mass (kg) of the animal.

Allometric relationships have also been used to derive biological half-lives (Tb) and thereby loss rates from the animals:

$$Tb = a_2 * X^{b2} \tag{2.16}$$

where:

a_2 is the multiplication constant in the allometric relationship for biological half-life [d];
b_2 is the exponent in the allometric relationship for biological half-life [r.u.]; and
X = dimensionless variable equal to the Mass (kg) of the animal.

Values of a_2 and b_2 relevant to the work described here are presented in Table 2.3.

Table 2.3. Parameters of the allometric relationship for the biological half-life (D).

Nuclide	a_2	b_2	Reference
Cs	13.22	0.237	[75]
Sr	645	0.26	[21]
Pu	1140	0.731	FASSET [26]
Am	1140	0.731	FASSET [26]

2.14. D-Max

This approach (developed at the University of Portsmouth, UK) was not applied to the two initial model-model inter-comparison exercises (Chapters 3 and 4) but results were submitted to both the aquatic (Chapter 5) and terrestrial (Chapter 6) ecosystem scenarios.

Limiting concentrations of a range of radionuclides in environmental media (soil, water, sediments) are calculated such that the predicted total dose to any organism in that environment is predicted to be the given screening value. In the model, maximum dose rates are calculated assuming a uniformly contaminated object which is infinite in extent. In this method external and internal dose rates are not summed, but, for each radionuclide, are compared and the higher of the two is used to calculate the total dose. This gives a conservative estimate of the maximum dose rate arising in a contaminated environment. Whilst tending to over-estimate dose rates (particularly for gamma emissions), the approach does not need to consider organism geometries or occupancy factors. By calculating the maximum dose rate arising in the soil, sediment, water or tissue, the model implicitly accounts for any changes in size and habitat occupancy.

2.14.1.1. Calculation of maximum dose rate arising in a contaminated environment

In an object which is infinite in extent and uniformly contaminated by a radionuclide, the average energy deposited (per unit mass or volume) at any point in that object is equal to the average energy generated at any point (per unit mass or volume) (e.g. [76, 77]). The rate of energy deposition (dose rate), E (mGy d^{-1}) in an object uniformly contaminated with concentration C (Bq kg^{-1} (fw)) of any radionuclide is given by:

$$E = C \times 60 \times 60 \times 24 \times 1000 \times 1.6 \times 10^{-19} \sum_i \varepsilon_i = 1.38 \times 10^{-11} \times C \times \sum_i \varepsilon_i \quad (2.17)$$

where:

ε_i is the mean energy in electron volts (1eV = 1.6×10^{-19} J), of the i^{th} radiation emitted (weighted by intensity) when the radionuclide undergoes decay.

For primarily alpha-emitting radionuclides, the above equation can be applied to calculate internal doses using the (measured or predicted) internal concentration, C. For beta- and gamma-emitting radionuclides, the (measured or predicted) activity concentration either in biota or the environmental medium (water, soil, sediment) may be used to determine the highest dose-rate in an environment [77]. Table 2.4 presents values of dose conversion coefficients based on the rate of energy absorbed per Bq kg^{-1} in an infinitely extended medium.

Table 2.4. Conversion factors for calculating dose in an object which is infinite in extent and uniformly contaminated by the given radionuclide as used by D-Max.

Radionuclide	DCC (μGy h^{-1} per Bq kg^{-1})	Radionuclide	DCC (μGy h^{-1} per Bq kg^{-1})
^{3}H	3.40×10^{-6}	^{238}Pu	3.15×10^{-3}
^{60}Co	1.50×10^{-3}	^{239}Pu	2.96×10^{-3}
$^{90}Sr/^{90}Y$	6.50×10^{-4}	^{240}Pu	2.94×10^{-3}
$^{137}Cs/^{137m}Ba$	4.67×10^{-4}	^{241}Am	3.13×10^{-3}

2.14.1.2. Transfer of radionuclides to organisms

Approaches used to estimate activity concentrations are selected to yield conservative estimates.

The activity concentration in plants is determined using soil-plant CR values.

2.14.1.3. Concentration ratios for mammals and birds

The transfer of radionuclides from an (typically farm) animal's diet to tissue is most often expressed as the equilibrium transfer coefficient (F_f, d kg^{-1}), defined as the ratio of the activity concentration in a tissue to the rate of radionuclide ingestion:

$$F_f = \frac{Activity\ conc.\ in\ tissue,\ Bq\,kg^{-1}}{Radionuclide\ ingestion\ rate,\ Bq\,d^{-1}} = \frac{C_f}{C_v I_f} \tag{2.18}$$

or

$$C_f = F_f C_v I_f \tag{2.19}$$

where:

C_f is the activity concentration in the tissue (Bq kg^{-1} (fw));
C_v is the activity concentration in its food (Bq kg^{-1} dry weight (dw)); and
I_f (kg d^{-1} (dw)) is the feed intake rate.

This can be rearranged to give equilibrium Concentration Ratio ($CR_{p\text{-}a}$, kg kg^{-1}), the ratio of activity concentration in tissue (fresh weight) to that in feed:

$$CR_{p-a} = \frac{Activity\ conc.\ in\ tissue,\ Bq\,kg^{-1}}{Activity\ conc.\ in\ \ feed,\ Bq\,kg^{-1}} = \frac{C_f}{C_v} = F_f I_f \tag{2.20}$$

For herbivorous mammals and birds, the soil-animal concentration ratio, $CR_{s\text{-}a}$, is then given by:

$$CR_{s-a} = \frac{Activity\ conc.\ in\ animal,\ Bq\,kg^{-1}}{Activity\ conc.\ in\ soil,\ Bq\,kg^{-1}} = CR_{s-p} \times CR_{p-a} \tag{2.21}$$

where: $CR_{s\text{-}p}$ is the soil plant concentration ratio.

For radiocaesium there is considerable evidence that there is bioaccumulation at higher trophic levels (e.g. [78]). It is also assumed that there is prey-predator bioaccumulation of Sr. Such bioaccumulation, however, is not expected to be greater than one order of magnitude since concentrations of their stable analogues, potassium and calcium, are similar in the bodies of predators and their prey. A prey-predator CR of 2 is assumed. Plutonium, americium and cobalt are not expected to bioaccumulate at high trophic levels (rather, the opposite; [79, 80], so, conservatively, the same concentration ratio is applied to omnivorous/predatory birds and mammals as for herbivorous species.

2.14.1.4. Concentration ratios for reptiles, amphibians and insects

The water-animal CRs for amphibians are assumed to be the same as those for predatory and omnivorous fish (see below). For terrestrial reptiles and insects (and for amphibians in cases where soil but no data on water activity concentrations is available), concentration ratios are assumed to be the same as those for herbivorous or omnivorous/predatory mammals, based on their feeding habits.

2.14.1.5. Concentration ratios in fish

With the exception of Sr and Cs the activity concentrations in fish are estimated using CR values [34, 81, 82]. For Cs and Sr available models to predict the water-fish CR using relationships with potassium and calcium concentration of the surrounding water are used (e.g. [81, 83, 84]).

2.14.1.6. Concentration ratios for mammals and birds feeding on aquatic biota

It is assumed that there is no significant biomagnification of these radionuclides in mammals and birds feeding on fish or aquatic invertebrates. Thus the limiting concentration is determined by concentrations in fish, aquatic plants or invertebrates.

2.15. LAKECO-B

LAKECO-B is a dynamic model for assessing the behaviour of radionuclides in freshwater systems. The model was developed by NRG (the Netherlands) and was applied to the freshwater scenario only (Chapter 5).

The original abiotic model principles of LAKECO–B are based on equations used to assess the behaviour of radiocaesium and plutonium in marine systems [85]. The biotic part is based on the uptake of mercury by fish in freshwater systems [86]. After several validation exercises, the important parameters of the model were substituted by predictive sub-models by which these parameters were estimated on the basis of easily available environmental parameters or lake morphological parameters [87], which reduced the amount of input parameters significantly.

The original lake ecosystem model LAKECO–B model has previously been tested against ^{137}Cs data [87–92] for fish, sediments, and water of various lake systems in Europe. Because the differential equations are based on physico-chemical and biological properties, the model is generally applicable for any radionuclide. After modifying the original model based on the principle of accumulation in the target – tissue, validation tests were performed on the uptake of ^{90}Sr in lakes in Finland and Russia [93]. For the marine environment the biota model was transformed and extended to include crustaceans and molluscs [94] and implemented in the coastal dispersion model POSEIDON [95].

2.15.1. Transfer module

The transfer of radionuclides to and from aquatic organisms is described as:

$$\frac{dC_{(pred)}}{dt} = a\, k_1 C_f + b\, K_w C_w(t) - K_{0.5} C_{(pred)} \qquad (2.22)$$

where:

$C_{(pred)}$ is the concentration of the radionuclide in the organism in Bq kg^{-1} whole-body;
C_f the concentration in the food in Bq kg^{-1} and C_w the concentration of the radionuclide in water in Bq m^{-3};
K_1 is the food consumption rate in (kg prey)/(kg predator, day) where $K_1 = K_{resp} + K_{growth}$;
K_{resp} is the respiration rate;
K_{growth} is the growth rate coefficient;
a is the food extraction efficiency;
b the water extractability;
K_w the water uptake rate in m^3 d^{-1}; and
$K_{1/2}$ the elimination rate.

The concentration in the food of a predator can be expressed by the equation:

$$C_f = \sum_{i=1}^{n} C_{prey,i} \; P_{prey,i} \; \frac{dw_{pred}}{dw_{prey,i}} \tag{2.23}$$

where:

$C_{prey,i}$ is the concentration in prey i in Bq kg^{-1} (fw);
$P_{prey,i}$ is the preference for the prey (ranging between 0-1); and
dw_{pred} is the dry weight fraction of the predator and $dw_{prey,i}$ the dry weight fraction of prey i.

For radionuclides which are not homogenously distributed throughout the organism the concentration in the prey can be described by the following equation:

$$C_{prey} = \sum_{k=1}^{n} C_k \; f_k \tag{2.24}$$

where:

C_{prey} is the concentration in a given prey;
f_k the weight fraction of tissue k; and
C_k the concentration of the nuclide in tissue k.

Each tissue has its own set of parameters; biological half-life, and storage preference (i.e. the distribution of the nuclide within the organs). Multiple tissues are modelled as boxes, with transfers to and from the other tissues. A simplification is modelling the various tissues with *n*, identical differential equations, assuming that the fish consist of *n* independent boxes.

A simplification is used if a radionuclide accumulates in one specific tissue, which has a fraction f of the total body weight:

$$C_f = \sum_{i=1}^{n} f_{i,k} \; C_{prey,i,k} \; P_{prey,i} \; \frac{dw_{pred}}{dw_{prey,i}} \tag{2.25}$$

where:

f_k is the weight fraction in the prey of the tissue k, where the radionuclide is accumulated (target tissue).

For application in decision support systems and for simplification reasons, the complex foodweb is simplified (see Figure 2.2). Consequently the food preference equals 1 as each predator is assumed to consume one prey type.

2.15.2. Dose module

2.15.2.1. Alpha dose rate

If it is assumed that the activity concentration in the whole organism with mass M (g) is 1 Bq kg^{-1} (fw) then, under the assumption that the total activity is concentrated in organ S with mass m_S (g), the number emitted alpha particles per second in organ S is $0.001 * M$.

This yields an emitted energy rate of $0.001 * M * \bar{E}_\alpha$ MeV s^{-1}, where $\bar{E}_\alpha$ is the average alpha energy per decay. The absorbed fraction of alpha radiation is 1, so the specific absorbed energy rate per unit mass of the source organ is $0.001 * \frac{M}{m_s} * \bar{E}_\alpha$ MeV g^{-1} s^{-1}.

The absorbed dose rate is then given by the equation:

Gy s^{-1}, which is equivalent to $5.76 \cdot 10^{-4} * \frac{M}{m_s} * \sum_\alpha y_\alpha E_\alpha$ µGy h^{-1}.

The dose rate outside the source organ is equal to zero. The weighted dose rate for the total organism is therefore $5.76 \cdot 10^{-4} * \sum_\alpha y_\alpha E_\alpha$ µGy h^{-1} per Bq kg^{-1} total body weight.

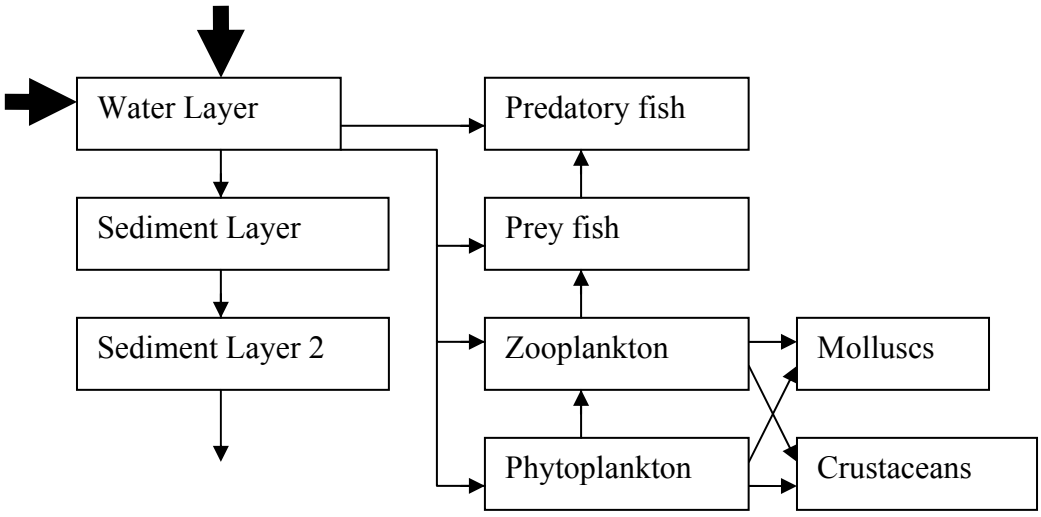

Fig. 2.2. Scheme of the simplified LAKECO-B model.

2.15.2.2. Beta dose rate

The same reasoning as for the alpha dose rate applies for the beta dose rate so that the beta dose rate is given by $5.76 \cdot 10^{-4} * \sum_{\beta} y_{\beta} \overline{E}_{\beta}$ µGy h^{-1} per Bq kg^{-1} total body weight, with $\bar{E}_{\beta}$ the average beta particle energy in MeV.

2.15.2.3. Gamma dose rate

Unlike the alpha and beta dose rate calculations, only a part of the emitted gamma energy is absorbed in the source organ, a part is absorbed in the surrounding tissues and the rest leaves the body without any interactions. The absorbed fractions depend on the energy of the gamma radiation, the size of the organs and of the total organism, and the distances between the source tissue S and the target tissues T.

Therefore, for gamma radiation the specific effective energy in tissue T is given by:

$$SEE_T = \sum_{\gamma} y_{\gamma} E_{\gamma} SAF_{T,\gamma} \text{, with } SAF_{T,\gamma} = \frac{AF_{T,\gamma}}{m_T} \tag{2.26}$$

The different absorbed fractions depend on the geometry. It is assumed as an approximation that the absorbed fractions in fish are proportional to the organ weight with respect to the organ weight of reference man.

Analogous to the alpha dose rate, the gamma absorbed dose rate in the source organ is then given by:

$$D_{\gamma,S} = 5.76 \cdot 10^{-4} * M * \sum_{\gamma} y_{\gamma} E_{\gamma} SAF^{h}_{S,\gamma} \quad \text{µGy h}^{-1} \text{ per Bq kg}^{-1}, \tag{2.27}$$

where: y_{γ} is the yield of gamma photons with energy E_{γ} per decay.

For the remainder of the body there are two options dependent upon the mass of the fish:

(1) The total mass M of the fish is smaller than the source organ mass in humans.

In this case the absorbed fraction in the remainder tissues is equal to:

$$AF^{f}_{R} = (M - m^{f}_{S}) * SAF^{h}_{S} = m^{f}_{R} * SAF^{h}_{S} \tag{2.28}$$

The absorbed dose rate in the remainder tissues is then given by:

$$D_{\gamma,R} = 5.76 \cdot 10^{-4} * M * \sum_{\gamma} y_{\gamma} E_{\gamma} SAF^{h}_{S,\gamma} \quad \text{µGy h}^{-1} \text{ per Bq kg}^{-1}. \tag{2.29}$$

Since this dose rate is equal to that in the source organ the total weighted dose rate is also given by:

$$D_{\gamma} = 5.76 \cdot 10^{-4} * M * \sum_{\gamma} y_{\gamma} E_{\gamma} SAF^{h}_{S,\gamma} \quad \text{µGy h}^{-1} \text{ per Bq kg}^{-1}. \tag{2.30}$$

(2) The total mass M of the fish is larger than the source organ mass in humans.

In this case it is assumed that the absorbed fraction in the total body is proportional to the body weight. The absorbed fraction in the remainder body is then equal to:

$$AF_R^f = M^f * SAF_{TB}^h - m_S^f * SAF_S^h \quad (2.31)$$

with: SAF_{TB} being the specific absorbed fraction in the total body.

The absorbed dose rate in the remainder tissues is then given by:

$$D_{\gamma,R} = 5.76 \cdot 10^{-4} * \frac{M}{M - m_S^f} * \sum_\gamma y_\gamma E_\gamma * (M * SAF_{TB,\gamma}^h - m_S^f * SAF_{S,\gamma}^h) \text{ µGy h}^{-1} \text{ per Bq kg}^{-1} \quad (2.32)$$

and the total weighted dose rate by:

$$D_\gamma = 5.76 \cdot 10^{-4} * M * \sum_\gamma y_\gamma E_\gamma * SAF_{TB,\gamma}^h \text{ µGy h}^{-1} \text{ per Bq kg}^{-1} \quad (2.33)$$

Specific absorbed fractions of gamma radiation originating from different source organs in target tissues and the total body are given for different energies in ICRP [96].

2.16. SÚJB approach

This approach for estimating absorbed DCCs uses derived dose rate formulas as published by the IAEA [73, 97]. Selected categories of organisms are represented by ellipsoid geometries which are used to estimate the absorbed fractions by numeric integration of point sources, and absorbed doses are determined from the absorbed fractions.

This approach was applied to the first model-model inter-comparison (Chapter 3) only.

CHAPTER 3. COMPARISON OF UNWEIGHTED ABSORBED DOSE RATES

3.1. Introduction

The purpose of this exercise was to perform an inter-comparison of internal and external dose conversion coefficients estimated by these approaches for selected organisms (as listed in Table 3.1) from the Reference Animals and Plants geometries as proposed by the International Commission on Radiological Protection (ICRP) [18]. The comparison was intended to establish whether the results from different approaches are reasonably comparable, thus testing the scientific rigour of the calculation of doses to biota.

The study covers a range of environmentally relevant media, allowing comparison of the underlying assumptions of the various approaches to dose calculation for different target organism-medium source configurations. The exercise was not intended to determine if the results of the different models were 'correct'. Results of the inter-comparison discussed in this chapter have been presented in Vives i Batlle et al. [98].

Eleven of the approaches described in Chapter 2 participated within this exercise: AECL, DosDiMEco, EA R&D128, ECOMOD, EDEN 2, EPIC DOSES3D, ERICA, FASSET, LIETDOS-BIOTA, RESRAD-BIOTA and SÚJB.

3.2. Description of the exercise

Participants were asked to use their methodologies to determine unweighted absorbed dose rates to the whole organism assuming either a biota activity concentration of 1 Bq kg^{-1} given on a fresh weight (fw) basis or a medium (water or soil) activity concentration of 1 Bq kg^{-1} (fw). Hence, the outputs corresponded to the modelled DCC values as determined for the whole organism only, and shall be referred to as such hereafter. Results were requested to be reported in units of μGy h^{-1} per Bq kg^{-1} (fw) (internal dose), μGy h^{-1} per Bq l^{-1} (aquatic external dose rates from water) and μGy h^{-1} per Bq kg^{-1} (fw) (external dose from sediment or soil). Five types of DCC, or "dose categories" were specified: internal exposure, and external exposure in water, in soil, on soil and in sediment. Inter-comparison of doses for organisms immersed in air was not possible due to a number of approaches not considering this and conflicting assumptions between those that did (e.g. some assume contaminated soil, others contaminated air). Unweighted dose rate estimates were requested to remove the uncertainty associated with the selection of multiplicative radiation weighting factors (accounting for radiation quality) from the comparison.

Table 3.1. Proposed ICRP organism geometries [18] for which DCCS were to be determined and habitats assumed for the exercise.

Organism	Dimension[a]			Mass	Surface	Surf./Volume	Ecosystem	Habitat
	a (cm)	b (cm)	c (cm)	(g)	(cm^2)	(cm^{-1})		
Duck	30	10	8	1.3E+03	6.3E+02	5.0E-01	Freshwater	S, OW
Frog	8	3	2.5	3.1E+01	5.2E+01	1.7E+00	Freshwater	S, IW
Salmonid egg	0.25	0.25	0.25	8.2E-03	2.0E-01	2.4E+01	Freshwater	BI, IW
Rat	20	6	5	3.1E+02	2.5E+02	7.9E-01	Terrestrial	U, S
Earthworm (elongated)	10	1	1	5.2E+00	2.5E+01	4.7E+00	Terrestrial	U, S

Notes: a, b, c: major, minor and second minor axis of ellipsoid; S: shore/soil surface; OW: on water; IW: in water; BI: benthic interface; U: underground.

The following five Reference Animals, as proposed by the ICRP [18], were considered: duck, frog, salmonid egg, rat and elongated earthworm. The dimensions of these organisms are given in Table 3.1 together with the habitat to be assumed. The organisms were chosen to represent different geometry size classes ranging from very small (salmonid egg) to medium-sized (duck). The shapes proposed are all ellipsoids with the exception of the elongated earthworm, which is considered (by the ICRP) as a cylinder. DCCs for each geometry were determined for seven radionuclides: ^{3}H, ^{14}C, ^{60}Co, ^{90}Sr, ^{137}Cs, ^{238}U and ^{241}Am, selected so as to cover a range of energies and different types of radiation.

Where alternatives were not in-built into the method of calculation, the exercise recommended the following assumptions. Firstly, the radionuclide distribution in the media for organisms living in soil should be uniform to a depth of 50 cm. Secondly, organisms in soil should be set to a depth of 25 cm. Thirdly, the radionuclide distribution in the media for organisms living on soil should be uniformly contaminated to a depth of 10 cm.

An effort was made to maximise independence among the approaches while the inter-comparison was in progress. There was open discussion about the different approaches prior to the exercise. However, all model runs were performed independently, and submitted to an independent analyst/data custodian who was not involved in the running of any of the models. Statistical analysis results were disseminated after a final submission, whereupon discussion occurred without the possibility for resubmission of results after learning the outcome of this inter-comparison.

Participants provided details of which radionuclide progeny had been included in the calculation of DCCs and on what basis, together with all relevant geometry and radionuclide assumptions (Tables 3.2–3.3). With the exception of EDEN 2 (see Section 2.11), all approaches used ICRP publication 38 [99] for nuclide specific energies and their emission probabilities.

3.3. Statistical analysis methodology

The data submitted by the participants were processed using the R software for Statistical Computing, version 2.3.0 [100]. Of the 25 packages supplied with R, the “Moments” package [101] was used here.

For any dose category, determinations were supplied by each of the participants for every radionuclide and organism (where included in their approach). To identify a central value for a given parameter to enable comparison of the different approaches, the following analysis strategy was adopted:

- Conduct initial exploratory data analysis to identify outliers.
- Perform statistical distribution tests on the remaining data.
- Calculate a robust mean and standard deviation for the parameter.
- Score each approach for performance.

Outliers in this exercise are identified from a purely statistical perspective, as there are no experimentally measured or previously agreed reference DCC values available. Likewise, no assumption is made that the mean from all predictions is the most accurate prediction. In the absence of reference data, the statistical methodology used in this study is simply a way to compare the outputs for all models.

3.3.1. Exploratory data analysis

Assuming the absence of any systematic bias for each individual approach, all results should follow a simple statistical distribution. If the distribution is normal, then the reported values should lie, with 95% probability, within two standard deviations (2-σ) uncertainty range of the calculated reference value. In practice, outliers straying well outside this range are found. Initial exploratory data analysis to assess this was conducted using a "box plot" diagram. The box plot displays a measure of central tendency (the median), two measures of dispersion (the range and inter-quartile range), the skewness (from the orientation of the median relative to the quartiles) and potential outliers (marked individually). Given the large number of data to be processed, the key advantage of the box plot over numerical methods (such as Grubb's, extreme studentised deviate, Dixon's or Rosner's tests) is ready outlier visualisation.

3.3.2. Normality tests

A preliminary test (Shapiro-Wilk) was performed to determine whether the results were normally distributed. Having found that to be the case, data were subjected to D'Agostino's test for skewness and the Anscombe-Glynn's test for kurtosis[3]. These tests are powerful at detecting deviations from normality caused by asymmetry or non-normal tail heaviness, respectively, by computing a p-value from the sum of the squares of these discrepancies.

The above tests differ in how they quantify the deviation from normality, but they test the null hypothesis – that the data are sampled from a normal distribution. The null hypothesis is rejected, and the alternative hypothesis accepted, when the p-value is small. If the distribution is normal, the p-value will tend to be large:

- p-values > 0.10 indicate no evidence against the null hypothesis;
- 0.05 < p < 0.10 indicates weak evidence against the null hypothesis in favour of the alternative hypothesis;
  - < p < 0.05 indicates moderate evidence against the null hypothesis in favour of the alternative hypothesis;
  - < p < 0.01 is indicative of strong evidence against the null hypothesis in favour of the alternative hypothesis; and
- p < 0.001 indicates very strong evidence against the null hypothesis in favour of the alternative hypothesis.

For this work, p > 0.05 was chosen as the criterion for passing the normality test.

D'Agostino's and Anscombe-Glynn's tests were chosen as tests in preference to the Kolmogorov-Smirnov or the Shapiro-Wilk test alone because the latter are generally less powerful than tests specifically designed to assess the shape of a distribution [102].

Any data point failing one or more tests was identified as an outlier and the remaining data were re-tested without it until no further outliers were detected and the eventual residual data were found to conform to a normal distribution. From this the robust mean and associated standard deviation were estimated.

[3] Skewness and kurtosis are measures of the lack of symmetry and the heaviness of the tails in a distribution, relative to the normal distribution.

Table 3.2. Geometry assumptions of the participating approaches in this and subsequent tbales, England and Wales Environment Agency 'R&D128', EDEN 2, EPIC DOSES3D, LIETDOS-BIO and RESRAD-BIOTA are abbreviated to EA, EDEN, EPIC, LIETDOS and RESRAD, respectively.

Geometry used	As proposed by the ICRP: EA, EDEN, EPIC, ERICA, LIETDOS, RESRAD, DosDiMEco, SUJB Nearest default geometries: AECL, ECOMOD, FASSET			
Assumed tissue/org. density	1 g cm^{-3}: EA, ECOMOD, EDEN, EPIC, RESRAD, DosDiMEco, SUJB; 1.05 g cm^{-3} AECL, ERICA, FASSET, LIETDOS			
Scenario	**Media density (g cm^{-3})**	**Uniform contamination assumed**	**Media depth**	**Assumed location of organism**
Underground	1 - EA, 1.35 - DosDiMEco, 1.5 - EPIC, 1.6 - AECL, EDEN, ERICA, FASSET, LIETDOS, RESRAD, SUJB	AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco, SUJB	Infinite - EA, ECOMOD, EPIC, SUJB, 50 cm – AECL, EDEN, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco	Fully immersed - EA, ECOMOD, EPIC, DosDiMEco, SUJB, 25 cm depth – AECL, EDEN, ERICA FASSET, LIETDOS, RESRAD
On soil surface	1 - EA, 1.35 - DosDiMEco, 1.5 - EPIC, 1.6 - AECL, EDEN, ERICA, FASSET, LIETDOS, RESRAD, SUJB	AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco, SUJB	Infinite - EA, ECOMOD, SUJB, 5 cm - EPIC, 10 cm - AECL, EDEN, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco	On soil surface - AECL, EA, ECOMOD, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco, SUJB
Benthic interface	1 - EA, ERICA, FASSET, SUJB, 1.2 - RESRAD, 1.35 - DosDiMEco, 1.5 - EPIC, 1.6 - AECL, EDEN, LIETDOS	AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco, SUJB	Infinite - AECL, EA, ECOMOD, ERICA, FASSET, SUJB, 5 cm - EPIC, 10 cm – EDEN, LIETDOS, RESRAD, DosDiMEco.	At water/sediment interface – AECL, EA, ECOMOD, EDEN, EPIC, ERICA, FASSET, LIETDOS, DosDiMEco, SUJB, On sediment surface - RESRAD
In water	1 - AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, SUJB	AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco, SUJB	Infinite - AECL,EA, ECOMOD, EPIC, ERICA, FASSET, SUJB, 100 cm - LIETDOS, 1000 cm - EDEN	Fully immersed - AECL, EA, ECOMOD, EPIC, ERICA, FASSET, SUJB, 50 cm - LIETDOS, 500 cm – EDEN
On water	1 - AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, DosDiMEco, SUJB	AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, DosDiMEco, SUJB	Semi-infinite - EPIC, Infinite - AECL, EA, ECOMOD, ERICA, FASSET, SUJB, 10 cm - RESRAD, 100 cm - EDEN, 200 cm - DosDiMEco	At water/air interface - AECL, EA, ECOMOD, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco, SUJB

Table 3.3. Radionuclide assumptions of the participating approches.

Nuclide	Daughters in equilibrium	Ext. α zero energy abs. assumed[f]	Ext. β zero energy absorption[a,g]	100 % int. α absorption[f]	100 % int. β absorption[g]	No. of α decays used[b,e]	No. of β decays used[b,e]	No. of γ decays used[b,d,e]
^{3}H			ECOMOD , EPIC, LIETDOS, SUJB		ECOMOD, EPIC, LIETDOS, SUJB	0 - AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco	1 - AECL, EA, ECOMOD, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco	0 - AECL, EA, EDEN, EPIC, ERICA, FASSET, DosDiMEco, 1 - LIETDOS
^{14}C			ECOMOD, LIETDOS		ECOMOD, EPIC, LIETDOS, SUJB	0 - AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, DosDiMEco	1 - AECL, EA, ECOMOD, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco	0 - AECL, EA, EDEN, EPIC, ERICA, FASSET, DosDiMEco, 3 - LIETDOS
^{90}Sr	^{90}Y: AECL, EA, ECOMOD, EDEN, EPIC, ERICA, FASSET, LIETDOS, RESRAD, SUJB; ^{90,90m}Y: DosDiMEco		LIETDOS		LIETDOS, SUJB	0 - AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, DosDiMEco	2 - AECL, EA, ECOMOD, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco 3 - EDEN	1 - EDEN, 3 - DosDiMEco, 5 - EPIC, 7 – AECL, EA, ERICA, FASSET, 12 - LIETDOS
^{137}Cs	^{137m}Ba: AECL, EA, EDEN, EPIC, ERICA, FASSET, RESRAD, DosDiMEco, SUJB				LIETDOS, SUJB	0 - AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, DosDiMEco	1 - ECOMOD, 2 - LIETDOS, DosDiMEco, 4 - RESRAD, 8 - EDEN, 13 – AECL, EA, EPIC, ERICA, FASSET	1 - DosDiMEco, 2 - ECOMOD, 3 - AECL, EA, EPIC, ERICA, FASSET, RESRAD, 7 - EDEN, 8- LIETDOS
^{60}Co					LIETDOS, SUJB	0 - AECL, EA, EDEN, EPIC, ERICA, FASSET, LIETDOS, DosDiMEco	1 - RESRAD, 2 - DosDiMEco, 3 – EDEN, LIETDOS, 4 - AECL, EA, EPIC, ERICA, FASSET	2 - AECL, EA, ECOMOD, EPIC, ERICA, FASSET, LIETDOS, RESRAD, DosDiMEco, 6 - EDEN
^{241}Am		AECL, EA, ECOMOD, EPIC, ERICA, FASSET, RESRAD, DosDiMEco, SUJB	EPIC, LIETDOS	AECL, EA, ECOMOD, EDEN[c], EPIC, ERICA, FASSET, LIETDOS, RESRAD, SUJB	EPIC, LIETDOS	5 – AECL, EA, ERICA, FASSET, LIETDOS, DosDiMEco, 10 - EPIC, 42 - EDEN	0 - DosDiMEco, 1 - ECOMOD, 5 - RESRAD, 13 - LIETDOS, 37 - AECL, EA, EPIC, ERICA, FASSET, 233 - EDEN	3 – ECOMOD, DosDiMEco, 5 - RESRAD, 8 - AECL, EA, EPIC, ERICA, FASSET, 12 - LIETDOS, 136 - EDEN
^{238}U	0: EPIC, SUJB, ^{234}Th, ^{234}Pa: ECOMOD, ^{234}Th, ^{234,234m}Pa: EA, EDEN, ERICA, RESRAD, DosDiMEco ^{234}Th, ^{234,234m}Pa, ^{234}U: AECL, FASSET	AECL, EA, ECOMOD, EPIC, ERICA, FASSET, RESRAD, DosDiMEco, SUJB	EPIC, LIETDOS	AECL, EA, ECOMOD, EDEN[c], EPIC, ERICA, FASSET, LIETDOS, RESRAD, SUJB	LIETDOS	3 - EA, EDEN, LIETDOS, DosDiMEco, 6 - EPIC	2 – ECOMOD, 4 - LIETDOS, RESRAD, 10 – AECL, EPIC, 26 – DosDiMEco, 109 - EA, 880 - EDEN	2 - ECOMOD, 4 - LIETDOS, RESRAD, 6 - EPIC, 147 - DosDiMEco, 170 - EA, 372 - EDEN

Notes: [a] For DoSDiMEco ^{3}H and ^{14}C: Yes; for DoSDiMEco ^{60}Co, ^{90}Sr and ^{137}Cs: No (except animals on water); [b] SUJB average energies used; [c] Except for salmon egg and earthworm; [d] EDEN includes X-rays; [e] SUJB average energies given; [f] Not assumed for salmon egg or bacteria; [g] Not assumed for salmon egg.

3.3.3. Calculation of reference data and scoring of each approach for performance

The performance of the participating approaches was assessed by comparing reported results with the estimated reference values, using a "Z-score", which is a measure of how many standard deviation units away from the mean a particular data value lies [103]. This approach represents a simple method to give each approach a normalised performance score for bias. The performance is considered satisfactory if a relative bias is equal to or better than 25% (absolute value of Z is between 0 and 2). Z-values between 2 and 3 indicate that the results are more biased, and Z-values ≥ 3 indicates that the measurements are highly biased [104]. This scoring system is now included in the International Organisation for Standardisation guidelines as a standard method for laboratory assessment [105] and has been used successfully by the IAEA in previous inter-comparisons [106].

In order to assess the overall performance of the approaches on a dose category basis, an "efficacy measure" was computed, as suggested by Lawn et al. [104]. This efficacy measure is defined as the "percentage of approaches producing results of acceptable quality" (i.e. with absolute value of Z between 0 and 2) and was calculated for each of the five dose categories (internal exposure, and external exposure in soil, water, sediment and on soil/shore).

3.3.4. Issues in results interpretation

The assumptions and calculation differences embedded within each approach are too numerous to enable a concise and systematic presentation of the data. Mean values and their confidence intervals were, therefore, statistically derived from data satisfying certain criteria with no regard to their inherent quality. This approach is not a substitute for expert opinion in cases where the data are more skewed with a greater spread.

No value judgement is, therefore, passed on outlying values. Statistical tests have limited power for screening such values out. When 'inaccurate' results outweigh 'accurate' ones numerically, the few good data may be rejected.

In the case of ^{3}H and ^{14}C external doses, a significant number of participants reported values as 'zero', whilst some reported a numerically small non-zero value (Tables 3.5–3.8). Consequently, the ^{3}H and ^{14}C external doses were excluded from consideration in the statistical analysis.

A limited number of cases, similarly requiring deviation from the basic analysis procedure, were treated individually. These were outputs when a number of models made different methodological assumptions (e.g. number of daughter radionuclides or source / target geometry), resulting in greater statistical spread than allowed by a simple normal distribution, as described in the Section 3.4.1.

In terms of evaluating Z-scores for every approach, it is important to exercise caution. The approaches are not absolutely independent from each other in terms of how the DCC is calculated. For example, a minority of the approaches adopt common DCC values in some instances. However, generally, there are variations in respect of radionuclide assumptions and even wider variations in the definitions of source-target geometries for external doses (e.g. media density, tissue/organism density, media depth, location of organism in media). All these varying assumptions (summarised in Tables 3.2 and 3.3) contribute to the variability in DCCs observed in this study, particularly for external exposure, with no individual set of assumptions being absolutely right or wrong. Therefore, in the present study, it was decided

to refrain from passing value judgements on each specific approach based on ranking their performance by means of Z-scores.

As some of the approaches (e.g. EDEN, ERICA, EPIC DOSES3D) were undergoing development, the results presented below, though they represent the current state of the art at the time of the exercise, may not be definitive.

3.4. Results

Calculated DCCs for internal irradiation and external irradiation in water, in soil, on soil and in sediment are given in Tables 3.4–3.8, respectively. It must be noted that the approaches used here have been applied by specific participants who were either involved in the development of the approach or its use in assessments. Some aspects of some approaches may be open to interpretation.

On initial inspection, internal exposure DCCs for the different approaches are relatively homogeneous; typically coefficients of variation (CVs)[4] are about 23% of the mean (range between 4 and 59%). Coefficients of variation between different approaches are greater for external exposure DCCs. Here, typical CVs are around 120% of the mean (range between 29 and 280%). Whilst external DCCs from ^{238}U are low, this radionuclide, along with ^{3}H and ^{14}C, were found to contribute the most to the variability; as noted above external DCCs for ^{3}H and ^{14}C were excluded from subsequent analyses. Without ^{3}H, ^{14}C and ^{238}U, typical CVs for external irradiation would be significantly reduced to around 71% of the mean (range between 29 and 230%).

Some groupings of apparently 'anomalous' DCCs stand out. Internal exposure DCCs for ^{238}U and ^{241}Am are consistently low for DosDiMEco. For external exposure DCCs in water, values from DosDiMEco for ^{90}Sr (reported as zero), as well as ^{238}U (significantly higher), are in contrast with the rest. External DCCs for ^{90}Sr reported by the ERICA and related FASSET approaches are lower for terrestrial organisms than those of other approaches. This is likely to be a consequence of the consideration of a shielding skin/fur layer within these approaches (for terrestrial but not aquatic organisms).

For external exposure DCCs in soil, EPIC DOSES3D reports comparatively high ^{137}Cs and ^{60}Co values. This may be due to the combined effect of infinite absorbing medium and, to a lesser extent, differing density assumptions (1.5 g cm^{-3}) compared with most other approaches, as shown in Table 3.2. To illustrate this, note that the density assumed in EPIC DOSES3D is 50% higher than that assumed by the England and Wales Environment Agency R&D128 methodology, and the DCCs differ in a similar proportion. Internal exposure DCC values for aquatic organisms for ^{238}U from FASSET are higher than those of all other approaches (with the exception of AECL, which adopts the FASSET values in these instances). FASSET assumes ^{234}U is in secular equilibrium with ^{238}U.

3.4.1. Identification of outliers

Identification of outliers using the box plot method is depicted in Figures 3.1 and 3.2, illustrating examples for internal irradiation and external irradiation in water, respectively. As the number of data contained in the box plot increases, there is an increasing likelihood that

[4] The CVs in Tables 3.4 – 3.8 are calculated using the raw rather than the robust mean and standard deviation of data, differing in this respect from efficiency measures. For this reason, CVs and Z-scores as given in this paper are not directly comparable.

data points may appear just slightly outside the box plot's upper and lower limit (quartile ± 1.5 × interquartile range). Grubb's outlier testing was applied, where necessary, to confirm that any such data points were, in fact, genuine outliers.

It is a general conclusion from the data set as examined that most outlier-stripped data follow normal distributions with a varying degree of skewness. This was confirmed by further statistical analysis, as described in the Section 3.3. Occasional exceptions to this were cases where the outlier-stripped data subset contained identical values, adversely affecting the Shapiro-Wilk test.

Although 11 approaches result in a relatively small sample size of 11 DCCs to be statistically analysed for each dose category/organism/radionuclide combination, normal distributions were consistently observed over some 90 separate samples, justifying a statistical analysis based on normality tests. A tendency to register normal rather than flat distributions implies that results tend to converge around some central value. This suggests that most of the approaches calculate a similar value of the DCC, with some random variation.

Deviations from normality observed during outlier identification and removal were as follows:

— All ^{3}H internal dose data were found to fail normality tests, but not Grubb's test for outliers. This is because for ^{3}H, all approaches except ECOMOD, EDEN and DosDiMEco generate a DCC of 3.3×10^{-6} μGy h^{-1} per Bq kg^{-1}, as seen in Table 3.4. The data are therefore not normally distributed, as there is significant repetition of a single value. Moreover, the ECOMOD, EDEN and DosDiMEco determinations are close to the values reported by the other participants.

— For internal doses, there is a case, namely ^{238}U for frog, for which there is some evidence against the null hypothesis of normality. The internal ^{90}Sr DCC for frog and ^{238}U DCC for rat and earthworm, with p-values of 7.4×10^{-3}, 1.5×10^{-3} and 2.8×10^{-3}, respectively, shows stronger evidence against the null hypothesis in the Shapiro-Wilk test. However, in all these cases, skewness and kurtosis tests are passed. To investigate this anomaly, which seems to occur when data are very closely grouped together, Grubb's outlier testing was performed. This showed that these data are not outliers.

— For external doses (water: ^{238}U for duck and frog; in soil: ^{238}U for rat and earthworm, on soil: ^{241}Am for rat and ^{238}U for earthworm), a similar anomaly (as above) was encountered. Variability in ^{238}U DCC determination is likely to have occurred due to the inclusion of different daughter decay products by different approaches, resulting in greater statistical spread which may not conform to a simple normal distribution but a multi-modal one (as discussed in the Section 3.3.4).

Table 3.4. Calculated DCCS (μGy h^{-1} per Bq kg^{-1}) for internal irradiation (data from [98]).

Nuclide	Organism	AECL	EA	ECOMOD	EDEN	EPIC	ERICA	FASSET	LIETDOS-BIO	RESRAD-BIOTA	DosDiMEco	SÚJB	Min	Max	Range[a]	CV (%)[b]
^{3}H	Duck	3.3E-06	3.3E-06	4.2E-06	5.9E-06	3.3E-06	3.3E-06	3.3E-06	3.3E-06	3.3E-06	2.4E-06	3.3E-06	2.4E-06	5.9E-06	2.4E+00	2.5E+01
	Frog	3.3E-06	3.3E-06	4.2E-06	5.9E-06	3.3E-06	3.3E-06	3.3E-06	3.3E-06	3.3E-06	2.4E-06	3.3E-06	2.4E-06	5.9E-06	2.5E+00	2.5E+01
	Salmonid egg	3.3E-06	3.3E-06	4.2E-06	5.9E-06	3.3E-06	n/a	n/a	3.3E-06	3.3E-06	2.5E-06	3.3E-06	2.5E-06	5.9E-06	2.3E+00	2.7E+01
	Rat	3.3E-06	3.3E-06	4.2E-06	5.9E-06	3.3E-06	3.3E-06	3.3E-06	3.3E-06	3.3E-06	2.4E-06	3.3E-06	2.4E-06	5.9E-06	2.5E+00	2.5E+01
	Earthworm	3.3E-06	3.3E-06	4.2E-06	5.9E-06	3.3E-06	3.3E-06	3.3E-06	3.3E-06	3.3E-06	2.9E-06	3.3E-06	2.9E-06	5.9E-06	2.1E+00	2.3E+01
^{14}C	Duck	2.8E-05	2.9E-05	2.8E-05	2.7E-05	2.8E-05	2.9E-05	2.9E-05	2.9E-05	2.9E-05	2.1E-05	2.9E-05	2.1E-05	2.9E-05	1.4E+00	8.1E+00
	Frog	2.8E-05	2.8E-05	2.8E-05	2.7E-05	2.8E-05	2.8E-05	2.8E-05	2.9E-05	2.9E-05	2.1E-05	2.9E-05	2.1E-05	2.9E-05	1.4E+00	8.1E+00
	Salmonid egg	2.8E-05	2.8E-05	4.2E-06	2.6E-05	2.8E-05	n/a	n/a	2.9E-05	2.8E-05	2.2E-05	2.9E-05	4.2E-06	2.9E-05	6.9E+00	3.2E+01
	Rat	2.8E-05	2.9E-05	2.8E-05	2.7E-05	2.8E-05	2.9E-05	2.9E-05	2.9E-05	2.9E-05	2.1E-05	2.9E-05	2.1E-05	2.9E-05	1.4E+00	8.4E+00
	Earthworm	2.8E-05	2.8E-05	2.8E-05	2.7E-05	2.8E-05	2.8E-05	2.8E-05	2.9E-05	2.8E-05	2.5E-05	2.9E-05	2.5E-05	2.9E-05	1.1E+00	3.7E+00
^{90}Sr	Duck	6.3E-04	6.3E-04	6.5E-04	6.0E-04	6.3E-04	6.3E-04	6.3E-04	6.5E-04	6.3E-04	4.7E-04	6.5E-04	4.7E-04	6.5E-04	1.4E+00	8.4E+00
	Frog	5.7E-04	5.8E-04	6.5E-04	5.6E-04	5.6E-04	5.9E-04	5.7E-04	6.5E-04	6.0E-04	4.6E-04	6.5E-04	4.6E-04	6.5E-04	1.4E+00	9.2E+00
	Salmonid egg	1.4E-04	2.0E-04	2.2E-04	2.1E-04	1.7E-04	n/a	n/a	2.6E-04	2.0E-04	3.0E-04	6.5E-04	1.4E-04	6.5E-04	4.6E+00	5.9E+01
	Rat	6.1E-04	6.2E-04	6.5E-04	5.9E-04	6.1E-04	6.2E-04	6.4E-04	6.5E-04	6.2E-04	4.6E-04	6.5E-04	4.6E-04	6.5E-04	1.4E+00	8.8E+00
	Earthworm	5.1E-04	4.7E-04	5.3E-04	4.7E-04	4.4E-04	5.2E-04	5.1E-04	5.3E-04	5.1E-04	5.5E-04	6.5E-04	4.4E-04	6.5E-04	1.5E+00	1.0E+01
^{137}Cs	Duck	1.8E-04	1.8E-04	1.9E-04	1.8E-04	1.9E-04	1.9E-04	1.8E-04	2.1E-04	1.9E-04	2.5E-04	1.6E-04	1.6E-04	2.5E-04	1.5E+00	1.1E+01
	Frog	1.5E-04	1.5E-04	1.5E-05	1.5E-04	1.6E-04	1.5E-04	1.5E-04	1.6E-04	1.6E-04	1.3E-04	1.2E-04	1.5E-05	1.6E-04	1.1E+01	3.1E+01
	Salmonid egg	7.9E-05	9.5E-05	1.0E-04	1.0E-04	1.1E-04	n/a	n/a	1.3E-04	1.0E-04	8.5E-05	1.1E-04	7.9E-05	1.3E-04	1.6E+00	1.4E+01
	Rat	1.6E-04	1.7E-04	1.7E-04	1.6E-04	1.7E-04	1.7E-04	2.0E-04	1.8E-04	1.7E-04	1.9E-04	1.4E-04	1.4E-04	2.0E-04	1.4E+00	8.8E+00
	Earthworm	1.4E-04	1.4E-04	1.5E-04	1.3E-04	1.5E-04	1.4E-04	1.4E-04	1.4E-04	1.4E-04	1.1E-04	1.2E-04	1.1E-04	1.5E-04	1.3E+00	8.2E+00
^{60}Co	Duck	2.0E-04	2.2E-04	1.4E-04	2.2E-04	2.4E-04	2.4E-04	2.0E-04	3.0E-04	2.3E-04	5.2E-04	2.9E-04	1.4E-04	5.2E-04	3.7E+00	3.9E+01
	Frog	9.9E-05	1.0E-04	3.6E-05	1.1E-04	1.1E-04	1.1E-04	9.9E-05	1.3E-04	1.1E-04	2.0E-04	1.3E-04	3.6E-05	2.0E-04	5.5E+00	3.4E+01
	Salmonid egg	5.0E-05	5.4E-05	1.4E-06	5.4E-05	5.9E-05	n/a	n/a	6.1E-05	5.7E-05	5.3E-05	6.0E-05	1.4E-06	6.1E-05	4.3E+01	3.7E+01
	Rat	n/a	1.6E-04	1.0E-04	1.6E-04	1.7E-04	1.7E-04	n/a	2.1E-04	1.7E-04	3.6E-04	2.1E-04	1.0E-04	3.6E-04	3.6E+00	3.8E+01
	Earthworm	n/a	7.6E-05	1.8E-05	7.4E-05	7.9E-05	7.7E-05	n/a	7.7E-05	7.8E-05	1.1E-04	9.4E-05	1.8E-05	1.1E-04	6.3E+00	3.3E+01
^{241}Am	Duck	3.2E-03	3.2E-03	3.2E-03	3.0E-03	3.2E-03	3.2E-03	3.2E-03	3.2E-03	3.2E-03	1.2E-03	3.2E-03	1.2E-03	3.2E-03	2.7E+00	2.0E+01
	Frog	3.2E-03	3.2E-03	3.2E-03	3.0E-03	3.2E-03	3.2E-03	3.2E-03	3.2E-03	3.2E-03	1.2E-03	3.2E-03	1.2E-03	3.2E-03	2.8E+00	2.0E+01
	Salmonid egg	3.2E-03	3.2E-03	3.2E-03	3.0E-03	3.2E-03	n/a	n/a	3.2E-03	3.2E-03	1.2E-03	3.2E-03	1.2E-03	3.2E-03	2.7E+00	2.2E+01
	Rat	3.2E-03	3.2E-03	3.2E-03	3.0E-03	3.2E-03	3.2E-03	3.2E-03	3.2E-03	3.2E-03	1.2E-03	3.2E-03	1.2E-03	3.2E-03	2.8E+00	2.0E+01
	Earthworm	3.2E-03	3.2E-03	3.2E-03	3.0E-03	3.2E-03	3.2E-03	3.2E-03	3.2E-03	3.2E-03	1.4E-03	3.2E-03	1.4E-03	3.2E-03	2.3E+00	1.8E+01
^{238}U	Duck	5.7E-03	2.9E-03	3.0E-03	2.8E-03	2.5E-03	2.4E-03	5.7E-03	2.4E-03	2.5E-03	1.8E-03	2.5E-03	1.8E-03	5.7E-03	3.2E+00	4.3E+01
	Frog	5.7E-03	2.9E-03	3.0E-03	2.7E-03	2.5E-03	2.4E-03	5.7E-03	2.4E-03	2.5E-03	1.5E-03	2.5E-03	1.5E-03	5.7E-03	3.9E+00	4.5E+01
	Salmonid egg	5.3E-03	2.6E-03	2.9E-03	2.4E-03	2.4E-03	n/a	n/a	2.4E-03	2.5E-03	1.3E-03	2.5E-03	1.3E-03	5.3E-03	4.2E+00	4.0E+01
	Rat	2.4E-03	2.9E-03	3.0E-03	2.8E-03	2.5E-03	2.4E-03	2.4E-03	2.4E-03	2.5E-03	1.6E-03	2.5E-03	1.6E-03	3.0E-03	1.8E+00	1.4E+01
	Earthworm	2.4E-03	2.8E-03	3.0E-03	2.7E-03	2.5E-03	2.4E-03	2.4E-03	2.4E-03	2.5E-03	1.6E-03	2.5E-03	1.6E-03	3.0E-03	1.8E+00	1.4E+01

Notes: n/a: radionuclide/reference organism combination not included in approach. [a] Range = ratio of maximum to minimum; [b] CV (coefficient of variation) = ratio of the standard deviation to the mean, expressed as a percentage. When the mean value is near zero, the CV is sensitive to change in the standard deviation, limiting its usefulness.

Table 3.5. Calculated DCCS (μGy h^{-1} per Bq l^{-1}) fo external irradiation in water (data from [98]).

Nuclide	Organism	AECL	EA	ECOMOD	EDEN	EPIC	ERICA	FASSET	LIETDOS-BIO	RESRAD-BIOTA	DosDiMEco	SÚJB	Min	Max	Range	CV (%)
^{3}H	Duck	6.6E-11	9.2E-11	n/a	4.9E-06	zero	3.6E-13	6.6E-11	6.4E-12	4.6E-11	zero	zero	3.6E-13	4.9E-06	n/c	2.6E+02
	Frog	1.8E-10	1.6E-10	n/a	4.9E-06	zero	2.5E-12	1.8E-11	2.2E-11	5.5E-11	zero	zero	2.5E-12	4.9E-06	n/c	2.6E+02
	Salmonid egg	4.8E-09	3.4E-09	n/a	4.9E-06	1.1E-14	n/a	n/a	2.7E-10	1.3E-09	zero	zero	1.1E-14	4.9E-06	n/c	2.4E+02
^{14}C	Duck	1.5E-08	1.8E-08	n/a	5.9E-07	1.2E-16	1.8E-08	1.5E-08	2.6E-09	9.9E-09	zero	2.0E-08	1.2E-16	5.9E-07	n/c	2.5E+02
	Frog	4.8E-08	4.2E-08	n/a	6.6E-07	4.0E-10	5.9E-08	4.8E-11	8.9E-09	1.6E-08	zero	6.7E-08	4.8E-11	6.6E-07	n/c	2.1E+02
	Salmonid egg	1.4E-06	9.1E-07	n/a	1.4E-06	5.4E-07	n/a	n/a	1.0E-07	4.8E-07	zero	1.0E-06	1.0E-07	1.4E-06	n/c	5.9E+01
^{90}Sr	Duck	2.3E-05	2.2E-05	n/a	2.8E-07	1.3E-05	2.0E-05	2.3E-05	n/a	1.7E-05	zero	2.7E-05	2.8E-07	2.7E-05	n/c	4.6E+01
	Frog	8.0E-05	7.3E-05	6.4E-05	6.7E-05	9.0E-05	6.3E-05	8.0E-05	n/a	2.2E-05	zero	9.0E-05	2.2E-05	9.0E-05	n/c	2.9E+01
	Salmonid egg	5.1E-04	4.5E-04	5.1E-04	3.2E-04	4.8E-04	n/a	n/a	n/a	4.1E-04	zero	1.0E-03	3.2E-04	1.0E-03	n/c	4.2E+01
^{137}Cs	Duck	2.9E-04	2.9E-04	2.8E-04	8.9E-05	1.5E-04	2.8E-04	2.9E-04	n/a	1.3E-04	8.4E-05	1.4E-04	8.4E-05	2.9E-04	3.4E+00	4.4E+01
	Frog	3.2E-04	3.2E-04	3.1E-04	2.1E-04	3.3E-04	3.2E-04	3.2E-04	2.8E-04	1.2E-04	8.4E-05	1.7E-04	8.4E-05	3.3E-04	3.9E+00	3.6E+01
	Salmonid egg	3.9E-04	3.7E-04	3.6E-04	1.4E-04	3.8E-04	n/a	n/a	n/a	1.9E-04	6.7E-05	2.0E-04	6.7E-05	3.9E-04	5.8E+00	4.9E+01
^{60}Co	Duck	1.3E-03	1.3E-03	1.3E-03	4.2E-04	6.3E-04	1.3E-03	1.3E-03	n/a	5.7E-04	3.3E-04	6.0E-04	3.3E-04	1.3E-03	3.9E+00	4.7E+01
	Frog	1.4E-03	1.4E-03	1.4E-03	1.0E-03	1.4E-03	1.4E-03	1.4E-03	1.2E-03	5.3E-04	3.6E-04	6.9E-04	3.6E-04	1.4E-03	3.9E+00	3.6E+01
	Salmonid egg	1.5E-03	1.4E-03	1.4E-03	4.8E-04	1.4E-03	n/a	n/a	n/a	7.0E-04	3.4E-04	8.5E-04	3.4E-04	1.5E-03	4.4E+00	4.7E+01
^{241}Am	Duck	1.2E-05	1.2E-05	1.0E-05	1.2E-05	5.6E-06	1.1E-05	1.2E-05	n/a	6.6E-06	1.8E-06	7.9E-06	1.8E-06	1.2E-05	6.6E+00	3.8E+01
	Frog	1.5E-05	1.5E-05	1.2E-05	2.3E-05	1.4E-05	1.4E-05	1.5E-05	8.0E-06	7.2E-06	2.8E-06	9.0E-06	2.8E-06	2.3E-05	8.1E+00	4.3E+01
	Salmonid egg	1.9E-05	1.9E-05	1.2E-05	6.0E-05	4.0E-05	n/a	n/a	n/a	6.1E-06	1.3E-06	9.5E-06	1.3E-06	6.0E-05	4.5E+01	9.4E+01
^{238}U	Duck	2.9E-05	2.9E-05	5.6E-06	8.4E-06	4.7E-08	9.5E-08	2.9E-05	n/a	1.9E-05	2.4E-04	2.4E-07	4.7E-08	2.4E-04	5.2E+03	2.0E+02
	Frog	7.3E-05	6.7E-05	6.0E-06	6.7E-05	2.5E-07	2.7E-07	7.3E-05	1.6E-06	2.2E-05	2.5E-04	2.7E-07	2.5E-07	2.5E-04	1.0E+03	1.4E+02
	Salmonid egg	4.2E-04	3.7E-04	9.6E-05	2.9E-04	1.1E-05	n/a	n/a	n/a	3.2E-04	2.2E-04	2.9E-07	2.9E-07	4.2E-04	1.5E+03	7.6E+01

Note: in this and subsequent tables n/a: radionuclide/reference organism combination not included in the approach in question; n/c: not calculated.

Table 3.6. Calculated DCCS (μGy h^{-1} per Bq kg^{-1}) for external irradiation in soil (data from [98]).

Nuclide	Organism	AECL	EA	ECOMOD	EDEN	EPIC	ERICA	FASSET	LIETDOS-BIO	RESRAD-BIOTA	DosDiMEco	SÚJB	Min	Max	Range	CV (%)
^{3}H	Rat	zero	9.5E-11	n/a	4.9E-06	Zero	zero	zero	4.3E-11	9.5E-11	zero	zero	4.3E-11	4.9E-06	n/c	2.0E+02
	Earthworm	zero	5.5E-10	n/a	4.9E-06	Zero	zero	zero	3.0E-10	3.5E-10	zero	zero	3.0E-10	4.9E-06	n/c	2.0E+02
^{14}C	Rat	zero	2.3E-08	n/a	6.1E-07	1.0E-14	zero	zero	1.6E-08	2.3E-08	zero	6.2E-08	1.0E-14	6.1E-07	n/c	2.0E+02
	Earthworm	zero	1.4E-07	n/a	6.8E-07	5.5E-10	zero	zero	8.1E-08	8.4E-08	zero	2.4E-07	5.5E-10	6.8E-07	n/c	1.2E+02
^{90}Sr	Rat	4.3E-04	3.6E-05	n/a	1.9E-05	6.5E-05	1.2E-10	zero	3.2E-05	3.8E-05	5.2E-06	4.3E-05	1.2E-10	4.3E-04	n/c	1.8E+02
	Earthworm	4.3E-04	1.8E-04	1.3E-04	9.5E-05	3.1E-04	1.5E-10	1.0E-11	3.4E-04	1.5E-04	5.9E-05	2.6E-04	1.0E-11	4.3E-04	4.3E+07	8.0E+01
^{137}Cs	Rat	2.4E-04	3.0E-04	1.5E-04	1.5E-04	4.7E-04	2.8E-04	7.9E-05	2.7E-04	2.7E-04	1.9E-04	3.1E-04	7.9E-05	4.7E-04	5.9E+00	4.3E+01
	Earthworm	3.0E-04	3.3E-04	3.0E-04	1.3E-04	5.2E-04	3.0E-04	1.5E-04	2.6E-04	3.0E-04	1.6E-04	3.4E-04	1.3E-04	5.2E-04	4.1E+00	3.9E+01
^{60}Co	Rat	1.5E-03	1.3E-03	1.3E-03	4.9E-04	2.0E-03	1.2E-03	n/a	1.2E-03	1.2E-03	7.2E-04	1.3E-03	4.9E-04	2.0E-03	4.0E+00	3.3E+01
	Earthworm	1.5E-03	1.4E-03	1.4E-03	5.6E-04	2.1E-03	1.3E-03	n/a	1.1E-03	1.4E-03	7.2E-04	1.4E-03	5.6E-04	2.1E-03	3.8E+00	3.4E+01
^{241}Am	Rat	1.7E-06	1.3E-05	1.1E-05	1.1E-05	1.9E-05	5.5E-06	9.6E-07	5.6E-06	3.5E-06	4.0E-06	1.7E-05	9.6E-07	1.9E-05	2.0E+01	7.3E+01
	Earthworm	2.6E-06	1.6E-05	1.2E-05	1.8E-05	2.4E-05	6.1E-06	2.6E-06	5.5E-06	3.8E-06	3.3E-06	1.8E-05	2.6E-06	2.4E-05	9.1E+00	7.6E+01
^{238}U	Rat	1.1E-06	3.9E-05	5.6E-06	2.4E-05	2.1E-07	1.0E-07	1.2E-09	1.4E-06	3.9E-05	5.5E-04	5.2E-07	1.2E-09	5.5E-04	4.6E+05	2.7E+02
	Earthworm	1.1E-06	1.5E-04	5.8E-06	8.3E-05	6.4E-07	1.2E-07	1.5E-08	2.1E-06	1.2E-04	5.3E-04	5.6E-07	1.5E-08	5.3E-04	3.6E+04	2.0E+02

Table 3.7. Calculated DCCS (μGy h^{-1} per Bq kg^{-1}) for external irradiation on soil (data from [98]).

Nuclide	Organism	AECL	EA	ECOMOD	EDEN	EPIC	ERICA	FASSET	LIETDOS-BIO	RESRAD-BIOTA	DosDiMEco	SÚJB	Min	Max	Range	CV (%)
^{3}H	Duck	zero	4.6E-11	n/a	n/a	n/a	zero	n/a	zero	9.4E-14	n/a	n/a	9.4E-14	4.6E-11	n/c	1.4E+02
	Frog	zero	7.8E-11	n/a	4.9E-06	n/a	zero	n/a	zero	9.6E-13	n/a	n/a	9.6E-13	4.9E-06	n/c	1.7E+02
	Rat	zero	4.8E-11	n/a	4.9E-06	n/a	zero	n/a	zero	2.7E-13	n/a	n/a	2.7E-13	4.9E-06	n/c	1.7E+02
	Earthworm	zero	2.8E-10	n/a	4.9E-06	n/a	n/a	n/a	zero	6.9E-12	n/a	n/a	6.9E-12	4.9E-06	n/c	1.7E+02
^{14}C	Duck	zero	8.9E-09	n/a	n/a	n/a	zero	n/a	zero	3.1E-09	n/a	2.0E-08	3.1E-09	2.0E-08	n/c	8.0E+01
	Frog	zero	2.1E-08	n/a	5.9E-07	n/a	zero	n/a	zero	1.0E-08	n/a	6.7E-08	1.0E-08	5.9E-07	n/c	1.6E+02
	Rat	zero	1.1E-08	n/a	5.9E-07	n/a	zero	n/a	zero	3.5E-09	n/a	3.1E-08	3.5E-09	5.9E-07	n/c	1.8E+02
	Earthworm	zero	7.0E-08	n/a	5.9E-07	n/a	n/a	n/a	zero	4.2E-08	n/a	1.2E-07	4.2E-08	5.9E-07	n/c	1.3E+02
^{90}Sr	Duck	2.6E-04	1.1E-05	n/a	1.6E-08	n/a	1.5E-11	n/a	8.8E-06	8.5E-06	1.8E-06	2.7E-05	1.5E-11	2.6E-04	1.7E+07	2.2E+02
	Frog	2.6E-04	3.6E-05	3.2E-05	2.4E-06	n/a	1.6E-11	n/a	n/a	2.6E-05	6.4E-06	9.0E-05	1.6E-11	2.6E-04	1.6E+07	1.5E+02
	Rat	2.6E-04	1.8E-05	n/a	3.2E-07	n/a	1.6E-11	9.0E-11	n/a	1.3E-05	3.0E-06	2.2E-05	1.6E-11	2.6E-04	1.6E+07	2.3E+02
	Earthworm	2.6E-04	9.0E-05	n/a	2.0E-05	n/a	n/a	1.1E-10	9.6E-07	6.8E-05	1.8E-05	1.3E-04	1.1E-10	2.6E-04	2.3E+06	1.2E+02
^{137}Cs	Duck	1.1E-04	1.4E-04	1.4E-04	3.3E-05	7.1E-05	1.1E-04	n/a	7.7E-05	8.4E-05	7.8E-05	1.4E-04	3.3E-05	1.4E-04	4.3E+00	3.7E+01
	Frog	1.2E-04	1.6E-04	1.6E-04	4.7E-05	7.6E-05	1.1E-04	n/a	1.1E-04	8.9E-05	7.9E-05	1.7E-04	4.7E-05	1.7E-04	3.5E+00	3.6E+01
	Rat	1.1E-04	1.5E-04	7.3E-05	3.7E-05	7.3E-05	1.1E-04	1.0E-04	8.6E-05	8.5E-05	8.3E-05	1.5E-04	3.7E-05	1.5E-04	4.2E+00	3.6E+01
	Earthworm	1.2E-04	1.7E-04	n/a	4.2E-05	7.7E-05	n/a	1.2E-04	1.5E-04	9.3E-05	7.1E-05	1.7E-04	4.2E-05	1.7E-04	4.1E+00	4.0E+01
^{60}Co	Duck	n/a	6.4E-04	6.4E-04	1.7E-04	2.8E-04	4.6E-04	n/a	3.6E-04	4.0E-04	2.8E-04	6.0E-04	1.7E-04	6.4E-04	3.9E+00	4.1E+01
	Frog	n/a	7.0E-04	6.9E-04	1.7E-04	3.0E-04	4.9E-04	n/a	4.4E-04	4.1E-04	3.1E-04	6.9E-04	1.7E-04	7.0E-04	4.1E+00	4.2E+01
	Rat	n/a	6.7E-04	6.7E-04	1.9E-04	2.9E-04	4.8E-04	n/a	4.0E-04	4.0E-04	3.0E-04	6.5E-04	1.9E-04	6.7E-04	3.6E+00	4.0E+01
	Earthworm	n/a	7.1E-04	n/a	1.8E-04	3.0E-04	n/a	n/a	4.2E-04	3.8E-04	3.0E-04	7.2E-04	1.8E-04	7.2E-04	4.0E+00	4.9E+01
^{241}Am	Duck	2.3E-06	5.8E-06	5.0E-06	1.3E-06	1.5E-06	2.4E-06	n/a	2.0E-06	1.4E-06	1.4E-06	7.9E-06	1.3E-06	7.9E-06	6.1E+00	7.4E+01
	Frog	2.9E-06	7.3E-06	6.0E-06	8.8E-06	1.7E-06	2.6E-06	n/a	1.9E-06	1.5E-06	2.1E-06	9.0E-06	1.5E-06	9.0E-06	6.0E+00	7.0E+01
	Rat	2.9E-06	6.5E-06	5.5E-05	8.8E-06	1.6E-06	2.5E-06	2.5E-06	2.1E-06	1.4E-06	2.0E-06	8.5E-06	1.4E-06	5.5E-05	3.9E+01	1.8E+02
	Earthworm	2.9E-06	8.2E-06	n/a	8.8E-06	1.8E-06	n/a	2.9E-06	2.2E-06	1.6E-06	1.6E-06	9.2E-06	1.6E-06	9.2E-06	5.8E+00	7.7E+01
^{238}U	Duck	3.2E-08	1.4E-05	2.6E-06	1.5E-06	1.1E-08	4.3E-08	n/a	6.1E-07	9.3E-06	2.2E-04	2.4E-07	1.1E-08	2.2E-04	2.0E+04	2.8E+02
	Frog	8.5E-08	3.3E-05	2.8E-06	7.7E-06	2.5E-08	4.8E-08	n/a	6.6E-07	2.1E-05	2.3E-04	2.7E-07	2.5E-08	2.3E-04	9.5E+03	2.4E+02
	Rat	8.3E-08	2.0E-05	2.6E-06	6.3E-06	1.6E-08	4.7E-08	7.3E-08	6.4E-07	1.3E-05	2.4E-04	2.6E-07	1.6E-08	2.4E-04	1.5E+04	2.8E+02
	Earthworm	8.6E-08	7.5E-05	n/a	2.1E-05	3.6E-08	n/a	8.6E-08	7.4E-07	5.5E-05	2.2E-04	2.8E-07	3.6E-08	2.2E-04	6.3E+03	1.8E+02

Table 3.8. Calculated DCCS (μGy h^{-1} per Bq kg^{-1}) for external irradiation on sediment (data from [98]).

Nuclide	Organism	AECL	EA	ECOMOD	EDEN	EPIC	ERICA	FASSET	LIETDOS-BIO	RESRAD-BIOTA	DosDiMEco	SÚJB	Min	Max	Range	CV (%)
^{3}H	Frog	n/a	1.6E-10	n/a	4.9E-06	n/a	1.3E-12	9.0E-11	n/a	n/a	n/a	n/a	1.3E-12	4.9E-06	3.9E+06	2.0E+02
	Salmonid egg	zero	3.4E-09	n/a	4.9E-06	n/a	n/a	n/a	1.4E-08	8.8E-11	n/a	0.0E+00	0.0E+00	4.9E-06	n/c	2.2E+02
^{14}C	Frog	n/a	4.2E-08	n/a	5.9E-07	n/a	3.0E-08	2.4E-08	n/a	n/a	n/a	n/a	2.4E-08	5.9E-07	2.5E+01	1.6E+02
	Salmonid egg	1.4E-06	9.1E-07	n/a	6.0E-07	n/a	n/a	n/a	1.4E-06	2.5E-07	n/a	1.0E-06	2.5E-07	1.4E-06	5.6E+00	4.8E+01
^{90}Sr	Frog	n/a	7.3E-05	n/a	7.8E-07	n/a	3.2E-05	4.0E-05	n/a	n/a	n/a	9.0E-05	7.8E-07	9.0E-05	1.1E+02	7.5E+01
	Salmonid egg	5.1E-04	4.5E-04	n/a	6.6E-05	n/a	n/a	n/a	1.1E-06	1.8E-04	2.1E-05	1.0E-03	1.1E-06	1.0E-03	9.6E+02	1.1E+02
^{137}Cs	Frog	n/a	3.2E-04	n/a	5.3E-05	n/a	1.6E-04	1.6E-04	n/a	n/a	n/a	n/a	5.3E-05	3.2E-04	6.0E+00	6.3E+01
	Salmonid egg	3.9E-04	3.7E-04	n/a	5.4E-05	7.8E-05	n/a	n/a	1.5E-04	9.6E-05	5.8E-05	2.0E-04	5.4E-05	3.9E-04	7.2E+00	7.8E+01
^{60}Co	Frog	n/a	1.4E-03	n/a	2.0E-04	n/a	7.0E-04	7.0E-04	n/a	n/a	n/a	n/a	2.0E-04	1.4E-03	7.1E+00	6.6E+01
	Salmonid egg	1.5E-03	1.4E-03	n/a	2.2E-04	3.0E-04	n/a	n/a	7.4E-04	5.0E-04	2.7E-04	8.5E-04	2.2E-04	1.5E-03	6.8E+00	7.0E+01
^{241}Am	Frog	n/a	1.5E-05	n/a	9.1E-06	n/a	7.0E-06	7.5E-06	n/a	n/a	n/a	n/a	7.0E-06	1.5E-05	2.1E+00	3.7E+01
	Salmonid egg	1.9E-05	1.9E-05	n/a	1.0E-05	1.8E-06	n/a	n/a	5.6E-06	3.4E-06	8.5E-07	9.5E-06	8.5E-07	1.9E-05	2.2E+01	8.2E+01
^{238}U	Frog	n/a	6.7E-05	n/a	7.0E-06	n/a	1.4E-07	3.7E-05	n/a	n/a	n/a	n/a	1.4E-07	6.7E-05	4.9E+02	1.1E+02
	Salmonid egg	4.2E-04	3.7E-04	n/a	5.8E-05	5.4E-08	n/a	n/a	2.6E-06	1.6E-04	1.9E-04	2.9E-07	5.4E-08	4.2E-04	7.8E+03	1.1E+02

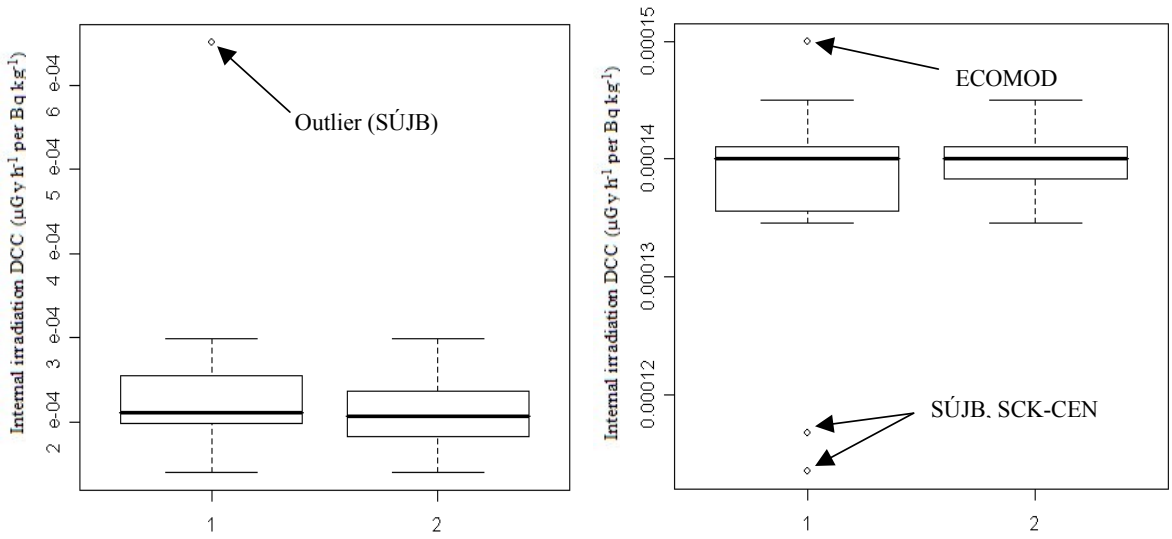

Fig. 3.1. Examples of internal irradiation DCC box plots for ^{90}Sr in salmonid egg (left) and ^{137}Cs in earthworm (right). Box plots prior and after outlier removal are labelled 1 and 2, respectively (applies also to next figure).

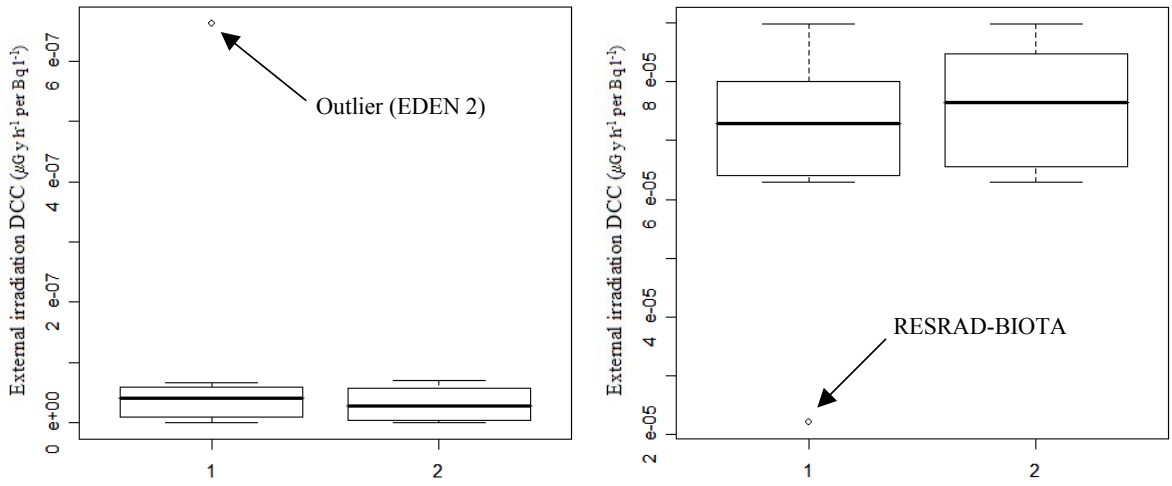

Fig. 3.2. Examples of external irradiation in water DCC box plots for frog: ^{14}C (left) and ^{90}Sr (right).

3.5. Discussion

The results as presented suggest a number of factors that might have caused variation in the DCC data for the different approaches:

— For both internal and external exposure, variability as a consequence of different number of decays or daughter products being included (most notably for ^{238}U) within the estimation of DCC (Table 3.3).

— For external exposure, differing media geometries being assumed, e.g. the effect of medium thickness and immersion depth of the target receptor for γ-emitters, or shielding effects such as varying soil density.

3.5.1. Effect of number of daughter products

The comparative effect of including different numbers of uranium daughters, from zero up to 4 (^{234}Th, ^{234m}Pa, ^{234}Pa and ^{234}U) on internal and external dose was assessed by performing repeated runs of the Environment Agency R&D128 biota dose calculation program for ^{238}U. Results are given in Figure 3.3, where it can be seen that ^{234}Th increases the ^{238}U low β-internal dose by a factor of 2.4, and further addition of daughters has no effect except for ^{234}U, which would increase the dose by a further factor of 1.5. ^{234}Th increases the ^{238}U β + γ internal dose by a factor of 7, ^{234m}Pa by a further factor of 4 to 12 (depending on geometry), and the remaining daughters result in no change. However, in terms of total dose, which is dominated by the α-dose, the only important effect is the doubling of α-dose when including ^{234}U in addition to the other daughters. This is clearly the reason for the high ^{238}U internal DCCs estimated for aquatic organisms by the FASSET methodology; previously published values from the Environment Agency R&D128 methodology [5, 6] are comparatively high (values presented for this exercise were re-estimated with ^{234}U excluded).

With respect to external exposure, ^{234}Th increases the ^{238}U low β-dose by a factor of approximately 3, and further addition of daughters has no effect except for ^{234}U which increases the dose by a further factor of 1.4. ^{234}Th increases the ^{238}U β + γ external dose by a factor of 9 to 20 (depending on geometry), ^{234m}Pa by a further 6 to 50-fold, and the remaining daughters result in little change. In terms of total dose (dominated by β + γ), the addition of the first two daughters has the biggest effect, with factors in the order of 9 to 20 (^{234}Th) and 6 to 50 (^{234m}Pa).

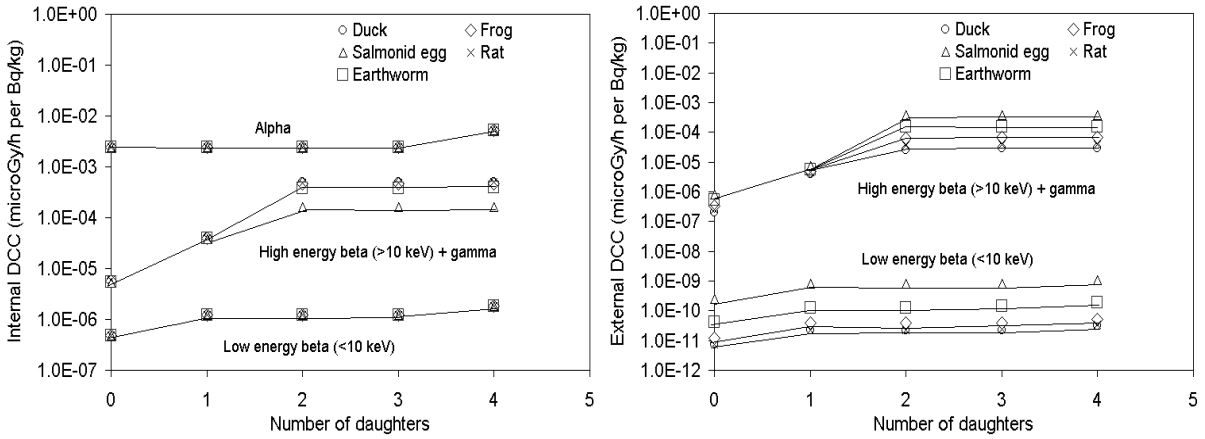

Fig. 3.3. Comparative effect of including different numbers of uranium daughters on the internal (left) and external (right) exposure DCCs, calculated using the England and Wales Environment Agency R&D128 methodology (data from [98]). External DCCs for α irradiation are zero.

3.5.2. Effect of soil/sediment depth and target position

Some methodologies employed in this paper differ in the consideration of infinite versus finite source depth (approximately 50 cm) for external doses. The effect of different depth assumptions on the DCC can be calculated, but this requires complex self-absorption calculations. Fortunately, most of this effort can be averted by using the dose-rate conversion factors in air for photon sources in soil derived by Kocher and Sjoreen [107]. These authors have performed calculations considering photon-emitting sources uniformly distributed within soil layers of different thickness. These calculations are for above-ground receptors; however, the result is insensitive to the height of the receptor for less than 10 m, so the results are applicable to organisms living on the soil. Moreover, photon transport in air is negligible when calculating the effect of sources below ground surface.

Analysis of the published data [107] reveals that, for each energy, data fit to the equation:

$$\mathrm{DCC} = D\mathrm{CC}_{\infty} \cdot (1 - e^{-\mu \times depth}) \tag{3.1}$$

where: DCC_{∞} and μ are fitting constants representing the DCC under the assumption of infinite soil thickness and the dependency with depth, respectively (Figure 3.4).

At 0.6 MeV, for example, the DCCs at 10 and 50 cm depths are 79% and virtually 100% of those of infinitely deep soil, respectively. The higher the energy, the more accentuated is the deviation of a DCC for a 10 cm soil slab from a DCC for infinitely deep soil. Hence, at 10 MeV and a 50 cm depth, the difference related to assuming infinite depth is less than 4%, but a difference of 52% is observed if 10 cm is taken as the soil depth. This analysis demonstrates that there is no appreciable difference in results between assuming either: (a) that the radioactivity is distributed within the first 50 cm of soil; or (b) it is distributed to an infinite depth. Under an assumption of a depth of less than or equal to 10 cm (as assumed by many of the approaches, see Table 3.2), there would, however, be some differences, especially for high-energy photons.

This interpretation is confirmed by published effective dose equivalent data [108, 109] for sources distributed to different depths of soil having a density of 1.6 g cm^{-3}. Such data reveal that, for depths of greater than or equal to 15 cm, the dose is more than 90% of that calculated under the assumption of infinite depth for energies below 1 MeV. This implies that for a selection of typical radionuclides (^{14}C, ^{60}Co, ^{90}Sr, ^{137}Cs, ^{238}U and ^{241}Am), 15 cm depth doses would be approximately 90% or more than the dose at an infinite depth. An exception is ^{60}Co where, on account of its greater than 1 MeV transitions, the proportion is somewhat lower at 84%[5].

Kamboj et al. [109] arrived at a mathematical fit of dose coefficients at different depths, covering the above selection of radionuclides, as described by:

$$D_T = D_{\infty} \cdot (1 - A \cdot e^{-K_a \rho T} - B \cdot e^{-K_b \rho T}) \tag{3.2}$$

where: A, B, K_a, and K_b are four fitting functions, ρ is the soil density (1.6 g cm^{-3}) and T is the thickness of soil (cm).

[5] In this calculation ^{90}Sr includes ^{90}Y, ^{137}Cs includes ^{137m}Ba, and ^{238}U includes ^{234}Th, ^{234m}Pa and ^{234}Pa in secular equilibrium with the parent radionuclide.

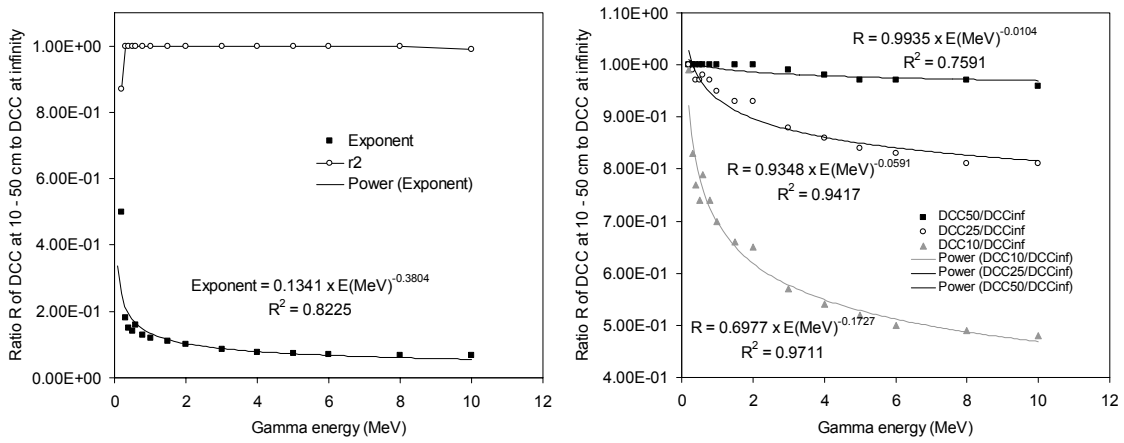

Fig. 3.4. Fitting constant μ (exponent) and coefficient of determination r^2 of equation $\mathrm{DCC} = DCC_{\infty} \cdot (1 - e^{-\mu \times depth})$ *(left), and variation of external DCCs for different source depths for a target above the soil – data from Kocher and Sjoreen [107] in respect of* DCC_{∞} *(right). DCC$_x$ is the DCC for assumed depth x.*

Using this fit, it is calculated that, for a depth of 50 cm, DCCs for ^{14}C, ^{60}Co, ^{90}Sr, ^{137}Cs, ^{238}U and ^{241}Am are virtually indistinguishable (less than 0.2% difference) from infinite depth DCCs, confirming the analysis in Figure 3.4.

Similar calculations were performed to illustrate the effect of a receptor organism that is at a 25 cm depth inside a soil slab of 50 cm. This case can be treated as two 25 cm slabs, one above and one below the target, i.e. as twice the dose for a homogeneous, isotropic source on top of a soil slab of 25 cm thickness. From the data presented in Figure 3.4, it is evident that there is little difference between the two assumptions used, of infinite soil thickness and 50 cm contaminated soil layer (with a less than 5 % difference at energy less than 1.25 MeV).

The above analyses are made on the assumption of uniform distribution of the source term within the depth profiles examined. In reality, sources are not distributed uniformly in aquatic sediments, but generally peak at a specific level that will change with time.

It is concluded that, for external exposure from soil/sediment, observed discrepancies in external dose are unlikely to be completely explained by variations in soil depth, source position in or above soil, or height above it. The factor that is more likely to have an influence on external dose variability for different calculation methodologies is difference in the number of decay modes and energies considered for the radionuclide (see Table 3.3), as well as shielding factors such as soil density. For example, the published data considered in this analysis [107] are for a soil density of 1.4 g cm^{-3}. However, this publication states that, in practice, the shielding provided by a given thickness of material is proportional to density. Hence, treating the soil at unit density (the lowest assumed within any of the approaches) should give an approximately 40% higher external dose estimate.

3.5.2.1. Analysis of robust means and Z-scoring

Performance statistics for internal and different external exposure DCCs (p-values for normality, skewness, kurtosis and Grubbs tests) from Vives i Batlle et al. [98] are summarised in Figures 3.5 (top) to 3.9 (top), respectively. A marked improvement in p-values in the robust data compared with the raw data is evident from these Figures. The effect of eliminating outlier data from averaging, based on the outcomes of these tests, is then illustrated in Figures 3.5 (bottom) to 3.9 (bottom), respectively, where the arithmetic means and associated standard deviations relating to the robust (i.e. outlier-removed) set, along with the raw equivalents, are given. The robust statistics were used in calculating Z-scoring values, as summarised in Figure 3.10.

As explained in the Section 3.3.4, the data from Figure 3.10 should not be used to pass value judgements on the validity of any approach, considering that most discrepancies are attributable to varying degrees of conservatism and/or radionuclide/source-target geometry assumptions. In addition, there are limitations inherent in ranking approaches when there are some in which the DCCs are calculated using a stand-alone code outside the approach itself (e.g. RESRAD-BIOTA used MCNP to calculate DCCs and LIETDOS-BIO used a Monte Carlo approach consistent with MCNPX. Similar limitations exist when the DCCs are simply taken from other approaches or published data (e.g. AECL, which adopts FASSET and RESRAD-BIOTA DCCs in several instances).

On a radionuclide-by-radionuclide basis, the highest Z-scores tend to relate to ^{3}H, ^{14}C and the α-emitters (^{238}U and ^{241}Am); the radionuclides whose emissions tend to have shorter ranges in matter. A shorter range implies a higher variability of the DCC in response to variations in density, target layering (i.e. the presence of skin or fur), or other assumptions by the dose calculation method influencing the degree of self-absorption within the target organism.

As described in Section 2.11, two types of calculation are implemented in EDEN 2: Monte Carlo and local deposition. The Monte-Carlo approach was used to provide DCC values for this exercise (Tables 3.4–3.8). As an additional test (conducted using the duck, frog and rat geometries), it was decided to investigate whether the alternative local deposition approach provided results closer to the robust mean than the Monte Carlo simulation.

The use of the local deposition approach tends to bring some EDEN 2 results somewhat closer to the robust mean of all the approaches. For internal exposure, the local deposition approach essentially modifies the β DCCs. For example, the ^{3}H DCC becomes 3.1×10^{-6} μGy h^{-1} per Bq kg^{-1} for all organisms, instead of 5.9×10^{-6} μGy h^{-1} per Bq kg^{-1}. For ^{90}Sr, the internal DCC calculated with the local deposition approach is evaluated at 6.2×10^{-4} μGy h^{-1} per Bq kg^{-1} versus 5.6 to 6.0×10^{-4} μGy h^{-1} per Bq kg^{-1} calculated using the Monte-Carlo code. For external exposure, the local deposition hypothesis leads to a zero DCC for pure α- and β-emitters and a decrease in the DCC for radionuclides comprising various types of radioactive decay. As an illustration, the ^{241}Am DCC for a duck in water falls from 1.2×10^{-5} μGy h^{-1} per Bq kg^{-1} (Monte-Carlo) to 4.3×10^{-6} μGy h^{-1} per Bq kg^{-1} (local deposition).

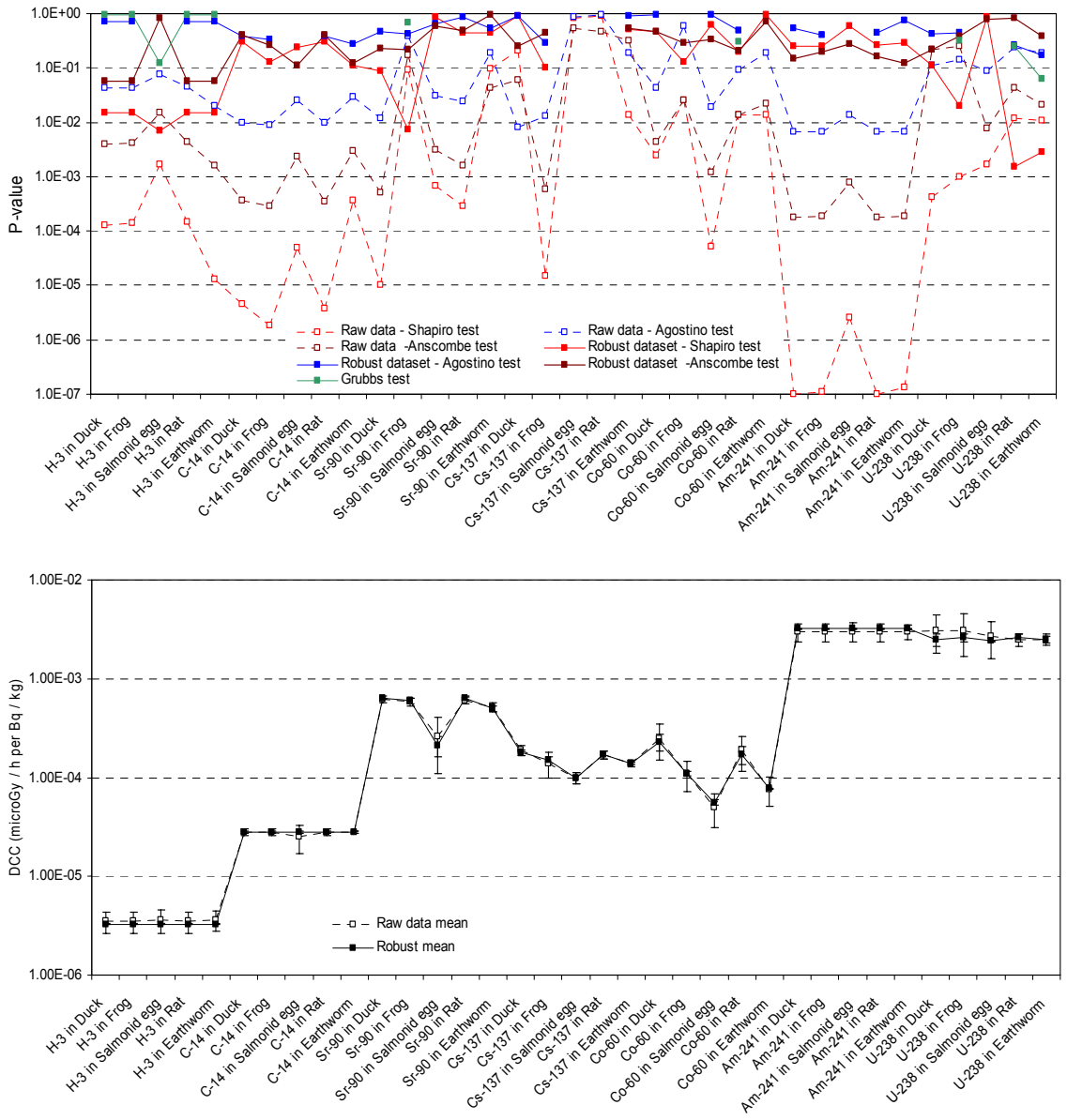

Fig. 3.5. Performance statistics for internal exposure DCCs (top) and resulting difference in robust mean DCCs compared with the mean of the raw data. In this and subsequent figures, non-continuous lines indicate non availability of p-values due to either (a) insufficient number of data to perform statistical test, (b) skewness and kurtosis tests not reported (data fail normality tests, but not Grubb's test for outliers), or (c) test not required to assess robustness.

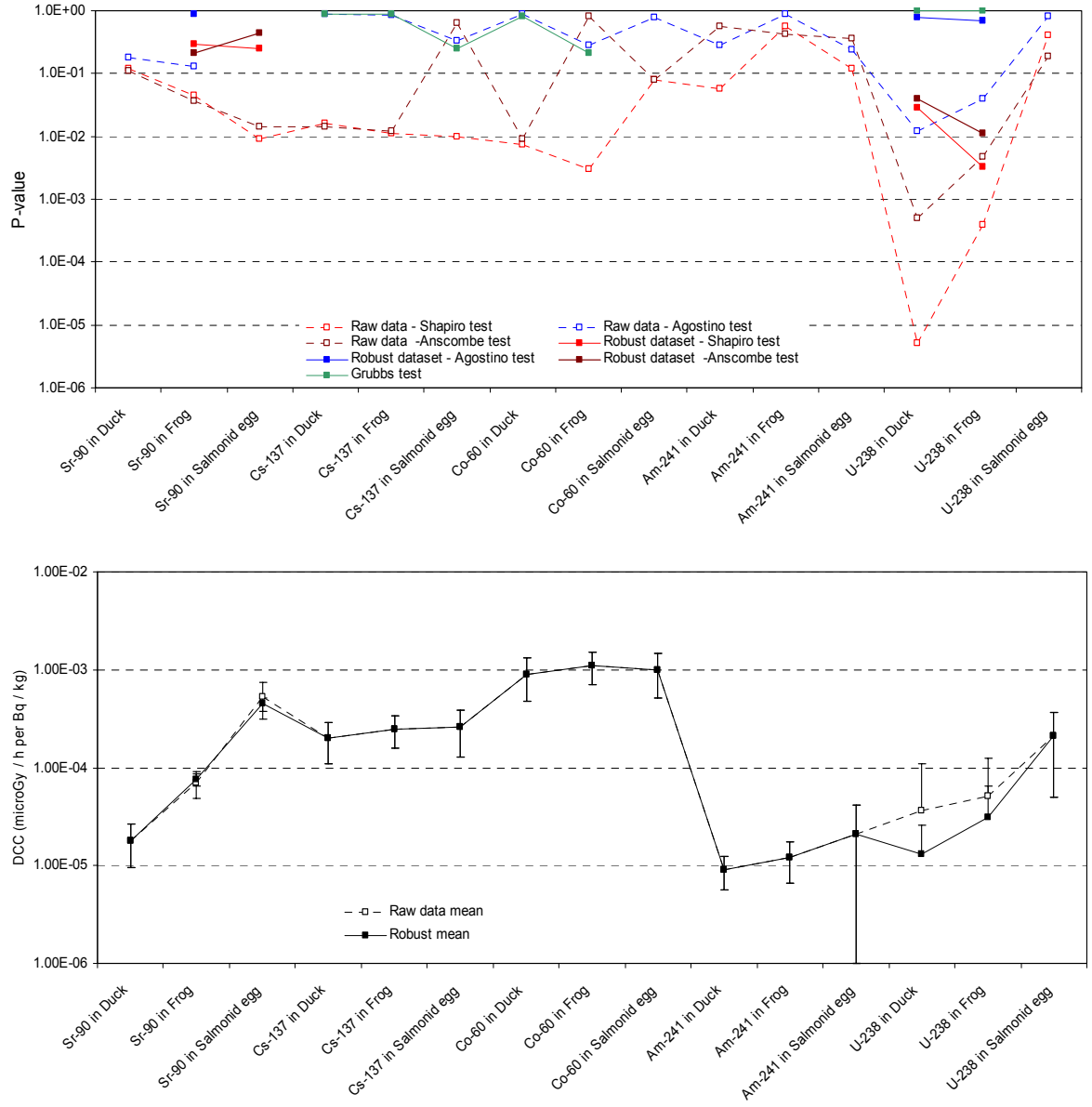

Fig. 3.6. Performance statistics for external exposure DCCs in water (top) and resulting difference in robust mean DCCs compared with the mean of the raw data.

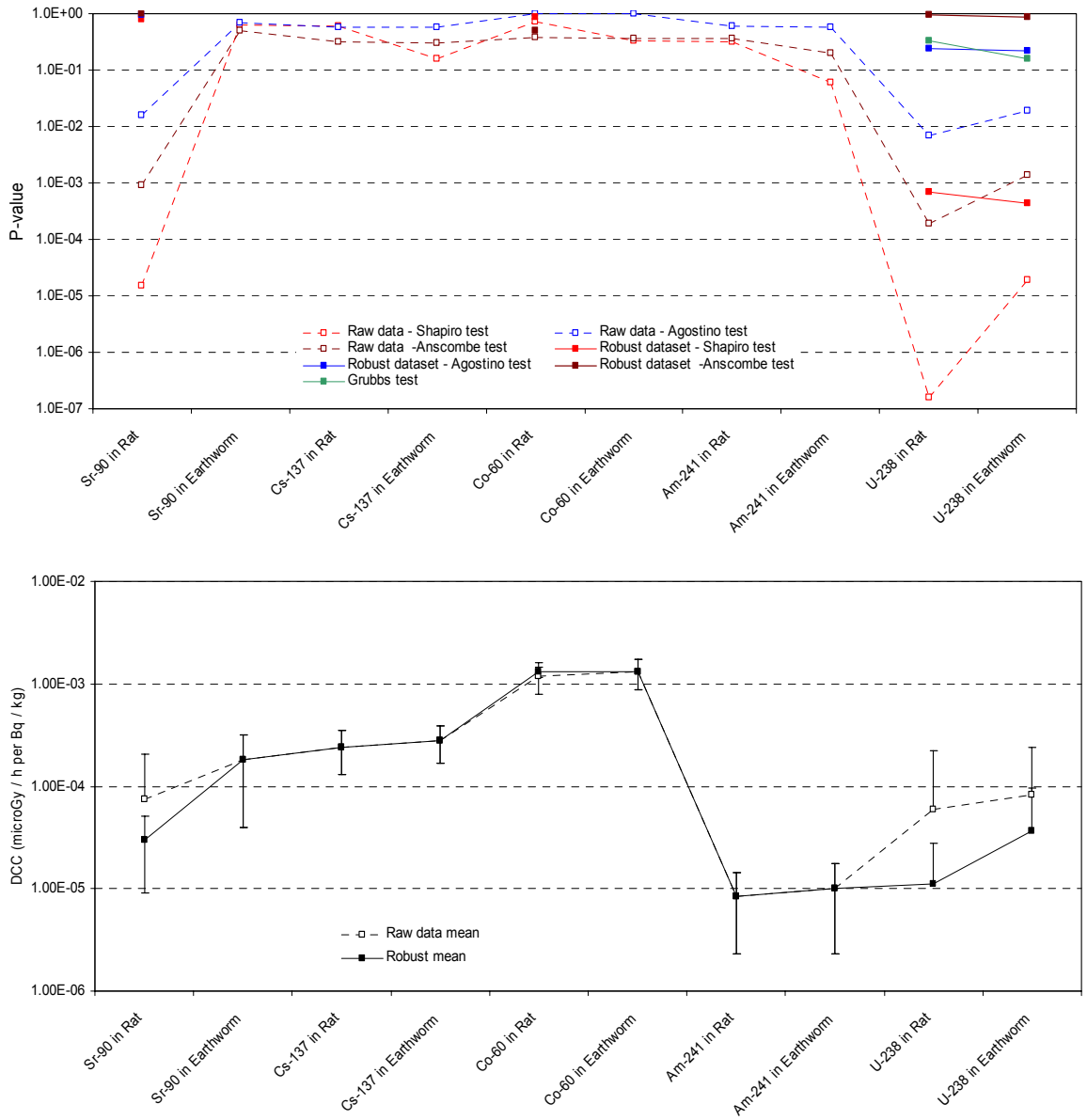

Fig. 3.7. Performance statistics for external exposure DCCs in soil (top) and resulting difference in robust mean DCCs compared with the mean of the raw data.

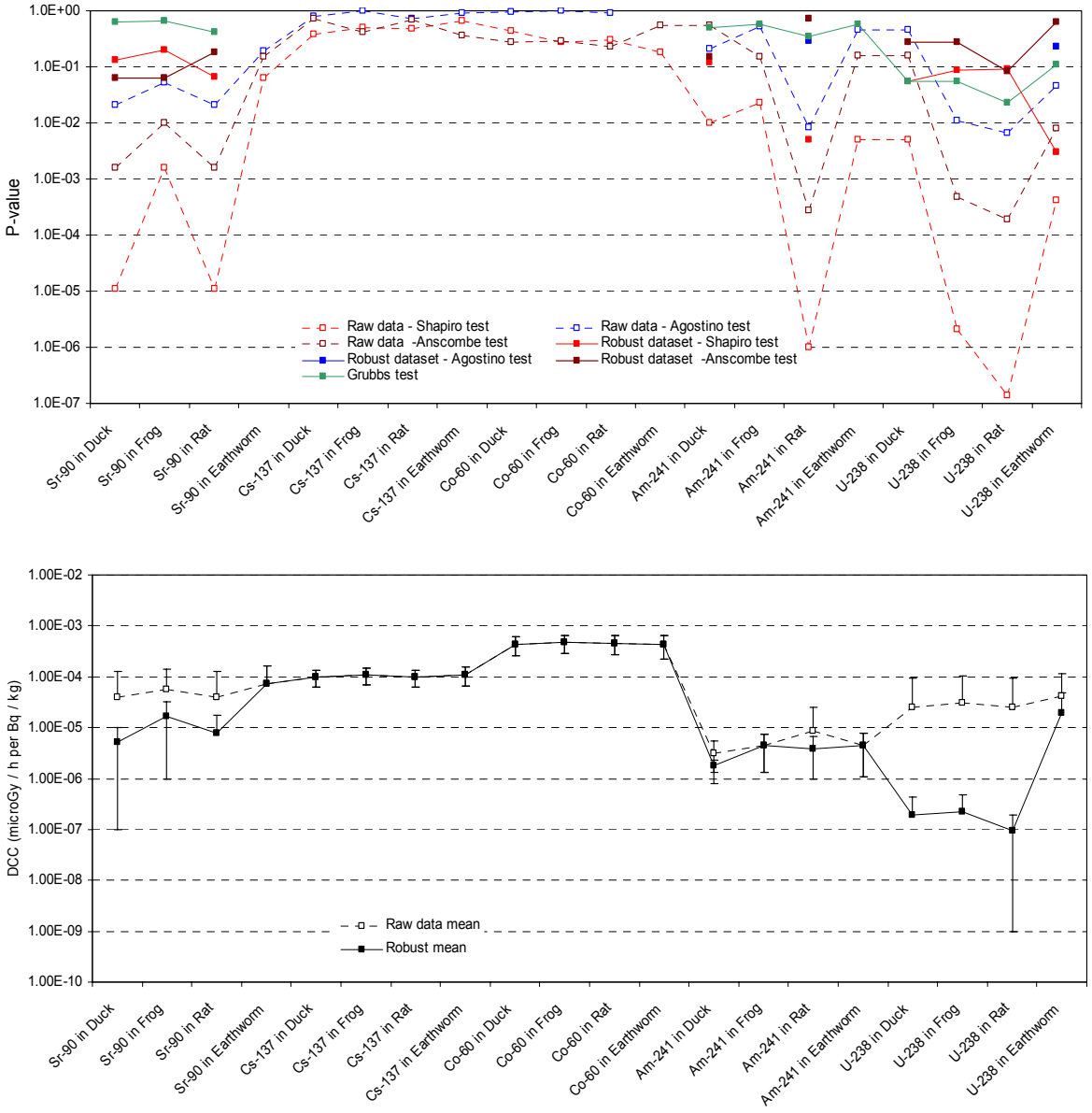

Fig. 3.8. Performance statistics for external exposure DCCs on soil / on shore (top) and resulting difference in robust mean DCCs compared with the mean of the raw data.

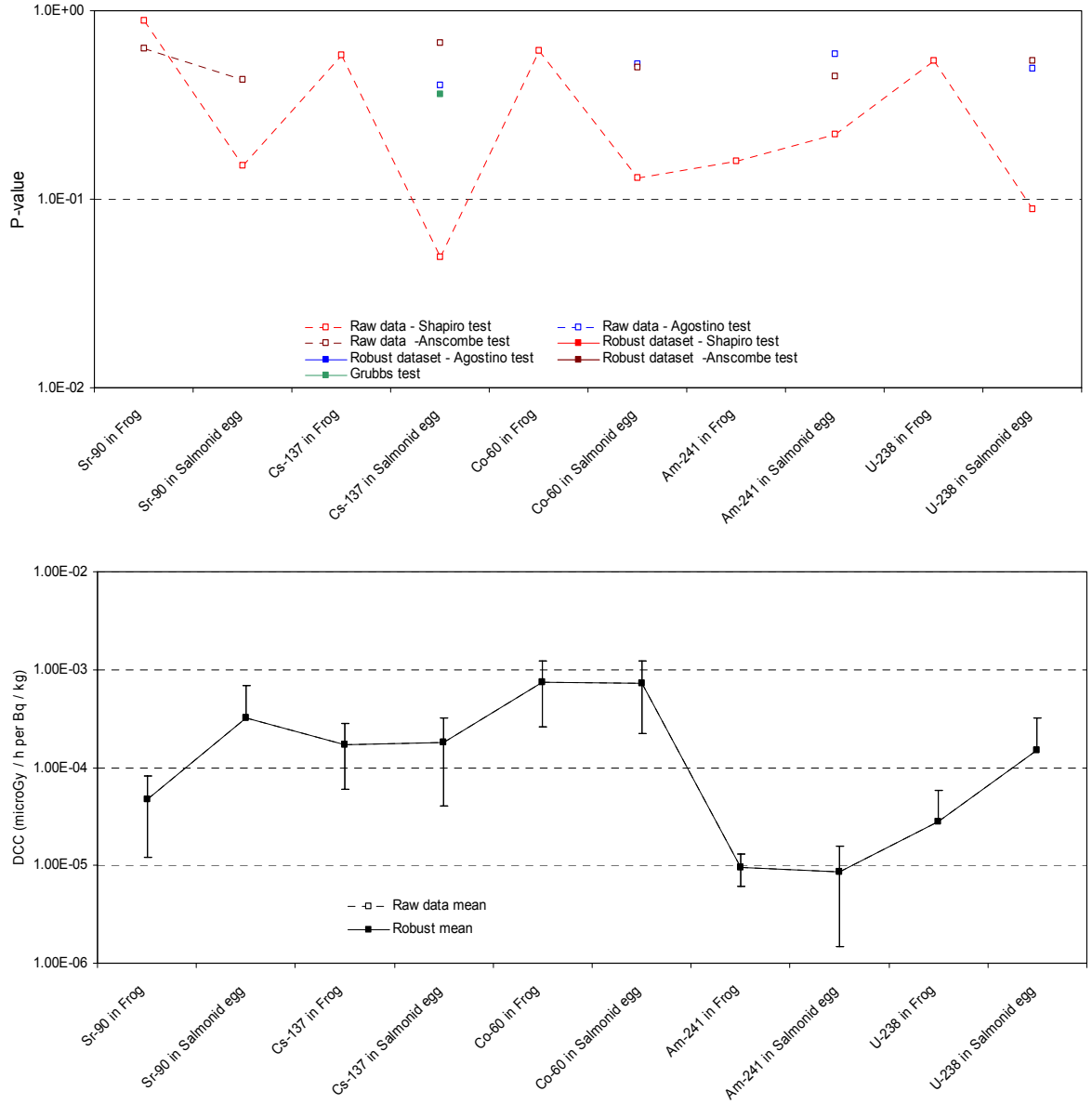

Fig. 3.9. Performance statistics for external exposure DCCs in sediment (top) and resulting difference in robust mean DCCs compared with the mean of the raw data (raw and robust means are identical in this case).

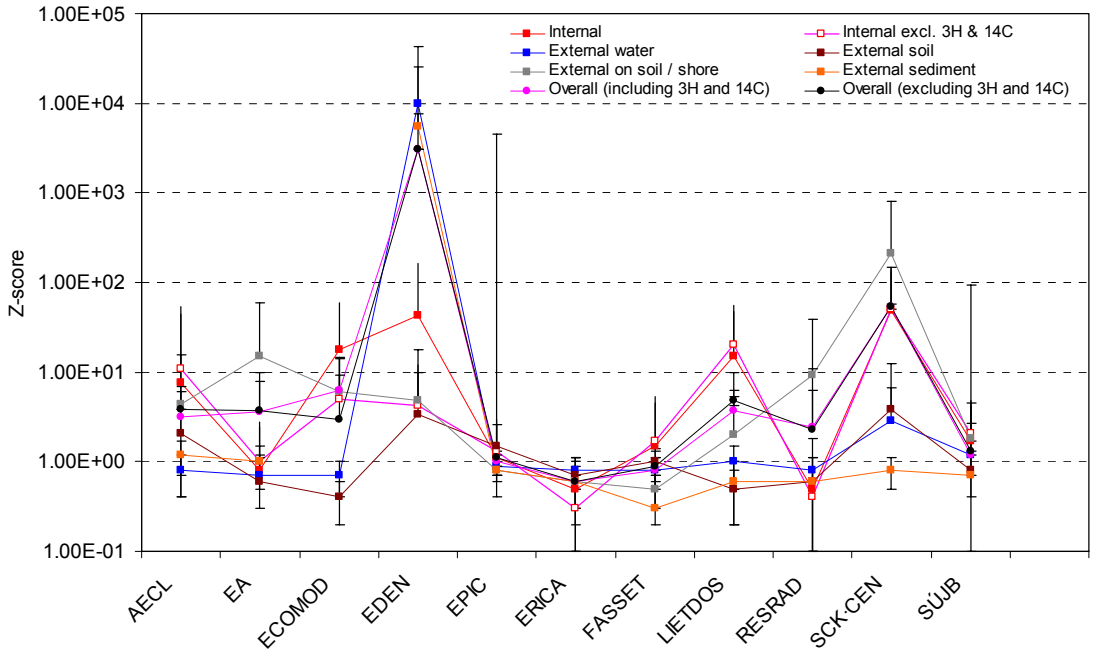

Fig. 3.10. Summary of all Z-scoring tables for the EMRAS DCC comparison exercise.

Efficacy measures rank as follows: External sediment (90%) > External water (82%) > External soil (73%) > Internal (55%) > External on soil/shore (45%), with only the latter scoring less than 50%. The relatively low efficacy measure for internal exposure reflects the fact that, for certain radionuclides, a few approaches (e.g. ECOMOD, EDEN 2 and DosDiMEco for ^{3}H, and DosDiMEco for ^{241}Am) give DCCs significantly off-range whilst the rest report almost identical values. This results in an isolated group of elevated Z-scores, reducing the overall efficacy measure. However, overall, the inter-compared DCCs for internal irradiation have relatively low dispersion, as illustrated by the low coefficients of variation in Table 3.4.

The lower efficacy measure for external on soil/shore DCCs, is likely to be due to additional assumptions concerning the position of the target above-ground and differences in source-target geometry/shielding factors (see Section 3.5.2).

3.6. Conclusions

An exercise directed at the comparison of screening-level approaches for the calculation of unweighted absorbed dose rates (reported as DCCs) in biota has been successfully performed. Unweighted internal and external DCCs for a selection of the proposed ICRP Reference Animal geometries were calculated. The data submitted by the participants were subject to exploratory statistical analysis to identify and remove outliers. Statistics were then calculated for the robust data (which were found to follow normal distributions) as the basis for scoring each approach for performance.

The purpose of this study was to compare screening and simple site-specific approaches designed for biota dose assessment for regulatory purposes. These approaches are not intended to generate a scientifically realistic representation of reality. Rather, they purport to

represent a highly variable quantity (the biota DCC) that cannot be measured, but rather must be modelled. Therefore, at the outset of this work, it was expected that the different approaches would give rise to differences, based on the different physical and ecological assumptions made. Hence, no value judgement was passed on the validity of any approach.

On initial inspection of the data, inter-comparison results indicated that, whilst DCCs for internal exposure compare well between the different approaches, variation is greater for external exposure DCCs. Whilst external doses from β-emitters are low, there is considerable variation for such doses between the different approaches. It is generally accepted that external exposure of living organisms by short-range α- or β-radiation (e.g. from ^{3}H, plutonium and some naturally-occurring radionuclides) is of little radiological significance, due to their low range in matter. This prevents such radiation from reaching the radiosensitive targets, including vitally important organs, such as germ cells and hemopoetic cells. For example, the range of ^{3}H β-radiation in soft biological tissue is less than 10 μm, and for α-particles of 5 MeV, the range is on the order of 50 μm, which is too short to cross surface tissue and reach radiosensitive cells. Therefore, whole-body averaging of the external low energy β doses received by non-radiosensitive integument tissue (i.e. the external covering of the body, such as skin, feathers, scales, etc.) makes little sense from a radiobiological point of view.

It is not practically feasible to investigate method-by-method to try to attribute all degrees of variability to a specific set of assumptions. However, it is possible to conclude here that where variation among internal DCCs is greatest, it is generally as a consequence of different daughter products being included (e.g. ^{238}U) in the DCC of the parent. In the case of external exposures, particularly to low-energy β-emitters, variations are most likely to be due to different media densities being assumed.

On a radionuclide-by-radionuclide basis, the approach Z-scores higher than 2 tend to relate to ^{3}H, ^{14}C and the α-emitters. This is consistent with radiation with the lowest range across matter being most adversely affected by source-target geometry effects.

The efficacy measure of this inter-comparison is about 70% on average, and on that basis it can be concluded that the inter-comparison was successful in demonstrating that all approaches to biota dose calculation considered in this exercise give reasonably comparable results. This is the case even though different assumptions (including the use of default geometry DCCs, rather than estimation of bespoke values for this exercise) are made by the various approaches (Tables 3.2 and 3.3).

Now that the differences between the approaches are known and some of them have been explained, the information can be utilised by users wishing to assess and interpret the consistency of their biota dosimetry methodology with the approaches participating in this inter-comparison. This study will also allow differences associated with dosimetry calculations to be put into context with those associated with transfer and other aspects of an assessment of non-human biota.

It should be noted some of the approaches applied in this chapter are ‘works in progress’, and as such their DCC values may change in the future (e.g. this exercise was conducted prior to the release of the ERICA Tool). Whilst RESRAD-BIOTA and EA R&D128 are freely available, the ICRP RAP geometries as used within this exercise are not available for users.

CHAPTER 4. COMPARISON OF PREDICTED WHOLE-BODY ACTIVITY CONCENTRATIONS

In this chapter we present and discuss the results of the second model-model comparison exercise conducted by BWG participants. The objective of this exercise was to compare the estimated whole-body activity concentrations for a range of radionuclides in 19 terrestrial and freshwater organisms assuming a nominal activity concentration of 1 Bq per unit media. Results of the inter-comparison discussed in this chapter are also presented in Beresford et al. [110].

4.1. Exercise description

Participants were asked to run their models to provide estimated fresh weight (fw) whole-body activity concentrations of: ^{241}Am, ^{14}C, ^{60}Co, ^{134}Cs, ^{137}Cs, ^{3}H, ^{129}I, ^{131}I, ^{210}Po, ^{239}Pu, ^{226}Ra, ^{90}Sr, ^{99}Tc, ^{232}Th, ^{234}Th, ^{234}U, ^{235}U and ^{238}U. Predictions were required for seven terrestrial organisms and twelve freshwater organisms (see Table 4.1). Organisms were selected on the basis of being common to most of the participating approaches. Marine organisms were not specified as only three of the participating models consider the marine environment. Model inputs were specified as 1 Bq kg^{-1} dry weight (dw) soil for terrestrial organisms and 1 Bq l^{-1} water from freshwater ecosystems. Exceptions were ^{14}C and ^{3}H for terrestrial organism, which were specified as 1 Bq m^{-3} air.

4.2. Application of the participating models

Eight of the approaches described in Chapter 2 participated within this exercise: AECL, DosDiMEco, EA R&D128, ECOMOD, ERICA, FASSET, LIETDOS-BIOTA and RESRAD-BIOTA. Only specifics of application to this exercise are noted in this section; Chapter 2 should be consulted for model descriptions.

4.2.1. AECL approach

Where possible, AECL applies site-specific transfer parameters in risk assessments, taking values from the scientific literature when site-specific data are not available. Therefore, for the purposes of this generic exercise, concentration ratios were taken from the Canadian literature to estimate transfer to freshwater macrophytes, invertebrates, frogs and fish. It was assumed that the CR values for small benthic crustaceans were comparable to zooplankton values. In addition, in the case of ^{226}Ra for which no frog CR was available, it was assumed that the frog CR was similar to that of benthic fish.

Table 4.1. Prganisms specified within the exercise instructions.

Terrestrial organisms	Freshwater organisms	
Grass/Herb	Phytoplankton	Pelagic fish
Shrub	Zooplankton	Benthic fish
Earthworm	Macrophyte	Fish egg
Herbivorous mammal	Benthic mollusc	Amphibian
Carnivorous mammal	Small benthic crustacean	Duck
Rodent	Large benthic crustacean	Mammal
Bird egg		

Radionuclide concentrations in fish eggs were estimated using a fish egg-to-fish muscle concentration ratio, as described in Equation 6 of Appendix III. As discussed above, these ratios were quantified based on a literature review conducted to compile tissue-specific concentration data for a range of non-human biota [14].

Tritium and ^{14}C transfer to aquatic plants and animals were estimated using a specific activity approach, (Appendix III.1). It was assumed that water contains 111 g H l^{-1}, aquatic plant tissues contain 120 g H kg^{-1} fresh weight and aquatic animal tissues contain 130 g H kg^{-1} fresh weight. In addition, it was assumed that there was 10 mg of dissolved inorganic carbon per litre of water, that aquatic macrophytes contain 45% C per unit dry weight with an 87% water content and that phytoplankton contain 45% C per unit dry weight with an 84% water content. By comparison, zooplankton were assumed to contain 45% C per unit dry weight with a 90% water content.

In most cases, soil-to-plant CR values were taken from the North American literature [38, 111]. Soil-to-animal CR values were taken from FASSET [112] as were the water-to-animal CR values for freshwater duck. In doing so, parameter values were selected to maximise the level of confidence wherever possible. For example, FASSET distinguishes low, medium and high confidence in its reported data. Therefore, the data for which the highest level of confidence could be achieved were used. When CR values were not available in FASSET, a food-chain transfer approach (as described in Appendix III.2) was applied to estimate radionuclide concentrations in the carnivorous terrestrial mammal (which was assumed to be a fox with respect to its body size and diet), the herbivorous terrestrial mammal (which was assumed to be a meadow vole) and the rodent (which was assumed to be a deer mouse). It was assumed that diet of the meadow vole consisted of 98% herbs and 2% aerial invertebrates (assumed to have similar radionuclide concentrations as aquatic invertebrates), with dietary transfer coefficients of: (i) Co – 34 d kg^{-1} (fw); (ii) Cs – 57 d kg^{-1} (fw); (iii) I – 46 d kg^{-1} (fw). Activity concentrations of dietary items were taken from estimates that were made as part of the exercise.

The aquatic mammal was assumed to be an herbivorous mammal (namely a muskrat) feeding only on freshwater macrophtyes with radionuclide concentrations as tabulated as part of this exercise. Activity concentrations in the aquatic mammals were then estimated in an Excel spreadsheet using the approach that has been described in Appendix III. For ^{241}Am, the soil-to-mammal CR value for the muskrat was taken from Copplestone et al. [113] for herbivorous mammals.

As for the aquatic biota, ^{14}C concentrations in terrestrial biota were estimated using a specific activity approach. In the case of terrestrial plants, it was assumed that air was contained 0.18 g C m^{-3} [114] and that the specific activity of the plants was equal to that of air. It was assumed that the plants contain 45% C per unit dry weight with a water content of 79% for herbivorous plants and 85% for the leaves of shrub. A 45% C content per unit dry weight was also assumed for worms and vertebrates, with water contents of 84% for worms and of 68% for mammals.

4.2.2. RESRAD-BIOTA

The RESRAD-BIOTA results presented in this chapter for terrestrial reference organisms were Level 3 calculations obtained by calculating tissue concentrations for actual biota species selected to represent each organism category. Whole-body concentrations were calculated by considering inhalation of resuspended radionuclides as well as ingestion of

water, soil and different foodstuffs (Figure 4.1 illustrates the conceptual model used in RESRAD-BIOTA for this exercise). Concentrations in earthworm and plants were calculated first, these then were used to calculate whole-body concentrations of herbivorous mammals (represented by Kangaroo rats (*Dipodomys* spp.)) and herbivorous birds (represented, for this exercise, by Mourning doves (*Zenaida macroura*)). Concentrations in carnivorous mammals (represented by Kit foxes (*Vulpes macrotis*)) could then be calculated from the estimated activity concentrations in herbivorous mammals and birds. Activity concentrations in bird eggs were calculated for this exercise using intake-to-egg transfer factors obtained from the literature. Allometric equations were used to develop inhalation rates. For ingestion rates, both allometric equations and literature data were used; diet composition being estimated from literature data and professional judgment. Radiological decay and biological loss from the organism were taken into account in calculating the whole-body activity concentrations. It was assumed that 100% of the air inhaled and food ingested were contaminated. The estimated whole-body concentrations are the maximum values for the life spans of the actual species. For freshwater reference organisms CR values from a variety of literature sources were used.

4.2.3. ERICA

The development of the ERICA Tool and associated databases was on-going during the course of the BWG. The CR values applied in this exercise were those from a pre-release version of the ERICA Tool default databases. As such some will differ to those described by Beresford et al. [23] and Hosseini et al [24] and applied in the work described in Chapters 5 and 6.

4.2.4. FASSET

FASSET provides CR values for plants on a dry matter basis; for this exercise a dry matter content of 25% was assumed to provide the specified fresh weight activity concentrations.

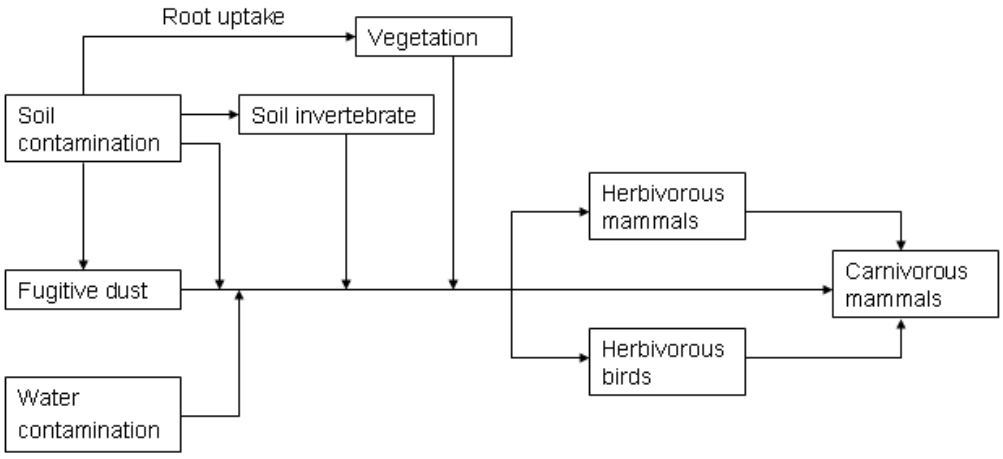

Fig. 4.1. Conceptual model used for RESRAD-BIOTA predictions for terrestrial organisms (the conceptual model is user defined).

4.2.5. DosDiMEco

For this exercise the representative species used were *Capreolus capreolus* for herbivorous mammal, *Canis lupus* for carnivorous mammals, *Apodemus sylvaticus* for rodent, *Anas platyrhynchos* for duck and *Passer domesticus* for bird egg. Ingestion of contaminated food was considered to be the sole contamination route. Herbivorous animals are considered to eat only herbs and the diet of carnivorous animals consisted of herbivorous animals. Contaminated grain was assumed to eaten by rodents and birds (to calculate concentration in bird egg). The duck was considered to eat solely plankton (50% zooplankton and 50% phytoplankton).

4.2.6. ECOMOD (Russia)

For application in this exercise CR, values predominantly from the Russian language literature were used.

4.3. Results

Predicted whole-body activity concentrations by the participating models are compared within Tables 4.2–4.14. For approaches using biota to media concentration ratios, the predicted activity concentrations equate to the CR value used (as 1 Bq per unit media concentrations were assumed). Where predictions were requested for different isotopes of the same radionuclide most participants reported the same activity concentrations for all isotopes. Consequently, results are presented for one isotope only. For clarity, the tables present predicted activity concentrations for large crustacean only, as most approaches report the same value for small and large crustacean for virtually all radionuclides.

Three of the approaches using CR values (ERICA, FASSET and EA R&D128) use a guidance methodology to select CR values when empirically derived values for a given radionuclide-organism combination are missing; Tables 4.2–4.14 identify when guidance methodology has been used. Whilst the guidance methodology used by all three of these approaches stems from documentation associated with the application of EA R&D128 [6] it does not result in the same values being recommended by each approach. This is because each approach has its own empirical CR dataset on which to base the guidance values and, especially in the case of the ERICA approach, the guidance given has evolved considerably.

Three of the approaches (AECL, LIETDOS-BIO and ERICA) use CR values from other of the participating approaches. For this exercise both AECL and LIETDOS-BIO used CR values from the FASSET methodology on a number of occasions. These are noted within Tables 4.2–4.14, as are occasions where ERICA adopts default CR values for freshwater organisms from EA R&D128. A criticism of the FASSET methodology [115] is that on occasions it presents more than one ‘recommended’ CR value for the same radionuclide-organism combination. Consequently, when using FASSET values, AECL and LIETDOS-BIO have not always predicted the same activity concentration as each other or the FASSET application within this exercise (e.g. predicted ^{137}Cs activity concentrations in pelagic fish).

Table 4.2. Comparison of predicted whole body ^{241}Am activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	0.35 (1.4)	2.00×10^{-2} (2.0)	0.13 (0.2)	9.99×10^{-2} (0.1)	n/r	0.13* (0.2)	0.13* (0.2)	n/r
Shrub	8.00×10^{-3} (1..3)	1.00×10^{-4} (1.2)	$2.50\times10^{-4\#}$ (0.6)	$4.96\times10^{-3\#}$ (1.1)	n/r	n/r	n/r	2.84×10^{-4} (0.6)
Grass/Herb	8.00×10^{-3} (1.6)	1.00×10^{-4} (0.7)	2.50×10^{-4} (0.2)	4.96×10^{-3} (1.4)	n/r	5.46×10^{-5} (1.1)	$2.50\times10^{-4*}$ (0.2)	1.07×10^{-4} (0.7)
Herbivorous mammal	3.23×10^{-3} (0.4)	1.50×10^{-4} (1.1)	4.06×10^{-3} (0.5)	4.08×10^{-2} (1.7)	n/r	$4.06\times10^{-3*}$ (0.5)	1.00×10^{-4} (1.3)	3.86×10^{-4} (0.7)
Carnivorous mammal	5.71×10^{-2} (0.8)	0.70^{***} (1.6)	4.00×10^{-7} (3.3)	4.08×10^{-2} (0.6)	n/r	$4.00\times10^{-7*}$ (3.3)	1.00×10^{-4} (1.4)	1.26×10^{-5} (2.1)
Rodent	n/r	2.70×10^{-4} (0.9)	$4.06\times10^{-3\#}$ (0.5)	$4.08\times10^{-2\#}$ (1.6)	n/r	$4.06\times10^{-3*}$ (0.5)	1.00×10^{-4} (1..4)	1.11×10^{-3} (0.2)
Bird egg	6.42×10^{-5} (1.3)	$0.70^{\#}$ (2.7)	$2.00\times10^{-3\#}$ (0.2)	$4.08\times10^{-2\#}$ (1.5)	n/r	n/r	n/r	5.59×10^{-4} (0.4)
Freshwater								
Duck	27.8 (<0.001)	$4.00\times10^{4\#}$ (3.4)	$390^{\#}$ (1.2)	2.00 (1.2)	n/r	n/r	n/r	8640 (2.7)
Amphibian	27.8 (<0.001)	$4.00\times10^{4\#}$ (3.4)	$390^{\#}$ (1.2)	$2.00^{\#}$ (1.2)	n/r	n/r	n/r	n/r
Pelagic fish	30.0 (0.3)	30.0 (0.3)	$17.0^{\#}$ (0.2)	1.80 (2.3)	30.0 (0.3)	50.0 (0.8)	n/r	50.0 (0.8)
Fish egg	n/r	$2.00\times10^{6\#}$ (9.3)	$390^{\#}$ (1.0)	n/r	n/r	50.0 (1.0)	n/r	n/r
Macrophyte	3000 (0.2)	3000 (0.2)	$2900^{\#}$ (0.4)	4200 (0.9)	5000 (1.5)	2020 (1.6)	n/r	n/r
Phytoplankton	n/r	4.00×10^{4} (0.2)	$8000^{\#}$ (1.1)	$4.00\times10^{+4\wedge}$ (0.2)	n/r	9450 (1.0)	n/r	$2.00\times10^{+5}$ (1.6)
Zooplankton	n/r	400 (0.9)	$390^{\#}$ (0.9)	400^{**} (0.9)	1000 (0.2)	2900 (1.5)	n/r	2000 (1.1)
Benthic mollusc	400 (0.4)	100 (2.0)	$390^{\#}$ (0.4)	470 (0.7)	n/r	n/r	400 (0.4)	n/r
Large benthic crustacean	400 (<0.001)	100 (1.0)	$390^{\#}$ (<0.02)	97.0 (1.0)	n/r	6750 (2.0)	400 (<0.001)	n/r
Benthic fish	n/r	30.0 (0.7)	$17.0^{\#}$ (1.1)	350 (0.7)	30.0 (0.7)	2500 (1.8)	100 (<0.001)	n/r
Aquatic mammal	n/r	$4.00\times10^{4\#}$ (3.4)	$390^{\#}$ (1.4)	$2.00^{\#}$ (1.0)	7.00 (0.4)	n/r	n/r	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported.

Table 4.3. Comparison of predicted whole body ^{14}C activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	n/r	350 (1.7)	430 (0.6)	430 (0.6)	n/r	0.48 (75.7)	430* (0.6)	n/r
Shrub	n/r	420 (0.9)	890# (1.0)	890# (1.0)	n/r	375 (1.1)	n/r	n/r
Grass/Herb	n/r	560 (1.1)	890 (0.8)	890 (0.8)	n/r	525 (1.4)	890* (0.8)	n/r
Herbivorous mammal	n/r	750 (1.3)	1340 (0.8)	1340 (0.8)	n/r	800 (1.1)	1340* (0.8)	n/r
Carnivorous mammal	n/r	690 (1.5)	1340 (0.8)	1340 (0.8)	n/r	800 (1.0)	1340* (0.8)	n/r
Rodent	n/r	690 (1.2)	1340# (1.0)	1340 (1.0)	n/r	800 (0.7)	n/r	n/r
Bird egg	n/r	280 (2.0)	890 (0.5)	890 (0.5)	n/r	890* (0.5)	890* (0.5)	n/r
Freshwater								
Duck	n/r	7300# (0.7)	5.00# (22.5)	7300^ (0.7)	n/r	1.49×10^{4} (1.4)	n/r	n/r
Amphibian	n/r	7300# (0.7)	5.00# (>100)	7300^ (0.7)	n/r	6750 (1.4)	n/r	n/r
Pelagic fish	n/r	4600 (0.8)	5.00# (11.9)	4600^ (0.8)	4500 (0.8)	1.13×10^{4} (0.7)	n/r	2.00×10^{4} (1.6)
Fish egg	n/r	$2.00\times10^{+4}$# (1.0)	5.00# (10.1)	n/r	n/r	4500 (1.0)	n/r	n/r
Macrophyte	n/r	4600 (0.5)	5.00# (63.9)	4600^ (0.5)	4500 (0.7)	5850 (1.7)	n/r	n/r
Phytoplankton	n/r	1800 (1.0)	5.00# (8.8)	1800^ (1.0)	n/r	7200 (0.8)	n/r	9000 (1.1)
Zooplankton	n/r	4000 (0.8)	5.00# (11.3)	4000^ (0.8)	$1.00\times10^{+4}$ (0.6)	4500 (0.6)	n/r	2.00×10^{4} (1.7)
Benthic mollusc	n/r	7300 (0.7)	5.00# (11.3)	7300^ (0.7)	n/r	3.15×10^{4} (1.4)	n/r	n/r
Large benthic crustacean	n/r	7300 (1.0)	5.00# (55.1)	7300^ (1.0)	n/r	9900 (1.3)	9100 (0.7)	n/r
Benthic fish	n/r	4600 (1.0)	5.00# (19.8)	4600^ (1.0)	n/r	9900 (1.1)	9100 (0.9)	n/r
Aquatic mammal	n/r	7300# (0.1)	5.00# (20.9)	7300^ (0.1)	6500 (0.4)	1.44×10^{4} (1.8)	5000 (1.2)	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported.

Table 4.4. Comparison of predicted whole body ^{60}Co activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	0.35 (0.5)	0.79# (0.9)	n/r	6.08×10^{-3}# (1.4)	n/r	n/r	n/r	n/r
Shrub	2.00×10^{-2} (0.5)	1.00×10^{-2} (1.0)	n/r	0.75 (1.7)	n/r	n/r	n/r	3.76×10^{-2} (0.2)
Grass/Herb	2.00×10^{-2} (1.7)	1.00×10^{-2} (1.2)	n/r	1.35×10^{-2} (0.1)	n/r	1.07×10^{-2} (0.9)	n/r	1.41×10^{-2} (0.3)
Herbivorous mammal	6.59×10^{-2} (0.9)	0.79# (0.7)	n/r	0.30 (0.03)	n/r	5.0 (1.8)	8.00×10^{-2} (0.8)	8.57×10^{-2} (0.8)
Carnivorous mammal	0.14 (0.6)	0.79# (0.3)	n/r	0.30 (0.2)	n/r	20.8 (2.1)	8.00×10^{-2} (0.9)	0.15 (0.6)
Rodent	n/r	0.79 (1.4)	n/r	0.30 (0.2)	n/r	54.7 (6.6)	8.00×10^{-2} (1.4)	0.22 (0.2)
Bird egg	1.70×10^{-3} (1.2)	0.79# (1.2)	n/r	0.30# (0.8)	n/r	n/r	n/r	4.64×10^{-3} (0.8)
Freshwater								
Duck	167 (1.3)	5000# (1.4)	n/r	437 (0.5)	n/r	n/r	n/r	1539 (0.4)
Amphibian	167 (0.03)	5000# (1.4)	n/r	140 (0.04)	n/r	4.8 (1.4)	n/r	n/r
Pelagic fish	300 (0.6)	300 (0.6)	n/r	437 (1.0)	20 (2.0)	209 (0.3)	n/r	100 (0.5)
Fish egg	n/r	2.0×10^{5}# (<0.001)	n/r	n/r	25 (<0.001)	1672 (0.0)	n/r	n/r
Macrophyte	1000 (0.02)	1000 (0.02)	n/r	3200 (1.5)	275 (1.7)	1240 (0.3)	n/r	n/r
Phytoplankton	n/r	1000 (1.0)	n/r	1000^ (1.0)	n/r	3001 (0.6)	n/r	5000 (1.3)
Zooplankton	n/r	400 (0.2)	n/r	700 (0.6)	200 (0.2)	18.9 (1.8)	n/r	2000 (1.2)
Benthic mollusc	2000 (0.9)	2000 (0.9)	n/r	550 (0.4)	300 (1.0)	2278 (1.0)	200 (1.4)	n/r
Large benthic crustacean	2000 (0.4)	2000 (0.4)	n/r	1500 (0.1)	n/r	3520 (1.0)	200 (1.9)	n/r
Benthic fish	n/r	300 (0.6)	n/r	437 (0.9)	20 (1.9)	133 (0.2)	300 (0.6)	n/r
Aquatic mammal	n/r	5000# (1.0)	n/r	437# (0.4)	80 (1.4)	4188 (0.9)	n/r	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported.

Table 4.5. Comparison of predicted whole body ^{137}Cs activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	0.35 (1.7)	1.30×10^{-2} (1.7)	5.66×10^{-2} (0.1)	8.94×10^{-2} (0.3)	n/r	$5.66\times10^{-2*}$ (0.1)	$5.66\times10^{-2*}$ (0.1)	n/r
Shrub	4.00×10^{-2} (1.6)	0.16 (0.6)	0.67 (0.4	3.97 (1.6)	n/r	1.01^{*} (0.7)	0.67^{*} (0.4)	0.12 (0..8)
Grass/Herb	4.00×10^{-2} (0.6)	0.14 (0.3)	0.58 (1.3)	0.69 (1.5)	n/r	1.22×10^{-2} (1.5)	5.00×10^{-2} (0.4)	4.35×10^{-2} (0.5)
Herbivorous mammal	1.81 (0.1)	2.20 (0.2)	1.84 (0.1)	2.87 (0.4)	n/r	8.62 (1.4)	1.84^{*} (0.1)	0.12 (2.2)
Carnivorous mammal	11.8 (0.6)	9.00 (0.4)	4.96 (0.1)	2.87 (0.5)	n/r	42.2 (1.5)	4.96^{*} (0.1)	0.38 (2.0)
Rodent	n/r	1.30×10^{-2} (2.6)	$4.96^{\#}$ (0.2)	2.87 (0.4)	n/r	94.2 (1.1)	100 (1.1)	0.16 (1.6)
Bird egg	7.46×10^{-3} (1.2)	$9.00^{\#}$ (6.1)	6.40×10^{-2} (1.0)	$3.00\times10^{-2\#}$ (0.3)	n/r	n/r	$6.40\times10^{-2*}$ (1.0)	7.29×10^{-3} (1.2)
Freshwater								
Duck	5.4×10^{4} (2.1)	$1.10\times10^{4\#}$ (0.7)	3000 (0.4)	3000 (0.4)	n/r	3000^{*} (0.4)	3000^{*} (0.4)	1.42×10^{3} (1.1)
Amphibian	5.41×10^{4} (1.2)	$1.10\times10^{4\#}$ (0.2)	$1.22\times10^{4\#}$ (0.3)	9300 (0.1)	0.60 (5.9)	414 (1.8)	n/r	n/r
Pelagic fish	2000 (0.5)	1.10×10^{4} (1.0)	1.02×10^{4} (1.0)	7100 (0.7)	400 (1.9)	6185 (0.5)	4900^{*} (0.3)	1000 (1.1)
Fish egg	n/r	$3000^{\#}$ (0.8)	$1.22\times10^{4\#}$ (1.4)	n/r	5.00 (10.9)	3402 (0.6)	n/r	n/r
Macrophyte	1000 (0.2)	2300 (1.1	1000 (0.2)	1160 (0.3)	100 (2.3)	1337 (0.5)	1000^{*} (0.2)	n/r
Phytoplankton	n/r	180 (1.1)	3400 (1.2)	4700 (1.4)	20.0 (1.1)	33.4 (0.9)	20.0 (1.1)	900 (0.6)
Zooplankton	n/r	20.0 (1.4)	3400 (1.2)	1560 (0.8)	90.0 (0.6)	72.0 (0.7)	3400^{*} (1.2)	100 (0.6)
Benthic mollusc	100 (1.9)	580 (0.2)	1000 (0.9)	460 (0.04)	200 (1.0)	1050 (0.9)	1000^{*} (0.9)	n/r
Large benthic crustacean	100 (0.9)	630 (0.05)	$1.22\times10^{4\#}$ (1.4)	1.04×10^{4} (1.3)	n/r	242 (0.5)	60.0 (1.2)	n/r
Benthic fish	n/r	1.10×10^{4} (1.0)	1.22×10^{4} (1.0)	6300 (0.7)	400 (0.9)	1743 (0.1)	100 (1.7)	n/r
Aquatic mammal	n/r	$1.10\times10^{4\#}$ (0.6)	$1.22\times10^{4\#}$ (0.7)	$9300^{\#}$ (0.4)	700 (2.0)	7514 (0.2)	n/r	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported.

Table 4.6. Comparison of predicted whole body ^{3}H activity concentrations (Bq kg^{-1} (FW)), and the estimated Z-scores (lower number in parenthesis).

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	n/r	150 (0.0)	150 (0.0)	150 (0.0)	n/r	150* (0.0)	150* (0.0)	n/r
Shrub	n/r	150 (0.0)	150# (0.0)	150# (0.0)	n/r	n/r	n/r	n/r
Grass/Herb	n/r	120 (2.0)	150 (0.5)	150 (0.5)	n/r	150* (0.5)	150* (0.5)	n/r
Herbivorous mammal	n/r	130 (2.0)	150 (0.5)	150 (0.5)	n/r	150* (0.5)	150* (0.5)	n/r
Carnivorous mammal	n/r	140 (2.0)	150 (0.5)	150 (0.5)	n/r	150* (0.5)	150* (0.5)	n/r
Rodent	n/r	140 (1.7)	150# (0.6)	150 (0.6)	n/r	150* (0.6)	n/r	n/r
Bird egg	n/r	150 (0.0)	150 (0.0)	150 (0.0)	n/r	150* (0.0)	150* (0.0)	n/r
Freshwater								
Duck	n/r	1 (n/a)	1# (n/a)	1^ (n/a)	n/r	n/r	n/r	n/r
Amphibian	n/r	1 (0.5)	1*** (0.5)	1^ (0.5)	1 (0.5)	1.17 (2.0)	n/r	n/r
Pelagic fish	n/r	1 (0.4)	1# (0.4)	1^ (0.4)	1 (0.4)	1.17 (2.2)	n/r	1 (0.4)
Fish egg	n/r	1 (0.7)	1# (0.7)	n/r	n/r	1.17 (1.4)	n/r	n/r
Macrophyte	n/r	1 (0.5)	1# (0.5)	1^ (0.5)	1 (0.5)	1.08 (2.0)	n/r	n/r
Phytoplankton	n/r	1 (0.5)	1# (0.5)	1^ (0.5)	n/r	1.08 (2.0)	n/r	1 (0.5)
Zooplankton	n/r	1 (0.4)	1# (0.4)	1^ (0.4)	1 (0.4)	1.17 (2.2)	n/r	1 (0.4)
Benthic mollusc	n/r	1 (0.4)	1# (0.4)	1^ (0.4)	1 (0.4)	1.17 (2.2)	1 (0.4)	n/r
Large benthic crustacean	n/r	1 (0.5)	1# (0.5)	1^ (0.5)	n/r	1.17 (2.0)	1 (0.5)	n/r
Benthic fish	n/r	1 (0.4)	1# (0.4)	1 (0.4)	1 (0.4)	1.17 (2.2)	1 (0.4)	n/r
Aquatic mammal	n/r	1 (0.6)	1# (0.6)	1^ (0.6)	n/r	2.57 (1.7)	n/r	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported; n/a not applicable (all predictions were the same).

Table 4.7. Comparison of predicted whole body ^{131}I activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	0.35 (0.7)	1.00# (0.4)	0.41# (0.5)	0.16 (1.5)	n/r	2.0 (1.2)	n/r	2.00 (1.2)
Shrub	0.40 (0.3)	1.00# (1.6)	0.15# (1.0)	0.14# (1.1)	n/r	3.4×10^{-4} (9.4)	n/r	0.40 (0.3)
Grass/Herb	0.40 (0.7)	1.00# (1.9)	0.15 (0.6)	0.14 (0.7)	n/r	3.40×10^{-4} (8.9)	0.15* (0.6)	0.15 (0.6)
Herbivorous mammal	1.31 (0.6)	1.00# (0.3	0.25 (1.1)	0.40# (0.6)	n/r	4.9 (1.9)	0.25* (1.1)	0.89 (0.2)
Carnivorous mammal	2.60 (1.7)	1.00# (0.4)	0.41 (0.9)	0.40# (0.9)	n/r	22.2 (4.8)	0.41* (0.9)	1.18 (0.6)
Rodent	n/r	1.00# (1.7)	0.41# (0.8)	0.40# (0.8)	n/r	53.5 (12.4)	n/r	0.53 (0.1)
Bird egg	0.21 (1.2)	1.00# (1.3)	0.41# (0.1)	160# (9.2)	n/r	n/r	n/r	3.30×10^{-2} (4.1)
Freshwater								
Duck	250 (0.8)	600# (0.2)	3000# (1.0)	130# (1.3)	n/r	n/r	n/r	3.84×10^{3} (1.2)
Amphibian	250 (0.4)	600# (1.8)	130 (0.7)	130 (0.7)	n/r	130* (0.7)	n/r	n/r
Pelagic fish	40.0 (0.3)	40.0 (0.3)	40.0 (0.3)	180 (0.9)	35.0 (0.4)	605 (1.9)	n/r	10.0 (1.4)
Fish egg	n/r	3000# (0.7)	3000# (0.7)	n/r	n/r	605 (1.4)	n/r	n/r
Macrophyte	300 (0.7)	400 (1.1)	200 (0.1)	300 (0.7)	50.0 (1.9)	129 (0.6)	n/r	n/r
Phytoplankton	n/r	200 (1.6)	700 (0.02)	2300 (1.5)	n/r	568 (0.3)	n/r	1000 (0.4)
Zooplankton	n/r	600 (0.5)	3000 (1.0)	1300 (0.2)	5.0 (4.9)	169 (1.7)	n/r	3000 (1.0)
Benthic mollusc	100 (0.6)	170 (1.4)	50.0 (0.4)	25.0 (1.4)	120 (0.9)	34.2 (1.0)	n/r	n/r
Large benthic crustacean	100 (1.3)	170 (0.2)	200 (0.1)	400 (1.5)	n/r	5.1 (7.3)	n/r	n/r
Benthic fish	n/r	40.0 (0.7)	40.0 (0.7)	180 (0.7)	35.0 (0.9)	528 (1.7)	n/r	n/r
Aquatic mammal	n/r	600# (0.4)	3000# (1.5)	130# (0.7)	40.0 (1.5)	464 (0.2)	n/r	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported.

Table 4.8. Comparison of predicted whole body ^{210}Po activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	n/r	$2.60\times10^{-2\#}$ (0.3)	$4.2^{\#}$ (1.3)	$2.78\times10^{-3\#}$ (1.0)	n/r	n/r	n/r	n/r
Shrub	n/r	$2.00\times10^{-3\#}$ (1.5)	0.12 (0.7)	9.85×10^{-2} (0.6)	n/r	0.18 (0.9)	0.12^{*} (0.7)	2.52×10^{-3} (1.3)
Grass/Herb	n/r	$2.00\times10^{-3\#}$ (1.0)	$0.12^{\#}$ (0.6)	0.12 (0.6)	n/r	0.41^{*} (1.1)	n/r	9.45×10^{-4} (1.4)
Herbivorous mammal	n/r	$1.00\times10^{-4\#}$ (2.7)	4.2 (0.8)	$2.78\times10^{-3\#}$ (1.6)	n/r	4.2^{*} (0.8)	4.2^{*} (0.8)	4.87×10^{-2} (0.7)
Carnivorous mammal	n/r	$0.70^{\#}$ (0.3	1.7 (0.7)	$2.78\times10^{-3\#}$ (2.1)	n/r	1.7^{*} (0.7)	1.7^{*} (0.7)	0.14 (0.4)
Rodent	n/r	$5.00\times10^{-4\#}$ (1.4)	$4.2^{\#}$ (1.0)	$2.78\times10^{-3\#}$ (1.0)	n/r	4.2^{*} (1.0)	n/r	0.66 (0.5)
Bird egg	n/r	$0.70^{\#}$ (0.6)	4.2^{***} (1.2)	$2.78\times10^{-3\#}$ (1.4)	n/r	n/r	n/r	4.73×10^{-2} (0.4)
Freshwater								
Duck	n/r	$1.00\times10^{5\#}$ (0.6)	$7.32\times10^{4\#}$ (0.5)	$240^{\#}$ (1.7)	n/r	n/r	n/r	9.00×10^{4} (0.6)
Amphibian	n/r	$1.00\times10^{5\#}$ (0.8)	$7.32\times10^{4\#}$ (0.7)	$240^{\#}$ (1.4)	n/r	n/r	n/r	n/r
Pelagic fish	n/r	50.0 (0.8)	$7.32\times10^{4\#}$ (2.0)	240 (0.2)	50.0 (0.8)	53.3 (0.7)	n/r	1000 (0.4)
Fish egg	n/r	$2.00\times10^{7\#}$ (<0.001)	$7.32\times10^{4\#}$ (0.0)	n/r	n/r	53.3 (<0.001)	n/r	n/r
Macrophyte	n/r	1400 (0.6)	1400 (0.6)	4000 (2.1)	2000 (0.3)	1400^{*} (0.6)	1400^{*} (0.6)	n/r
Phytoplankton	n/r	6000 (0.7)	2.73×10^{4} (0.6)	2.70×10^{4} (0.6)	n/r	1400^{*} (1.9)	$2.73\times10^{4*}$ (0.6)	3.00×10^{4} (0.7)
Zooplankton	n/r	6000 (2.4)	2.73×10^{4} (0.5)	$2.70\times10^{4\#}$ (0.4)	2.00×10^{4} (0.1)	$2.73\times10^{4*}$ (0.5)	$2.73\times10^{4*}$ (0.5)	3.00×10^{4} (0.6)
Benthic mollusc	n/r	$1.00\times10^{5\#}$ (1.5)	7.32×10^{4} (0.8)	3.80×10^{4} (0.6)	n/r	2.76×10^{4} (1.3)	$4.05\times10^{4*}$ (0.4)	n/r
Large benthic crustacean	n/r	$1.00\times10^{5\#}$ (2.0)	1.09×10^{4} (0.4)	9900 (0.6)	n/r	9900 (0.6)	$1.09\times10^{4*}$ (0.4)	n/r
Benthic fish	n/r	50.0 (0.5)	$7.32\times10^{4\#}$ (2.2)	$240^{\#}$ (0.1)	50.0 (0.5)	38.3 (0.6)	50.0 (0.5)	n/r
Aquatic mammal	n/r	$1.00\times10^{5\#}$ (1.1)	$7.32\times10^{4\#}$ (0.9)	$240^{\#}$ (1.1)	400 (0.9)	n/r	n/r	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported.

Table 4.9. Comparison of predicted whole body ^{239}Pu activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	0.35 (1.5)	2.60×10^{-2} (0.3)	$0.22^{\#}$ (1.2)	2.90×10^{-2} (0.2)	n/r	9.12×10^{-3} (1.1)	n/r	9.12×10^{-3} (1.1)
Shrub	1.00×10^{-2} (1.0)	2.00×10^{-3} (0.3)	$1.00\times10^{-4\#}$ (1.0)	3.15×10^{-2} (1.5)	n/r	6.00×10^{-5} (1.2)	n/r	3.20×10^{-4} (0.5)
Grass/Herb	1.00×10^{-2} (1.0)	2.00×10^{-3} (0.3)	1.00×10^{-4} (1.1)	1.44×10^{-2} (1.2)	n/r	5.88×10^{-3} (0.8)	$1.00\times10^{-4*}$ (1.1)	1.20×10^{-4} (1.1)
Herbivorous mammal	3.25×10^{-3} (0.4)	1.00×10^{-4} (1.9)	1.83×10^{-3} (0.04)	2.34×10^{-2} (1.7)	n/r	$1.82\times10^{-3*}$ (0.04)	$1.83\times10^{-3*}$ (0.04)	1.00×10^{-3} (0.4)
Carnivorous mammal	5.74×10^{-2} (1.0)	$0.70^{\#}$ (2.0)	1.60×10^{-7} (4.1)	2.34×10^{-2} (0.7)	n/r	$1.60\times10^{-7*}$ (4.1)	3.00×10^{-3} (0.1)	8.49×10^{-5} (1.6)
Rodent	n/r	5.00×10^{-4} (1.5)	1.83×10^{-3} (0.3)	2.34×10^{-2} (1.9)	n/r	$1.82\times10^{-3*}$ (0.3)	3.00×10^{-3} (0.1)	2.85×10^{-3} (0.1)
Bird egg	8.11×10^{-6} (5.0)	$0.70^{\#}$ (4.1)	$2.00\times10^{-3\#}$ (0.6)	$2.34\times10^{-2\#}$ (1.4)	n/r	n/r	n/r	1.46×10^{-3} (0.8)
Freshwater								
Duck	36.3 (0.2)	2.0 (1.1)	$390^{\#}$ (1.3)	$2.0^{\wedge}$ (1.1)	n/r	n/r	n/r	113.26 (0.7)
Amphibian	36.3 (1.4)	$1.00\times10^{5\#}$ (6.4)	$390^{\#}$ (1.0)	$230^{\#}$ (0.4)	n/r	n/r	n/r	n/r
Pelagic fish	30.0 (0.4)	69.0 (1.2)	17.0 (0.2)	60.0 (1.1)	30.0 (0.4)	5.9 (1.2)	n/r	4.0 (1.6)
Fish egg	n/r	$1.00\times10^{5\#}$ (1.0)	$390^{\#}$ (1.0)	n/r	n/r	n/r	n/r	n/r
Macrophyte	890 (0.8)	1800 (0.1)	2900 (0.5)	2600 (0.4)	350 (1.9)	6870 (1.5)	2900^{*} (0.5)	n/r
Phytoplankton	n/r	180 (1.9)	8000 (0.7)	5900 (0.4)	n/r	1.19×10^{4} (0.9)	8000^{*} (0.7)	900 (0.8)
Zooplankton	n/r	20.0 (1.9)	390 (0.6)	450 (0.7)	100 (0.6)	939 (1.3)	390^{*} (0.6)	100 (0.6)
Benthic mollusc	100 (1.2)	820 (1.0)	$390^{\#}$ (0.3)	$820^{\wedge}$ (1.0)	n/r	2.38×10^{5} (7.0)	100 (1.2)	n/r
Large benthic crustacean	100 (1.0)	140 (0.7)	$390^{\#}$ (0.1)	1100 (1.0)	n/r	2348 (1.6)	100 (1.0)	n/r
Benthic fish	n/r	69.0 (1.4)	$17.0^{\#}$ (0.8)	60.0 (1.2)	30.0 (0.1)	14.8 (1.1)	17.0^{*} (0.8)	n/r
Aquatic mammal	n/r	230 (0.7)	$390^{\#}$ (1.4)	$230^{\wedge}$ (0.7)	0.15 (30.2)	n/r	n/r	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported.

Table 4.10. Comparison of predicted whole body ^{226}Ra activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	0.35 (0.9)	1.1# (2.1)	8.14×10^{-2} (0.6)	$9.00\times10^{-2\#}$ (0.5)	n/r	7.50×10^{-2} (0.7)	$8.14\times10^{-2*}$ (0.6)	7.50×10^{-2} (0.7)
Shrub	0.10 (0.3)	2.20×10^{-1} (0.5)	0.27 (0.7)	2.40×10^{-2} (1.7)	n/r	0.41* (1.1)	0.27* (0.7)	4.00×10^{-2} (1.2)
Grass/Herb	0.10 (1.1)	1.90×10^{-1} (1.7)	2.00×10^{-2} (0.5)	3.94×10^{-2} (0.2)	n/r	8.61×10^{-3} (1.3)	$2.00\times10^{-2*}$ (0.5)	1.50×10^{-2} (0.8)
Herbivorous mammal	0.23 (0.6)	1.1 (1.9)	4.13×10^{-2} (0.8)	2.65×10^{-2} (1.1)	n/r	0.24* (0.7)	$4.13\times10^{-2*}$ (0.8)	5.53×10^{-2} (0.002)
Carnivorous mammal	0.55 (1.1)	1.1# (1.5)	3.53×10^{-2} (0.8)	2.65×10^{-2} (1.0)	n/r	0.37* (0.8)	$3.53\times10^{-2*}$ (0.8)	4.16×10^{-2} (0.7)
Rodent	n/r	2.30×10^{-2} (1.2)	6.01×10^{-2} (0.2)	2.65×10^{-2} (1.1)	n/r	0.24* (1..3)	0.21 (1.2)	7.11×10^{-2} (0.1)
Bird egg	7.29×10^{-3} (1.2)	1.1# (4.1)	$8.14\times10^{-2\#}$ (1.4)	$3.62\times10^{-2\#}$ (0.5)	n/r	n/r	n/r	1.17×10^{-2} (6.4)
Freshwater								
Duck	802 (6.0)	2.0# (1.7)	8.00×10^{-2} (0.6)	80.0# (4.4)	n/r	$8.00\times10^{-2*}$ (0.6)	$8.00\times10^{-2*}$ (0.6)	1321 (1.7)
Amphibian	802 (1.0)	$1.00\times10^{5\#}$ (5.7)	750# (1.0)	80.0# (0.3)	n/r	122 (0.8)	n/r	n/r
Pelagic fish	50.0 (0.1)	69.0# (0.1)	10.0 (1.4)	80.0 (0.1)	50.0 (0.1)	179 (0.9)	10.0* (1.4)	500 (1.0)
Fish egg	n/r	$1.00\times10^{5\#}$ (2.1)	750# (0.3)	n/r	n/r	179 (0.3)	n/r	n/r
Macrophyte	3.00×10^{4} (2.1)	1800# (0.1)	2000 (0.1)	1800 (0.1)	260 (1.6)	2033 (0.04)	2000* (0.1)	n/r
Phytoplankton	n/r	180# (1.3)	1100 (0.4)	1100 (0.4)	100 (1.8)	1513 (0.7)	1100* (0.4)	2000 (1.0)
Zooplankton	n/r	20.0# (1.6)	750# (1.0)	1100# (1.3)	250 (0.2)	86.9 (0.5)	n/r	100 (0.4)
Benthic mollusc	250 (1.4)	820# (0.1)	330 (1.1)	1500 (0.6)	n/r	1871 (0.8)	2700* (1.3)	n/r
Large benthic crustacean	250 (1.0)	140# (1.6)	750 (0.1)	1500 (0.8)	n/r	920 (0.3)	2400* (1.3)	n/r
Benthic fish	n/r	69.0# (0.4)	10.0# (2.1)	80.0 (0.6)	50.0 (0.02)	122 (1.1)	50.0 (0.02)	n/r
Aquatic mammal	n/r	230# (n/a)	2.00×10^{-2} (n/a)	80.0# (n/a)	n/r	$2.00\times10^{-2*}$ (n/a)	$2.00\times10^{-2*}$ (n/a)	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported n/a not applicable (all predictions which were not outlier were the same value).

Table 4.11. Comparison of predicted whole body ^{90}Sr activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	0.35 (1.3)	5.0# (1.1)	2.0# (0.2)	8.97×10^{-3} (4.7)	n/r	n/r	n/r	n/r
Shrub	0.30 (1.5)	1.70×10^{-2} (1.9)	0.11 (0.3	4.96×10^{-2} (0.7)	n/r	0.16 (0.7)	0.11* (0.3)	7.60×10^{-2} (0.2)
Grass/Herb	0.30 (0.9)	5.0# (4.5)	0.17 (0.2)	0.21 (0.4)	n/r	0.17 (0.1)	0.25 (0.6)	2.85×10^{-2} (2.2)
Herbivorous mammal	17.5 (1.9)	5.0# (0.3)	2.0 (1.0)	1.7 (1.1)	n/r	3.8* (0.1)	3.8* (0.1)	0.19 (4.1)
Carnivorous mammal	770 (5.0)	5.0# (0.7)	1.3 (0.5)	1.7 (0.2)	n/r	7.0* (1.0)	7.0* (1.0)	0.26 (1.9)
Rodent	n/r	5.0 (1.3)	2.0# (0.2)	1.7 (0.1)	n/r	3.8* (1.0)	0.6 (1.1)	0.46 (1.4)
Bird egg	7.96×10^{-3} (10.6)	5.0# (1.4)	2.0# (0.4)	1.4# (1.0)	n/r	n/r	n/r	0.11 (5.7)
Freshwater								
Duck	6270 (3.8)	1200# (1.4)	300# (0.6)	17.0# (4.7)	n/r	n/r	n/r	252 (0.8)
Amphibian	6270 (2.4)	1200# (1.3	300# (0.4)	17.0# (1.6)	20.0 (1.5)	130 (0.2)	n/r	n/r
Pelagic fish	60.0 (1.3)	43.0 (0.9)	25.0 (0.3)	17.0 (0.2)	5.0 (1.6)	518 (3.9)	n/r	10.0 (0.8)
Fish egg	n/r	1000# (42.2)	300# (31.7)	n/r	7.0 (1.0)	8.8 (1.0)	n/r	n/r
Macrophyte	3000 (1.8)	1200 (1.0)	150 (0.7)	250 (0.3)	80.0 (1.2)	376 (0.1)	150* (0.7)	n/r
Phytoplankton	n/r	40.0 (0.2)	40.0 (0.2)	40.0 (0.2)	15.0 (1.6)	70.7 (0.6)	40.0* (0.2)	200 (2.0)
Zooplankton	n/r	20.0 (0.9)	60.0 (0.6)	60.0 (0.6)	25.0 (0.6)	11.3 (1.7)	60.0* (0.6)	100 (1.3)
Benthic mollusc	100 (0.6)	250 (0.5)	300 (0.7)	270 (0.6)	30.0 (2.0)	9000 (4.8)	300* (0.7)	n/r
Large benthic crustacean	100 (1.2)	270 (0.9)	300# (1.1)	200 (0.3)	n/r	n/r	100 (1.3)	n/r
Benthic fish	n/r	43.0 (0.8)	25.0# (0.1)	17.0 (0.3)	5.0 (1.7)	1054 (4.5)	60.0 (1.1)	n/r
Aquatic mammal	n/r	1200# (1.0)	300# (1.0)	17.0# (5.1)	n/r	n/r	n/r	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported.

Table 4.12. Comparison of predicted whole body ^{99}Tc activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis).

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	0.35 (0.6)	1.30×10^{-2}# (1.7)	0.37# (0.6)	0.37# (0.6)	n/r	n/r	n/r	n/r
Shrub	8.0 (0.6)	0.16# (1.8)	2.0# (0.2)	20.0# (1.2)	n/r	n/r	n/r	3.24 (0.1)
Grass/Herb	8.0 (0.9)	0.14# (1.9)	2.0 (0.04)	20.0 (1.6)	n/r	1.7 (0.1)	2.0* (0.04)	1.2 (0.4)
Herbivorous mammal	1.8 (1.2)	2.2# (1.5)	0.37 (0.9)	0.37# (0.9)	n/r	0.37* (0.9)	0.37* (0.9)	1.29 (0.8)
Carnivorous mammal	4.1 (1.3)	9.0# (1.7)	0.10 (0.9)	0.37# (0.1)	n/r	0.10* (0.9)	0.10* (0.9)	0.31 (0.2)
Rodent	n/r	1.30×10^{-2}# (2.0)	0.37# (0.4)	0.37# (0.4)	n/r	0.37* (0.4)	n/r	0.74 (0.9)
Bird egg	2.0 (0.1)	9.0# (0.8)	0.37# (0.7)	27.0# (1.3)	n/r	n/r	n/r	0.06 (1.5)
Freshwater								
Duck	28.8 (0.2)	1300# (1.8)	5.0# (1.2)	40.0# (0.1)	n/r	n/r	n/r	25.7 (0.3)
Amphibian	28.8 (0.3)	1300# (1.6)	5.0# (1.2)	40.0# (0.1)	n/r	n/r	n/r	n/r
Pelagic fish	20.0 (0.3)	45.0 (1.3)	5.0# (1.4)	40.0 (1.1)	20 (0.3)	5.0 (1.4)	n/r	15 (0.1)
Fish egg	n/r	8000# (1.0)	5.0# (1.0)	n/r	n/r	n/r	n/r	n/r
Macrophyte	5000 (1.3)	1300 (0.8)	5.0# (1.2)	1300^ (0.8)	40.0 (0.5)	5.3 (1.2)	n/r	n/r
Phytoplankton	n/r	8.0 (0.2)	5.0# (0.8)	8.0^ (0.2)	n/r	11.6 (0.3)	4.0 (1.1)	40 (2.0)
Zooplankton	n/r	20.0 (0.1)	5.0# (1.2)	20.0^ (0.1)	5.0 (1.2)	n/r	100 (1.3)	100 (1.3)
Benthic mollusc	100 (1.8)	24.0 (0.5)	5.0# (1.0)	24.0^ (0.5)	n/r	6.7 (0.7)	5.0 (1.0)	n/r
Large benthic crustacean	100 (1.8)	13.0 (0.2)	5.0# (1.1)	13.0^ (0.2)	n/r	33.9 (0.7)	5.0 (1.1)	n/r
Benthic fish	n/r	45.0 (1.0)	5.0# (0.9)	40.0 (0.9)	20 (0.3)	2.0 (1.7)	20 (0.3)	n/r
Aquatic mammal	n/r	1300# (1.6)	5.0# (0.7)	40.0# (0.1)	2.9 (1.0)	n/r	n/r	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported.

Table 4.13. Comparison of predicted whole body ^{232}Th activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	0.35 (2.2)	$1.00\times10^{-2\#}$ (0.3)	$1.18\times10^{-2\#}$ (0.1)	$8.84\times10^{-3\#}$ (0.3)	n/r	5.00×10^{-3} (0.7)	n/r	5.00×10^{-3} (0.7)
Shrub	1.00×10^{-3} (1.2)	1.00×10^{-2} (0.5)	8.81×10^{-3} (0.4)	1.60×10^{-2} (0.9)	n/r	$1.32\times10^{-2*}$ (0.8)	$8.81\times10^{-3*}$ (0.4)	4.40×10^{-4} (1.9)
Grass/Herb	1.00×10^{-3} (0.5)	1.00×10^{-2} (0.9)	2.75×10^{-3} (0.1)	4.37×10^{-2} (1.7)	n/r	6.09×10^{-4} (0.8)	$2.75\times10^{-3*}$ (0.1)	1.65×10^{-4} (1.5)
Herbivorous mammal	2.65×10^{-3} (0.3)	1.00×10^{-4} (1.5)	7.74×10^{-3} (0.9)	1.22×10^{-4} (1.4)	n/r	$1.18\times10^{-2*}$ (1.1)	$7.74\times10^{-3*}$ (0.9)	7.83×10^{-4} (0.4)
Carnivorous mammal	2.82×10^{-2} (1.1)	$1.00\times10^{-2\#}$ (0.7)	5.52×10^{-3} (0.4)	1.22×10^{-4} (1.3)	n/r	$5.52\times10^{-3*}$ (0.4)	$5.52\times10^{-3*}$ (0.4)	4.98×10^{-5} (1.7)
Rodent	n/r	$1.00\times10^{-2\#}$ (0.8)	1.18×10^{-2} (0.9)	1.22×10^{-4} (1.9)	n/r	7.74×10^{-3} (0.7)	1.00×10^{-3} (0.6)	2.94×10^{-3} (0.1)
Bird egg	6.19×10^{-5} (1.5)	$1.00\times10^{-2\#}$ (1.0)	$1.18\times10^{-2\#}$ (1.1)	$3.89\times10^{-4\#}$ (0.6)	n/r	n/r	n/r	1.42×10^{-3} (0.04)
Freshwater								
Duck	121 (0.6)	$1.00\times10^{4\#}$ (1.5)	$50.0^{\#}$ (1.1)	$110^{\#}$ (0.7)	n/r	n/r	n/r	2241 (0.8)
Amphibian	121 (0.4)	$1.00\times10^{4\#}$ (1.7)	$50.0^{\#}$ (0.8)	$110^{\#}$ (0.5)	n/r	n/r	n/r	n/r
Pelagic fish	100 (0.5)	100 (0.5)	50.0 (1.3)	110 (0.4)	100 (0.5)	542 (1.5)	n/r	600 (1.6)
Fish egg	n/r	$2.00\times10^{6\#}$ (<0.001)	$50.0^{\#}$ (<0.001)	n/r	n/r	3325 (0.0)	n/r	n/r
Macrophyte	3000 (1.1)	3000 (1.1)	1200 (0.2)	1260 (0.2)	500 (0.7)	n/r	170 (1.8)	n/r
Phytoplankton	n/r	4000 (0.2)	$1200^{\#}$ (1.6)	$4000^{\wedge}$ (0.2)	8800 (0.7)	n/r	3700 (0.3)	2.00×10^{4} (1.7)
Zooplankton	n/r	2000 (0.5)	$50.0^{\#}$ (1.7)	$2000^{\wedge}$ (0.5)	500 (0.3)	292 (0.6)	n/r	1.00×10^{4} (1.5)
Benthic mollusc	500 (1.2)	100 (0.5)	$50.0^{\#}$ (1.3)	$100^{\wedge}$ (0.5)	n/r	n/r	500 (1.2)	n/r
Large benthic crustacean	500 (1.0)	100 (0.4)	$50.0^{\#}$ (1.0)	$100^{\wedge}$ (0.4)	30.0 (1.4)	724 (1.3)	500 (1.0)	n/r
Benthic fish	n/r	100 (0.4)	$50.0^{\#}$ (0.9)	110 (0.4)	n/r	3750 (1.7)	n/r	n/r
Aquatic mammal	n/r	$1.00\times10^{4\#}$ (12.4)	$50.0^{\#}$ (1.0)	$110^{\#}$ (1.0)	0.50 (12.7)	n/r	n/r	n/r

*FASSET value used; $^{\wedge}$EA R&D128 value used; $^{\#}$Value derived from guidance methodology; n/r not reported.

Table 4.14. Comparison of predicted whole body ^{238}U activity concentrations (Bq kg^{-1} (FW)), and estimated Z-scores (lower number in parenthesis). Shaded cells denote identified outlying predictions.

Organism	RESRAD-BIOTA	EA R&D128	FASSET	ERICA	ECOMOD	AECL	LIETDOS- BIO	DosDiM-Eco
Terrestrial								
Earthworm	0.35 (0.5)	0.70# (0.9)	2.91×10^{-3}# (2.0)	8.84×10^{-3} (1.4)	n/r	n/r	n/r	n/r
Shrub	4.00×10^{-3} (0.8)	2.00×10^{-3} (1.6)	1.43×10^{-2} (0.6)	7.06×10^{-3} (0.2)	n/r	2.0 (5.9)	1.43×10^{-2}* (0.6)	3.40×10^{-2} (1.5)
Grass/Herb	4.00×10^{-3} (0.5)	2.00×10^{-3} (1.5)	5.75×10^{-3} (0.1)	1.46×10^{-2} (1.5)	n/r	3.15×10^{-3} (0.8)	5.75×10^{-3}* (0.1)	1.28×10^{-2} (1.3)
Herbivorous mammal	2.33×10^{-2} (2.1)	4.00×10^{-3} (0.1)	1.80×10^{-3} (0.8)	1.06×10^{-4} (4.0)	n/r	2.91×10^{-3}* (0.3)	1.80×10^{-3}* (0.8)	2.67×10^{-3} (0.4)
Carnivorous mammal	5.40×10^{-2} (5.5)	0.70# (8.2)	7.09×10^{-4} (0.8)	1.06×10^{-4} (1.2)	n/r	7.09×10^{-4}* (0.8)	7.09×10^{-4}* (0.8)	1.06×10^{-4} (1.2)
Rodent	n/r	2.00×10^{-3} (0.6)	2.91×10^{-3}# (0.9)	1.06×10^{-4} (1.9)	n/r	5.50×10^{-4} (0.5)	1.00×10^{-3} (0.03)	3.55×10^{-3} (1.0)
Bird egg	1.57×10^{-2} (1.9)	0.70# (5.3)	2.00×10^{-3} (0.1)	5.41×10^{-4}# (1.1)	n/r	2.00×10^{-3} (0.1)	2.00×10^{-3}* (0.1)	5.97×10^{-4} (1.0)
Freshwater								
Duck	29.7 (0.7)	6500# (5.3)	200# (1.4)	30.0# (0.7)	n/r	n/r	n/r	1.35 (4.2)
Amphibian	29.7 (0.7)	6500# (5.3)	200# (1.4)	30.0# (0.7)	n/r	n/r	n/r	n/r
Pelagic fish	10.0 0.4)	10 (0.4)	200 (3.1)	30.0 (1.4)	10.0 (0.4)	2.4 (0.9)	n/r	1.0 (1.7)
Fish egg	n/r	1000# (1.0)	200# (1.0)	n/r	n/r	4.5 (5.7)	n/r	n/r
Macrophyte	900 (0.5)	6500 (1.4)	2800 (0.6)	2900 (0.6)	1000 (0.4)	293 (1.7)	n/r	n/r
Phytoplankton	n/r	4.0 (1.8)	120 (0.5)	120 (0.5)	100 (0.3)	390 (1.3)	n/r	20 (0.7)
Zooplankton	n/r	1.0 (1.6)	50.0 (0.6)	48.0 (0.6)	100 (1.0)	n/r	n/r	5.0 (0.7)
Benthic mollusc	60.0 (1.2)	180 (0.8)	200# (0.9)	180^ (0.8)	n/r	n/r	60.0 (1.2)	n/r
Large benthic crustacean	60.0 (0.5)	180 (0.5)	200# (0.6)	500 (1.5)	n/r	18.0 (1.6)	60.0 (0.5)	n/r
Benthic fish	n/r	10.0 (0.3)	200# (1.6)	30.0 (0.4)	10.0 (0.3)	1.7 (1.4)	n/r	n/r
Aquatic mammal	n/r	6500# (4.7)	200# (1.0)	30.0# (1.0)	n/r	n/r	n/r	n/r

*FASSET value used; ^EA R&D128 value used; #Value derived from guidance methodology; n/r not reported.

4.4. Statistical analyses

Predicted whole-body activity concentrations for the selected radionuclides had a considerable statistical spread. In some cases there are extreme variations resulting from three of the approaches (EA R&D128, FASSET and ERICA) making assumptions to derive default CR values due to the lack of empirical CR values (see Tables 4.2–4.14). The results do not lend themselves to the same statistical treatment as was possible for our previous analyses of DCC values (Chapter 3) because there is a large range in predicted values which tend to be lognormally distributed. Consequently, we have performed normality tests on the log-transformed data and used the geometric mean and standard deviation as the representative values of the data set. Analysis was carried out on a radionuclide-by-radionuclide basis.

The first step of the analysis was the estimation of the geometric mean of predicted activity concentrations for each radionuclide-organism combination. Data for an individual approach for a given radionuclide-organism were then divided by the corresponding geometric mean. This effectively normalised the data into values indicative of departure from unity and thus enabled treating all the information as a single, more statistically representative dataset of N (number of organisms = 19) × M (number of models = 8) = 152 values for each radionuclide. It was anticipated that, due to the large spread of the data, treating all the information for a given radionuclide as a larger set of 152 values rather than individual subsets of 8 values would increase the statistical significance of the analysis.

The normalised data, as well as the natural logarithms of these data, were then subjected to the following statistical tests: the Shapiro-Wilk test to determine if the results are normally distributed; the D'Agostino's test for skewness; the Anscombe-Glynn's test for kurtosis (Table 4.15). Generally, the logarithmically transformed data were found to follow normal distributions, indicating that the data themselves were log-normal distributed. This justifies the use of the geometric mean, rather than the arithmetic, as a representative value for the population.

Inspection of the normalised logarithm data (visually as a graph for each radionuclide) and interpretation of the statistical tests enabled the iterative process of identification of outliers for exclusion; the Grubb's outlier test was applied, where necessary, to confirm that these data were genuine outliers (outliers are identified in Tables 4.2–4.14). The geometric mean for each organism was recalculated as outliers were removed to generate a 'robust mean' free from outliers. This approach provides a consistent method for identifying a central value and associated measure of dispersion for the purposes of comparison. No value judgement is passed on whether extreme predicted values (identified as outliers) represent erroneous data (as there are no reference values against which to base such a judgement).

As the data considered here were log-normally distributed, for the purposes of this comparison the Z-value has been estimated as:

$$Z = \frac{\ln A_i - \ln \mu_g}{\ln \sigma_g} \tag{4.1}$$

where:

A_i is the activity concentration of an organism;
μ_g is the geometric mean; and
σ_g is the geometric standard deviation.

The resultant Z-scores are presented within Tables 4.2–4.14.

Table 4.15. Performance statistics presented by radionuclide.

Nuclide*	Original dataset			Modified dataset with outliers removed			
	Shapiro	D'Agostino	Anscombe	Shapiro	D'Agostino	Anscombe	Grubbs
^{241}Am	0.004	0.35	0.04	0.08	0.54	0.69	0.37
^{14}C	<0.001	0.001	0.03	0.20	0.41	0.74	0.34
^{60}Co	0.27	0.51	0.18	n/r	n/r	n/r	0.21
^{137}Cs	0.01	0.07	0.005	0.26	0.57	0.30	0.14
^{131}I	<0.001	0.10	<0.001	0.91	0.70	0.34	0.15
^{210}Po	0.26	0.53	0.24	n/r	n/r	n/r	0.23
^{239}Pu	0.001	0.36	0.002	0.19	0.95	0.06	0.10
^{226}Ra	0.56	0.86	0.49	n/r	n/r	n/r	0.33
^{90}Sr	0.03	0.51	0.08	0.06	0.93	0.26	0.33
^{99}Tc	0.48	0.75	0.87	n/r	n/r	n/r	0.93
^{232}Th	0.18	0.84	0.07	n/r	n/r	n/r	0.07
^{238}U	0.01	0.08	0.08	0.09	0.27	0.19	0.20

*^{3}H not presented as all predictions were so closely grouped that no outliers were identified, n/r = additional test not required to assess robustness.

4.4.1. Evaluation of statistical analyses

The following observations can be made from an evaluation of the identified outlying predictions and Z-scores (note no organism-radionuclide combinations for which less than four participants made predictions are considered in the following list):

^{241}Am (see Table 4.2)

- EA R&D128 – comparatively high predictions6 for a number of organisms.
- AECL, FASSET and DosDiMEco – comparatively low prediction for carnivorous mammal.
- DosDiMEco – comparatively high prediction for duck.

^{14}C (see Table 4.3)

- FASSET – comparatively low predictions for all freshwater organisms.
- AECL – comparatively very low prediction for earthworm.

^{60}Co (see Table 4.4)

- AECL – comparatively high predictions for rodent and carnivorous mammal.

^{137}Cs (see Table 4.5)

- EA R&D128 – comparatively very low prediction for rodent and high prediction for bird egg.
- ECOMOD – comparatively low predictions for fish egg and amphibian.

^{3}H (see Table 4.6)

- AECL – relatively high Z-scores for most freshwater organisms.

[6] Note where comparatively high or low predicted activity concentrations (i.e. outliers) are noted relatively high z-scores can also be assumed.

^{131}I (see Table 4.7)

- AECL – comparatively high prediction for rodent.
- ERICA – comparatively high prediction for bird egg.
- ECOMOD – comparatively low prediction for zooplankton.
- DosDiMEco – comparatively low prediction for bird egg.

^{210}Po (see Table 4.8)

- EA R&D128 – comparatively low prediction for herbivorous mammal and high prediction for fish egg.

^{239}Pu (see Table 4.9)

- RESRAD-BIOTA – comparatively low prediction for bird egg.
- EA R&D128 – comparatively high predictions for a number of organisms.
- AECL – comparatively very high value for benthic mollusc.
- FASSET and AECL – comparatively low prediction for carnivorous mammal.
- ECOMOD – comparatively very low value for aquatic mammal.

^{226}Ra (see Table 4.10)

- RESRAD-BIOTA and DosDiMEco – comparatively high prediction for duck.
- ERICA and EA R&D128 – comparatively high prediction for aquatic mammal.
- EA R&D128 – comparatively high predictions for bird egg, amphibian and fish egg.

^{90}Sr (see Table 4.11)

- RESRAD-BIOTA – comparatively high predictions for three organisms whilst comparatively low prediction for bird egg.
- AECL – comparatively high predictions for a number of freshwater organisms.
- EA R&D128 – comparatively high predictions for grass/herb and fish egg.
- ERICA – comparatively low for four organisms.
- DosDiMEco – comparatively low prediction for bird egg and herbivorous mammal.

^{99}Tc (see Table 4.12)

- Notably there were no identified outliers and few high Z-scores for this radionuclide.

^{232}Th (see Table 4.13)

- EA R&D128 – comparatively high predictions for aquatic mammal and fish egg, and comparatively low prediction for herbivorous mammal.
- ECOMOD – comparatively low value for aquatic mammal.

^{238}U (see Table 4.14)

- RESRAD-BIOTA – comparatively high prediction for carnivorous mammal.
- FASSET – comparatively low prediction for earthworm and high prediction for pelagic fish.
- ERICA – comparatively low prediction for herbivorous mammal.
- AECL – comparatively high prediction for shrub.
- EA R&D128 – comparatively high prediction for a number of organisms.
- DosDiMEco – comparatively low prediction for duck.

Potential reasons for some of these observations are explored below.

4.5. Discussion

A limitation of the exercise is that there was not a complete set of predictions for any radionuclide. Four of the approaches, AECL, EA R&D128, ERICA and FASSET, submitted predictions for most of the required radionuclide-organism combinations. It is perhaps not surprising, therefore, that there is a tendency for some of these four approaches to appear more often than other approaches, which submitted less results in the list of identified outlying predictions above. Furthermore, few predictions for any one of the outputs specified in the exercise may lead to false identification of outliers. An illustrative example of this is the predictions of ^{232}Th activity concentrations in fish egg (Table 4.13) for which two, of the three submitted, predictions are identified as outliers.

Two of the approaches, AECL and LIETDOS-BIO obtained some of their CR values from FASSET documentation, whilst ERICA is an evolution of the FASSET approach and also uses some freshwater CR values from EA R&D128. This commonality in model parameters influenced comparisons with other approaches in some instances. Examples of this are ^{210}Po and ^{90}Sr predictions for some of the terrestrial mammalian organisms considered, and ^{226}Ra predictions for duck.

The FASSET documentation [26] presents more than one value in a number of instances. Consequently, the 'FASSET' values quoted by LIETDOS-BIO and AECL are not always the same as those cited for FASSET in Tables 4.2–4.14. This is an illustration of the subjectivity which may be encompassed in some of the reported predictions. Three of the approaches (EA R&D128, FASSET and ERICA) provide guidance on how to select appropriate CR values in the absences of empirically derived estimates. Those values which have been reported here using such guidance (see Tables 4.2–4.14) are subject to the interpretation of those using the guidance. A similar caveat applies to predictions of the RESRAD-BIOTA model for terrestrial organisms. The RESRAD-BIOTA predictions were Level 3 species-specific results and rely upon the operator selecting representative species with associated characteristics (e.g. dietary composition). The CR value approach as utilized by most of the other models can be implemented in RESRAD-BIOTA within Level 2 analyses.

As already discussed there was considerable variation in predicted whole-body activity concentrations for some radionuclide-organism combinations. Whilst at this stage in our comparison of the available approaches, we make no attempt to define 'the correct' prediction. However, explanation for some of the observations listed in Section 4.3 above can be given.

Predictions by the EA R&D128 approach represented more than 50 % of the comparatively high outlying predictions. These are generally values estimated from 'guidance methodology' rather than empirically derived values. However, this screening approach aims to be conservative [6] and consequently when it gives rise to comparatively high predictions in this exercise it meets the originators objectives. An example of guidance methodology being applied and resulting in comparatively high predictions is the application of marine fish egg CR values to estimate activity concentrations for all radionuclides in freshwater fish eggs (e.g. see results for ^{241}Am, ^{90}Sr and ^{232}Th). The guidance presented within the ERICA methodology, and used on identified occasions here, represents an evolution of the Copplestone et al. [6] approach. However, whilst aiming to be conservative it is less so than the latter approach and this is reflected in the results of this exercise.

The FASSET prediction of whole-body ^{241}Am activity concentrations in the whole-body of carnivorous mammals (Table 4.2) is comparatively low compared to estimates from most other participating approaches (note AECL use the FASSET CR value in this instance). The FASSET value is a prediction of the FASTer model [29] which utilises similar allometric expressions to define radionuclide behaviour in animals to those used within RESRAD-BIOTA. The difference in predictions between RESRAD-BIOTA and FASSET must presumably largely be a consequence of representative species, food sources, diet fractions, prediction times and the soil-to-plant transfer of radionuclides

The comparatively low predictions of ^{14}C in freshwater organisms by FASSET (Table 4.3) are all based upon guidance presented in Brown et al. [26] to use the sediment-water distribution coefficient (K_d) to provide a conservative estimate if no CR values for any freshwater organisms are available (FASSET presents no default CR values for ^{14}C in freshwater ecosystems). However, this is a misrepresentation of the original text on which it was based [6] which notes that the application of K_d values to provide conservative CR values is not applicable to radioisotopes, such as ^{14}C, which are likely to be present as anions.

Compared to other participants the EA R&D128 approach predicts very low ^{137}Cs activity concentrations in rodents (Table 4.5). The CR value used by EA R&D128 for this prediction was based upon the results for *Microtus agrestis* (field vole) from a coastal sand dune ecosystem close to the Sellafield reprocessing plant [116]. Characteristics of the site (i.e. 'soil' comprised virtually only of sand) and contamination routes (predominantly aerial discharges from the Sellafield plant with some sea-land transfer of marine discharges also probable) mean that the CR value used by EA R&D128 is likely to be highly site specific. Some of the overall variability demonstrated in predictions in this exercise may also be attributable to the data sources used in the different approaches. For instance, AECL aim to use Canadian literature if available, both ECOMOD and LIETDOS-BIO used Russian language publications for a number of predictions and EA R&D128 targeted CR values from the UK where possible.

The Z-scores for AECL predictions of ^{3}H activity concentrations in freshwater biota were comparatively high (Table 4.6). However, when the actual data are inspected it can be seen that for all organisms ^{3}H predictions by the different approaches are similar (reflecting the similar modelling assumptions used by all participants). The high Z-scores for AECL predictions for freshwater organisms reflects the identical predictions made by all other approaches compared to the AECL predictions which differ from the others by <20 % in all cases other than for aquatic mammal.

The ERICA prediction of the ^{131}I activity concentration in bird egg was high compared to predictions by all other approaches (see Table 4.7). In the absence of any empirical data the ERICA default value is based upon the comparative transfers of radioiodine to meat and eggs from the feed of domesticated poultry (from IAEA 1994). The ratio of these transfers is then applied to the predicted whole-body activity concentrations ^{131}I terrestrial wild birds. However, as the majority of the body burden of ^{131}I will be in the thyroid the use of a meat to egg activity concentration ratio is likely to result in an over prediction.

The EA R&D128 predictions for ^{210}Po were comparatively low for herbivorous mammal compared to the other reporting approaches (Table 4.8). However, AECL and LIETDOS-BIO both cited FASSET documentation (CR=4.2) as the source of most of their terrestrial CR values. The FASSET database for terrestrial animals for ^{210}Po contained data for reindeer only. The air-lichen-reindeer pathway is unlikely to be representative of contamination routes for other terrestrial animals and will result in over predictions and alternative data were found on which CR values could be estimated. The ERICA ^{210}Po CR value for terrestrial mammals (2.78×10^{-3}) contains no reindeer data and is more comparable to that used by EA R&D128 (1×10^{-4}).

Technecium-99 stands out as there were no outlying predictions. However, this is rather misleading and should not be taken to infer that we are collectively better able to make predictions for Tc. Conversely, it may be a reflection of the fewer available environmental measurements of transfer. For example, the ERICA methodology contained only one empirically derived ^{99}Tc CR value for terrestrial organisms relevant to this exercise, and that was the recommended value for transfer to grass taken from IAEA (1994). The values for mammals predicted by FASSET and ERICA use allometric relationships suggested by USDOE [7] and which are incorporated into the RESRAD-BIOTA methodology. Both AECL and LIETDOS-BIO use the values suggested in the FASSET methodology. Hence, there is a tendency for all approaches to obtain information from the same few available sources. Compare this to the situation for more studied radionuclides such as ^{137}Cs and ^{90}Sr for which there may be many hundreds of observations on which to base recommended transfer parameters. As a consequence of this large amount of data, the individual approaches may select very different transfer parameters.

A number of organisms, namely, carnivorous mammal, fish egg, bird egg, duck, amphibian and aquatic mammals, contributed approximately 70% of the identified outlying predictions. These organisms are comparatively poorly studied and predictions were therefore often based upon modelling approaches or guidance methodology.

It should be noted that whilst most approaches have employed default values for this exercise they have the option of user defined (including site specific) data to be entered within assessments if this is available.

4.6. Conclusions

The comparison of predicted activity concentrations in a range of freshwater and terrestrial biota by models being used or developed for the protection of the environment demonstrated considerably more variability than the comparison of internal and external dose estimates described in Chapter 3. For the majority of radionuclides, Higley et al. [117] suggested that the most important predictor of biota dose is the method used to estimate activity concentrations in biota. In a discussion of data availability and available approaches to address the acknowledged data gaps, Beresford et al. [118] recommend a greater transparency

in methodology and data provenance than that currently available. The active participation of key models and approaches within this exercise is a useful step towards achieving this.

A number of the approaches participating in this exercise are under development. Therefore, transfer parameters they have used in this exercise should not be assumed to represent their final recommendations; subsequent chapters use further developments of some models transfer databases. Indeed an objective of the BWG is to improve the models as a consequence of participation (see Chapter 7 and also discussion of revision to DosDiMEco parameters in Chapter 6).

The exercise discussed in Chapters 3 and 4 enabled us to compare two basic elements of the approaches (transfer and dosimetric models). The subsequent chapters will compare predictions to available measurement data.

CHAPTER 5. FRESHWATER SCENARIO: PERCH LAKE

5.1. Scenario description

Perch Lake, located on Atomic Energy of Canada Limited (AECL)'s Chalk River Laboratories site, has received chronic, low-level inputs of ^{90}Sr, tritium, ^{60}Co and ^{137}Cs over a period of approximately 50 years. As a result, Perch Lake surface waters are routinely monitored as part of AECL's routine environmental monitoring program and the lake has been extensively studied historically, as well as in the recent past.

Fish, aquatic primary producers (including unrooted free-floating, floating-leafed, rooted submergent and emergent species), invertebrates, frogs, turtles and freshwater mammals were collected in Perch Lake in the mid- to late-1990s and subsequently analysed for radionuclides. Where possible, archived samples were processed and measured for radionuclide levels and historical data on radionuclides in water were compiled to depict temporal changes in ^{60}Co, ^{90}Sr, tritium and ^{137}Cs concentrations in the lake. A summary of the time-points for which radiological data are available for sediments and/or receptor biota in Perch Lake was compiled and, based on this summary, three time periods were selected by the EMRAS BWG for inclusion in the *Perch Lake Freshwater Scenario* (see Table 5.1): 1968–1971 (^{90}Sr, ^{60}Co and/or ^{137}Cs); 1994–1998 (^{90}Sr, ^{60}Co and/or ^{137}Cs); and 2003–2004 (from tritium).

The scenario description provided to participants can be found in Appendix IV, including a table of the required predictions for each data entry. The description provided data on activity concentrations in both water and sediments at the time points when biota were collected. A list of useful web-sites was also included within the scenario description to provide sources of biological information for the species being considered from which participants could extract any additional information they required.

To provide dose-rate data for this scenario, thermoluminescent dosimeters (TLDs) were deployed in both the sediments and the water of Perch Lake for a period of six weeks; however, the resultant dose rates did not significantly differ from control TLD readings. Therefore, it was not possible to provide data to validate predicted dose rates.

5.2. Application of models to the scenario

Scenario participants were asked to predict ^{90}Sr, ^{137}Cs, ^{60}Co and ^{3}H concentrations in a variety of receptor organisms and the corresponding unweighted dose rates they would have been expected to receive ([119], Appendix IV).

Eleven sets of predictions were provided for the Perch Lake freshwater scenario, including those from: AECL, D-Max, EA R&D128, ECOMOD, CASTEAUR, ERICA, EPIC DOSES3D, LAKECO-B, LIETDOS-BIO and RESRAD-BIOTA.

The specifics of the application of each approach to this scenario are described below; general model descriptions can be found in Chapter 2. A compilation of the CR values used and associated assumptions can be found in Table 5.2 for ^{60}Co, in Table 5.3 for ^{137}Cs, in Table 5.4 for ^{90}Sr and in Table 5.5 for tritium.

Table 5.1. Summary of receptor species and radionuclides for which predictions to be made as part of the Perch Lake Freshwater Scenario.

Receptor Common Name	1968–1971	1994–1997	2003–2004
Primary Producers:			
Free-floating (unrooted), submergent primary producers	^{90}Sr, ^{60}Co	^{90}Sr, ^{60}Co, ^{137}Cs	^{3}H
Rooted, submergent macrophytes	^{90}Sr, ^{60}Co	^{90}Sr, ^{60}Co, ^{137}Cs	No data
Rooted, floating-leafed macrophytes	^{90}Sr, ^{60}Co, ^{137}Cs	^{90}Sr, ^{60}Co, ^{137}Cs	No data
Emergent macrophytes	^{90}Sr, ^{60}Co	^{90}Sr, ^{60}Co, ^{137}Cs	^{3}H
Invertebrates:			
Zooplankton	No data	^{90}Sr, ^{60}Co, ^{137}Cs	^{3}H
Macroinvertebrates	^{90}Sr	^{90}Sr, ^{60}Co, ^{137}Cs	No data
Snails	^{90}Sr	^{90}Sr, ^{60}Co, ^{137}Cs	No data
Freshwater mussels	^{90}Sr	^{90}Sr, ^{60}Co, ^{137}Cs	^{3}H
Fish:			
Cyprinids	^{90}Sr, ^{60}Co	^{90}Sr, ^{60}Co	No data
Pumpkinseeds	^{90}Sr, ^{60}Co	^{90}Sr, ^{60}Co	No data
Brown bullheads	^{60}Co	^{90}Sr, ^{60}Co	^{3}H
Yellow perch	^{60}Co	No data	No data
Northern pike	Species not present	^{90}Sr, ^{60}Co	^{3}H
Amphibians:			
Green frogs	No data	^{90}Sr, ^{137}Cs, ^{60}Co	No data
Bullfrogs	^{90}Sr	^{90}Sr, ^{137}Cs, ^{60}Co	No data
Reptiles:			
Painted turtles	^{60}Co	^{90}Sr, ^{137}Cs, ^{60}Co	No data
Snapping turtles	^{60}Co	^{90}Sr, ^{137}Cs, ^{60}Co	No data
Aquatic Mammals:			
Star-nosed moles	No data	^{90}Sr, ^{137}Cs, ^{60}Co	No data
American water shrews	No data	^{90}Sr, ^{137}Cs, ^{60}Co	No data

5.2.1. RESRAD-BIOTA

In contrast to the other exercises in which RESRAD-BIOTA was run by the developing organisation, for this scenario, RESRAD-BIOTA was run by two organisations (neither of which had previous experience of the tool). The model application by these two organisations was based on intuitive use of the model and reference to the available guidance documentation.

Descriptions of the two model applications and results are referred to below as RESRAD-BIOTA (UK) and RESRAD-BIOTA (NRPA). Many of the RESRAD-BIOTA (UK) predictions were made using the model's allometric functionality whilst those for RESRAD-BIOTA (NRPA) were produced using the CR-based 'BiV approach'.

Table 5.2. Summary of ^{60}Co concentration ratios for Perch Lake receptor species.

Species	AECL (Mean ± SE)	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	CASTEAUR	LAKECO-B	LIETDOS-BIO	ERICA (NRPA)	RESRAD-BIOTA (UK)	RESRAD-BIOTA (NRPA)
	^{60}Co CR [Derivation]	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR
Free-floating (unrooted) Submergents	1900 ± 560 [Based on Perch Lake CRs measured in 1996]	15 000 [Maximum value]	1000 [literature review on vascular plants]	600 ± 400 [Equilibrium concentration ratio]	3200 [literature review on vascular plants]	32 [experimental for data phytoplankton]	200 [literature value]	3200 [ERICA value]	n.a.	2000 [code default value]*	n.a.
Rooted, Submergent Macrophytes	1800 ± 120 [Based on Perch Lake CRs measured in 1996]	15 000 [Maximum value]	1000 [[literature review on vascular plants]	500 ± 200 [Equilibrium concentration ratio]	3200 [literature review on vascular plants]	n.a.	200 [literature value]	500 [Ignalina site-specific value]	n.a.	2000 [code default value]	n.a.
Rooted, Floating-Leafed Macrophytes	470 ± 100 [Based on Perch Lake CRs measured in 1996]	15 000 [Maximum value]	1000 [literature review on vascular plants]	400 ± 200 [Equilibrium concentration ratio]	3200 [literature review on vascular plants]	n.a.	200 [literature value]	3200 [ERICA value]	n.a.	2000 [code default value]	n.a.
Emergent Macrophytes	230 ± 140 [Based on Perch Lake CRs measured in 1996]	15 000 [Maximum value]	1000 [literature review on vascular plants]	300 ± 200 [Equilibrium concentration ratio]	3200 [literature review on vascular plants]	n.a.	200 [literature value]	3200 [ERICA value]	n.a.	2000 [code default value]	n.a.
Zooplankton	n.a.	23 000 [Maximum value]	400 [literature review on zooplankton]	140 ± 60 [Equilibrium concentration ratio]	700 [literature review on zooplankton]	132 [experimental data for zooplankton]	570 [literature value]	n.a.	n.a.	2000 [code default value]	n.a.
Macroinvertebrates	n.a.	23 000 [Maximum value]	2000 [literature review on aquatic invertebrates]	100 (50–500) [Equilibrium concentration ratio]	10 000 [literature review on insect larvae]	100 [experimental data for *Gammarus*]	n.a.	n.a.	n.a.	2000 [code default value]	n.a.
Snails	n.a.	23 000 [Maximum value]	2000 [literature review on aquatic invertebrates]	30 ± 20 [Equilibrium concentration ratio]	3200 [literature review on gastropods]	100 [assumed same as for macroinvertebrates]	n.a.	n.a.	2900 [ERICA data run probabilistically (assuming lognormal distribution) – median CR values used in derivation.]	Allometric function used	2000 [code default value]

Species	AECL (Mean ± SE)	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	CASTEAUR	LAKECO-B	LIETDOS-BIO	ERICA (NRPA)	RESRAD-BIOTA (UK)	RESRAD-BIOTA (NRPA)
	^{60}Co CR [Derivation]	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR
Cyprinid species	81 ± 29 [Based on Perch Lake CRs measured in 1996]	650 [Maximum value; assumed to be omnivorous]	300 [literature review on benthic fish]	60 ± 40 [Equilibrium concentration ratio]	437 [literature review on benthic fish]	2 [experimental data for planktivorous fish with consideration of fish physiology]	9 [literature value]	437 [ERICA value]	130 [ERICA data run probabilistically (assuming lognormal distribution) – median CR values used in derivation.]	Allometric function used	n.a.
Pumpkinseed	110 ± 35 [Based on Perch Lake CRs measured in 1996]	650 [Maximum value; assumed to be omnivorous]	300 [literature review on benthic fish]	60 ± 40 [Equilibrium concentration ratio]	437 [literature review on benthic fishes]	2 [assumed same as for cyprinids]	9 [literature value]	437 [ERICA value]	130 [ERICA data run probabilistically (assuming lognormal distribution) – median CR values used in derivation.]	Allometric function used	2000 [code default value]
Brown Bullhead	55 ± 8.1 [Based on Perch Lake CRs measured in 1996]	650 [Maximum value; assumed to be omnivorous]	300 [literature review on benthic fish]	60 ± 40 [Equilibrium concentration ratio]	437 [literature review on benthic fishes]	6 [experimental data for benthivorous fish with consideration of fish physiology]	9 [literature value]	437 [ERICA value]	130 [ERICA data run probabilistically (assuming lognormal distribution) – median CR values used in derivation.]	Allometric function used	2000 [code default value]
Yellow Perch	3.5 ± 0.56 [Based on Perch Lake CRs measured in 1996]	650 [Maximum value]	300 [literature review on benthic fish]	60 ± 40 [Equilibrium concentration ratio]	437 [literature review on pelagic fishes]	4 [experimental data for piscivorous fish with consideration of fish physiology]	0.25 [literature value]	300 [site-specific value]	n.a.	Allometric function used	n.a.
Green Frog	n.a.	650 [Assumed same as predatory/ omnivorous fish]	5000 [Guidance derived value – assumed maximum available CR]	n.a.	140 [literature review on amphibians]	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.

Species	AECL (Mean ± SE)	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	CASTEAUR	LAKECO-B	LIETDOS-BIO	ERICA (NRPA)	RESRAD-BIOTA (UK)	RESRAD-BIOTA (NRPA)
	^{60}Co CR [Derivation]	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR	^{60}Co CR
Bullfrog	n.a.	650 [Assumed same as predatory/ omnivorous fish]	5000 [Guidance derived value – assumed maximum available CR]	n.a.	140 [literature review on amphibians]	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.
Painted Turtle	n.a.	650 [Assumed same as predatory/ omnivorous fish]	5000 [Guidance derived value – assumed maximum available CR]	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.
Common Snapping Turtle	n.a.	650 [Assumed same as predatory/ omnivorous fish]	5000 [Guidance derived value – assumed maximum available CR]	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.
Star-nosed Mole	n.a.	23 000 [Represents ratio of activity concentration in tissue relative to that in animal feed, where values represent the range in benthic invertebrates x 1.0 1.0 CR]	5000 [Guidance derived value – assumed maximum available CR]	n.a.	437	n.a.	n.a.	n.a.	700 [ERICA data run probabilistically (assuming exponential distribution) – median CR values used in derivation.]	Allometric function used	n.a.
American Water Shrew	n.a.	23 000 [Represents ratio of activity concentration in tissue relative to that in animal feed, where values represent the range in benthic invertebrates x 1.0 CR]	5000 [Guidance derived value – assumed maximum available CR]	n.a.	437 [assumes same value as for benthic/ pelagic fishes]	n.a.	n.a.	n.a.	700 [ERICA data run probabilistically (assuming exponential distribution) – median CR values used in derivation.]	Allometric function used	2000 [code default value]

n.a. – not applicable: prediction not made by this approach.
*See Section 5.2.1 for discussion of RESRAD-BIOTA default BiV values.

Table 5.3. Summary of ^{137}Cs for Perch Lake receptor species.

Species	AECL (Mean ± SE)	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	CASTEAUR	LAKECO-B	LIETDOS	ERICA (NRPA)	RESRAD-BIOTA (UK)	RESRAD-BIOTA (NRPA)
	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR
Free-floating (unrooted) Submergents	7400 ± 7200 [Based on Perch Lake CRs measured in 1996]	1500 [Maximum value]	2300 [literature review on vascular plants]	400 ± 200 [Equilibrium concentration ratio]	4700 [literature review on phytoplankton]	0.7 [experimental for phytoplankton]	250 [generated by LAKECO-B sub-model]	1600 [FASSET value]	n.a.	2000 [from BCF database see Section 5.2.1]	n.a.
Rooted, Submergent Macrophytes	230 ± 50 [Based on Perch Lake CRs measured in 1996]	1500 [Maximum value]	2300 [literature review on vascular plants]	200 ± 100 [Equilibrium concentration ratio]	1160 [literature review on vascular plants]	n.a.	250 [generated by LAKECO-B sub-model]	1000 [FASSET value]	n.a.	2000 [from BCF database see Section 5.2.1]	n.a.
Rooted, Floating-Leafed Macrophytes	670 ± 99 [Based on Perch Lake CRs measured in 1996]	1500 [Maximum value]	2300 [literature review on vascular plants]	100 ± 50 [Equilibrium concentration ratio]	1160 [literature review on vascular plants]	n.a.	250 [generated by LAKECO-B sub-model]	1600 [FASSET value]	n.a.	2000 [from BCF database see Section 5.2.1]	n.a.
Emergent Macrophytes	350 ± 99 [Based on Perch Lake CRs measured in 1996]	1500 [Maximum value]	2300 [literature review on vascular plants]	70 ± 30 [Equilibrium concentration ratio]	1160 [literature review on vascular plants]	n.a.	250 [generated by LAKECO-B sub-model]	1600 [FASSET value]	n.a.	2000 [from BCF database see Section 5.2.1]	n.a.
Zooplankton	n.a.	3370 [Maximum value]	19000 [literature review on zooplankton]	30 ± 20 [Equilibrium concentration ratio]	1560 [literature review on zooplankton]	30 [experimental for zooplankton]	710 [generated by LAKECO-B sub-model]	n.a.	n.a.	22 000 [code default value]*	n.a.
Macroinvertebrates	n.a.	3370 [Maximum value]	580 [literature review on aquatic invertebrates]	10 (5–80) [Equilibrium concentration ratio]	10 400 [insect larvae, which assumes crustacean value]	400 [experimental for *Gammarus*]	n.a.	n.a.	n.a.	22 000 [code default value]	n.a.
Freshwater Mussels	n.a.	n.a.	n.a.	50 ± 20 [Equilibrium concentration ratio]	n.a.	400 [assumed same as for macroinvertebrates]	n.a.	100 [FASSET value]	n.a.	Allometric function used	n.a.
Green Frog	n.a.	10 700 [Assumed same as predatory/ omnivorous fish]	11000 [Guidance derived value – assumed maximum available CR]	n.a.	9300 [literature review on amphibians]	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.

Species	AECL (Mean ± SE)	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	CASTEAUR	LAKECO-B	LIETDOS	ERICA (NRPA)	RESRAD-BIOTA (UK)	RESRAD-BIOTA (NRPA)
	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR	^{137}Cs CR
Bullfrog	n.a.	10 700 [Assumed same as predatory/ omnivorous fish]	11000 [Guidance derived value – assumed maximum available CR]	n.a.	9300 [literature review on amphibians]	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.
Painted Turtle	n.a.	10 700 [Assumed same as predatory/ omnivorous fish]	11000 [Guidance derived value – assumed maximum available CR]	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.
Common Snapping Turtle	n.a.	10 700 [Assumed same as predatory/ omnivorous fish]	11000 [Guidance derived value – assumed maximum available CR]	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.
Star-nosed Mole	n.a.	3370 [Represents ratio of activity concentration in tissue relative to that in animal feed, where values represent the range in benthic invertebrates x 1.0 (CR)]	11000 [Guidance derived value – assumed maximum available CR]	n.a.	9300 [literature review on amphibians]	n.a.	n.a.	n.a.	6700 [ERICA data run probabilistically (assuming exponential distribution) – median CR values used in derivation.]	Allometric function used	n.a.
American Water Shrew	n.a.	3370 [Represents ratio of activity concentration in tissue relative to that in animal feed, where values represent the range in benthic invertebrates x 1.0 (CR)]	11000 [Guidance derived value – assumed maximum available CR]	n.a.	9300 [literature review on amphibians]	n.a.	n.a.	n.a.	6700 [ERICA data run probabilistically (assuming exponential distribution) – median CR values used in derivation.]	Allometric function used	22 000 [code default value]

n.a. – not applicable: prediction not made by this approach.

*See Section 5.2.1 for discussion of RESRAD-BIOTA default BiV values.

Table 5.4. Summary of ^{90}Sr concentration ratios for Perch Lake receptor species.

Species	AECL (Mean ±SE)	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	CASTEAUR	LAKECO-B	LIETDOS-BIO	ERICA (NRPA)	RESRAD-BIOTA (UK)	RESRAD-BIOTA (NRPA)
	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR
Free-floating (unrooted) Submergents	140 ± 9.3 [Based on Perch Lake CRs measured in 1996]	220 Maximum value	1200 [literature review on vascular plants]	80 ± 50 [Equilibrium concentration ratios]	250 [literature review on vascular plants]	n.a.	100 [generated by LAKECO-B sub-model]	250 [ERICA value]	n.a.	600 [from BCF database see Section 5.2.1]	n.a.
Rooted, Submergent Macrophytes	120 ± 11 [Based on Perch Lake CRs measured in 1996]	220 Maximum value	1200 [literature review on vascular plants]	70 ± 40 [Equilibrium concentration ratios]	250 [literature review on vascular plants]	n.a.	100 [generated by LAKECO-B sub-model]	150 [FASSET value]	n.a.	600 [from BCF database see Section 5.2.1]	n.a.
Rooted, Floating-Leafed Macrophytes	330 ± 130 [Based on Perch Lake CRs measured in 1996]	220 Maximum value	1200 [literature review on vascular plants]	30 ± 20 [Equilibrium concentration ratios]	250 [literature review on vascular plants]	n.a.	100 [generated by LAKECO-B sub-model]	250 [ERICA value]	n.a.	600 [from BCF database see Section 5.2.1]	n.a.
Emergent Macrophytes	150 ± 38 [Based on Perch Lake CRs measured in 1996]	220 Maximum value; asumed same as aquatic macrophytes	1200 [literature review on vascular plants]	20 ± 10 [Equilibrium concentration ratios]	250 [literature review on vascular plants]	n.a.	100 [generated by LAKECO-B sub-model]	250 [ERICA value]	n.a.	600 [from BCF database see Section 5.2.1]	n.a.
Zooplankton	n.a.	1700 Maximum value	20 [literature review on zooplankton]	50 ± 30 [Equilibrium concentration ratios]	60 [literature review on zooplankton]	n.a.	290 [generated by LAKECO-B sub-model]	n.a.	n.a.	320 [code default value]*	n.a.
Macroinvertebrates	n.a.	1700 [Maximum value]	250 [literature review on aquatic invertebrates]	5–50 [Equilibrium concentration ratios]	200 [insect larvae, which assumes crustacean value]	n.a.	n.a.	n.a.	n.a.	320 [code default value]	n.a.
Freshwater Mussels	n.a.	1700 [Maximum value]	250 [literature review on aquatic invertebrates]	450 ± 250 [Equilibrium concentration ratios]	270 [bivalve mollusc]	n.a.	n.a.	320 [US-DOE Rad BCG calculator]	n.a.	Allometric function used	n.a.

Species	AECL (Mean ±SE)	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	CASTEAUR	LAKECO-B	LIETDOS-BIO	ERICA (NRPA)	RESRAD-BIOTA (UK)	RESRAD-BIOTA (NRPA)
	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR
Cyprinid Species	580 ± 150 [Based on Perch Lake CRs measured in 1996]	1000 [Maximum value]	43 [literature review on fish]	570 [Equilibrium concentration ratios estimated using the Ca concentration in water]	17 [literature review on benthic fishes]	n.a.	460 [generated by LAKECO-B sub-model]	17 [ERICA value]	10 [ERICA data run probabilistically (assuming lognormal distribution) – median CR values used in derivation.]	Allometric function used	n.a.
Pumpkinseed	830 ± 79 [Based on Perch Lake CRs measured in 1996]	1000 [Maximum value]	43 [literature review on fish]	570 [Equilibrium concentration ratios estimated using the Ca concentration in water]	17 [literature review on benthic fishes]	n.a.	460 [generated by LAKECO-B sub-model]	17 [ERICA value]	10 [ERICA data run probabilistically (assuming lognormal distribution) – median CR values used in derivation.]	Allometric function used	320 [code default value]
Brown Bullhead	780 ± 150 [Based on Perch Lake CRs measured in 1996]	1000 [Maximum value]	43 [literature review on fish]	310 ± 220 [Equilibrium concentration ratios accounting for competition between ^{90}Sr and Ca, where: $CR_{(Sr-90)}$ =2760/[Ca] in mg/L]	17 [literature review on benthic fishes]	n.a.	460 [generated by LAKECO-B sub-model]	17 [ERICA value]	10 [ERICA data run probabilistically (assuming lognormal distribution) – median CR values used in derivation]	Allometric function used	320 [code default value]
Green Frog	n.a.	1000 [Assumed same as predatory/ omnivorous fish]	1200 [Guidance derived value – assumed maximum available CR]	n.a.	17 [frog, which assumes same value as for benthic/ pelagic fishes]	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.

Species	AECL (Mean ±SE)	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	CASTEAUR	LAKECO-B	LIETDOS-BIO	ERICA (NRPA)	RESRAD-BIOTA (UK)	RESRAD-BIOTA (NRPA)
	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR	^{90}Sr CR
Bullfrog	n.a.	1000 [Assumed same as predatory/ omnivorous fish]	1200 [Guidance derived value – assumed maximum available CR]	n.a.	17 [frog, which assumes same value as for benthic/ pelagic fishes]	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.
Painted Turtle	n.a.	1000 [Assumed same as predatory/ omnivorous fish]	1200 [Guidance derived value – assumed maximum available CR]	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.
Common Snapping Turtle	n.a.	1000 [Assumed same as predatory/ omnivorous fish]	1200 [Guidance derived value – assumed maximum available CR]	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	Allometric function used	n.a.
Star-nosed Mole	n.a.	1700 [Represents ratio of activity concentration in tissue relative to that in animal feed, where values represent the range in benthic invertebrates x 1.0 CR]	1200 [Guidance derived value – assumed maximum available CR]	n.a.	17 [assumes same value as for benthic/ pelagic fishes]	n.a.	n.a.	n.a.	870 [ERICA data run probabilistically (assuming exponential distribution) – median CR values used in derivation]	Allometric function used	n.a.
American Water Shrew	n.a.	300–1700 [Represents ratio of activity concentration in tissue relative to that in animal feed, where values represent the range in benthic invertebrates x1.0 CR]	1200 [Guidance derived value – assumed maximum available CR]	n.a.	17 [assumes same value as for benthic/ pelagic fishes]	n.a.	n.a.	n.a.	870 [ERICA data run probabilistically (assuming exponential distribution) – median CR values used in derivation]	Allometric function used	320 [code default value]

n.a. – not applicable: prediction not made by this approach; *See Section 5.2.1 for discussion of RESRAD-BIOTA default BiV values.

Table 5.5. Summary of tritium CRs for Perch Lake receptor species.

Species	AECL	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	CASTEAUR	LAKECO-B	LIETDOS-BIO	ERICA (NRPA)	RESRAD-BIOTA (UK)	RESRAD-BIOTA (NRPA)
	Tritium CR	**Tritium CR**	**Tritium CR**	**Tritium CR**	**Tritium CR**	**Tritium CR**	**Tritium CR**	**Tritium CR**	**Tritium CR**	**Tritium CR**	**Tritium CR**
Free-floating (unrooted) Submergents	5200 ± 98 [(Cw/111)*120]	1 Maximum value	1 [specific activity model]	n.a.	1 [literature review on vascular plants]	n.a.	n.a.	0.94	n.a.	0.2 [code default value]*	n.a.
Emergent Macrophytes	5200 ± 98 [(Cw/111)*120]	1 Maximum value; assumed same as aquatic macrophytes	1 [specific activity model]	n.a.	1 [taken from EA R&D128**]	n.a.	n.a.	0.85	n.a.	0.2 [code default value]	n.a.
Zooplankton	5600 ± 110 [(Cw/111)*130]	1 Maximum value	1 [specific activity model]	n.a.	1 [taken from EA R&D128]	n.a.	n.a.	n.a.	n.a.	0.2 [code default value]	n.a.
Freshwater Mussels	5600 ± 110 [(Cw/111)*130]	1 Maximum value	1 [specific activity model]	n.a.	1 [taken from EA R&D128]	n.a.	not provided	1.05 [FASSET value]	n.a.	Allometric function used	n.a.
Brown Bullhead	5600 ± 110 [(Cw/111)*130]	1 Maximum value	1 [specific activity model]	n.a.	1 [literature review on benthic fish	n.a.	n.a.	0.96	1 [ERICA arithmetic mean selected]	Allometric function used	n.a.
Northern Pike	5600 ± 110 [(Cw/111)*130]	1 Maximum value	1 [specific activity model]	n.a.	1 [taken from EA R&D128]	n.a.	n.a.	0.96	1 [ERICA arithmetic mean selected]	Allometric function used	0.2 [code CR]

n.a. – not applicable: prediction not made by this approach.
*See Section 5.2.1 for discussion of RESRAD-BIOTA default BiV values.
**Copplestone et al. [6].

5.2.1.1. Assumptions made for RESRAD-BIOTA (UK) predictions

For the 1968-71 and 1994-97 time periods when activity concentrations were only available for the water, default sediment-to-water partition coefficients (K_d) of 0.001 l kg^{-1} (dw) were used to predict the tritium concentration in the sediments. For the 2003-4 time period, data were provided as organically-bound tritium and free-water tritium concentrations. Since RESRAD-BIOTA only permits one tritium value to be entered for sediment, the OBT and HTO values were summed to produce a total tritium activity in the sediment.

Water activity concentrations were provided in units of Bq l^{-1}, whereas units of Bq kg^{-1} were required in the RESRAD-BIOTA code; for the purposes of this assessment, the Bq l^{-1} data were treated as Bq kg^{-1} for input into the RESRAD-BIOTA code.

The 'New Organism' wizard in the RESRAD-BIOTA tool was applied to create specific biota corresponding to those found in Perch Lake. Each new organism was assigned the range of parameter values to allow the tool to perform the necessary calculations for a given receptor (Table 5.6). When particular organism-specific parameters could not be identified, the default settings for these parameters in RESRAD-BIOTA were retained. However, the indicative geometries for some receptor species did not always equate to the most appropriate geometry based on the dimensions of the organism that was being created. For example, Table 5.7 (taken from RESRAD-BIOTA documentation) indicated Geometry 2 as being appropriate for molluscs, but the measurement data for the Barnes mussel (*Elliptio complanata*) suggested that Geometry 3 is more appropriate (likely to be due to the relatively large range of body sizes that can be found for different species of molluscs). In such instances, the geometry was selected on an organism-specific basis.

To derive organism activity concentrations for aquatic primary producers, zooplankton and macroinvertebrates, BiVs were used. For all other organisms, the allometric approach was used.

RESRAD-BIOTA has an associated online BCF (again equivalent to CR) database (http://homer.ornl.gov/nuclearsafety/nsea/oepa/bdac/bcfs.html) and this was consulted to identify organism-specific values that may be more appropriate to use than the default BiVs. No organism-specific values were found for zooplankton or macroinvertebrates for the four radionuclides included within the Perch Lake scenario, so these organisms were run using the default aquatic organism BiVs from RESRAD-BIOTA. For aquatic plants, there were CRs for ^{137}Cs and ^{90}Sr of 2000 and 600 respectively, so these were used instead of the default values. Note the default BiVs provided in the RESRAD-BIOTA model are for screening purposes only and are not meant to represent any specific species; for the aquatic environment, default values are not provided for plants.

Before the assessment was run, the 'external exposure geometry factor' of each organism was modified to reflect the relationship between the organism and the surrounding media (Table 5.6). A factor of 0.5 indicates a 2π geometry and a factor of 1 indicates a 4π geometry.

Table 5.6. Parameters used to create new organisms for Perch Lake Scenario for the allometric approach, which was carried out as part of the RESRAD-BIOTA (UK) Model run.

Organism Group	Common name	Species	X-axis (cm)	Y-axis (cm)	Z-axis (cm)	RESRAD-BIOTA geometry equivalent	Mass (kg)	Life span (yr)	Ingestion pathway		Geometry Factor	
									Consumes food from sediment?	Consumes food from water?	Sediment	Water
Aquatic Primary Producers												
Free-floating (unrooted) submergents						1					0.5	1
Rooted, submergent macrophytes						1					1	1
Rooted, floating-leafed macrophytes						1					1	1
Emergent macrophytes						1					1	0.5
Aquatic Invertebrates												
Zooplankton						1					0.5	1
Macroinvertebrates						1					1	0.5
Snails	Represented by Amnicola spp	*Amnicola limosa*	3	2	2	2	0.004	1	N	Y	0.5	0.5
Freshwater mussels	Barnes Mussel	*Elliptio complanata*	10	6	3	3	0.015	60	Y	Y	0.5	1
Fish												
Forage Fish	Cyprinid spp (represented by Lake chub)	*Couesius plumbeus*	30	7	7	4	3.5	5	Y	Y	0.5	1
	Pumpkinseeds	*Lepomis gibbosus*	11	3	5	3	0.48	6	Y	Y	0.5	1
Benthivorous Fish	Brown Bullhead	*Ameiurus nebulosis*	48	8	11	4	2.5	8	Y	Y	0.5	1
Piscivorous fish	Yellow Perch	*Perca flavescens*	18	3	5	3	0.22	8	Y	Y	0.5	1
	Northern Pike	*Esox lucius*	60	7	9	4	1.6	10	N	Y	0.5	1
Amphibians	Green frogs	*Rana clamitans*	7.5	3	3	3	0.02	10	Y	Y	0.5	1
	Bullfrogs	*Rana catesbeiana*	12	6	5	3	0.225	5	Y	Y	0.5	1
Reptiles	Painted Turtle	*Chrysemys picta*	16	11	8	4	0.057	35	N	Y	0.5	1
	Common Snapping Turtle	*Chelydra serpentina*	28	12	19	4	10	30	N	Y	0.5	1
Aquatic Mammals	Star Nose Mole	*Condylura cristata*	15	7	6	3	0.055	4	Y	Y	0.5	1
	American Water Shrew	*Sorex palustris*	15	6	5	3	0.013	1.5	Y	Y	0.5	1

Table 5.7. Geometries available within RESRAD-BIOTA (from tool help function).

Model Geometry No.	Mass Category (Kg)	Example Receptors	References	Specific Geometry Dimensions Applied (cm)	Specific Mass Applied (Kg)
1	1.00E-05	Fish egg* Fish (larvae) Plant root (meristem) Plant seed Plant shoot (meristem)	Thorp, J.H. and A.P. Covich, 1991. Ecology and Classification of North American Freshwater Invertebrates. Academic Press, Inc., San Diego, CA. 911 pp.	0.2 x 0.2 x 0.2	4.20E-06
2	1.00E-03	Fish (young-of-year) Molluscs* Plant seedling Tadpoles	IAEA (1988), NCRP(1991), UK R&D Publication 128, and INFO-0730	2.5 x 1.2 x 0.62	1.00E-03
3	1.00E-02	Fathead minnow Frogs Hispid cotton rat Sculpins Shrews Voles White-footed Mouse*	Patton et al. (2001), UK R&D Publication 128, and DOE-STD-1153-2002	10 x 2 x 2	2.10E-02
4	1	Black bass Large fish* Suckers	IAEA (1988), UK R&D Publication 128, and INFO-0730	45 x 8.7 x 4.9	1
5	1.00E+01	Beaver Carp Catfish (Channel and Coyote Fox (red or grey) Raccoon* Striped bass	DOE-STD-1153-2002	50 x 26 x 13	8.8
6	1.00E+02	Mule deer White-tailed deer*	Mule and Black-tailed Deer of North America (1981)	100 x 42 x 33	72.6
7	5.00E+02	Elk*	Wild Mammals of North America, 1982. J.A. Chapman and G.A. Feldhamer, editors. Johns Hopkins University Press, Baltimore.	270 x 66 x 48	447.9
8	1.00E+03	Grizzly bear*	Wild Mammals of North America, 1982. J.A. Chapman and G.A. Feldhamer, editors. Johns Hopkins University Press, Baltimore.	220 x 100 x 100	1150

Table 5.8. Assumptions that were made for the RESRAD-BIOTA (NRPA) application.

Reference Organism	RESRAD-BIOTA geometry assumed*	Fraction of time assumed to be in Perch Lake	Fraction of time assumed to be in the water column	Fraction of time assumed to be at the water–sediment interface
Snail	2	1	0	1
Water shrew	3	0.4	0.2	0.2
Brown bullheads	4	1	0.3	0.7
Pumpkinseed	4	1	0.5	0.5
Northern pike	4	1	0.7	0.3

*See Table 5.7.

5.2.1.2. Assumption made for RESRAD-BIOTA (NRPA) predictions

The participants applied the default BiV values from the tool; the assumptions that were applied with regard to geometry and occupancy factors can be found in Table 5.8. Where organisms were assumed to spend time in both the water column and at the sediment-water interface, to estimate total external dose rates the model was run twice: (i) firstly for the water column assuming exposure geometry factors[7] of 0 for sediment and 1 for water; (ii) secondly for the sediment water interface assuming geometry factors of 0.5 for both water and sediment. The results were then summed.

5.2.2. ERICA

Predicted whole-body activity concentrations were estimated using the default CR values for the most appropriate reference organisms; the CR database version, as described by Hosseini et al. [24] was used. Where more than one CR value may have been applicable (e.g. for free-floating (unrooted) submergent primary producers the CR value for either vascular plant or phytoplankton may have been appropriate), the highest value was selected. This was compatible with the guidance provided for selecting CR values for use in the ERICA Tool [23]. Occupancy factors and appropriate geometries were identified from the websites provided within the scenario description. Results from this ERICA application are referred to as ERICA (CEH) in the remainder of this chapter.

5.2.3. LIETDOS-BIO

The aquatic ecosystem in the LIETDOS-BIO model has been sub-divided into two compartments, representing the bed sediments and the water column. The time-dependent behaviour of a nuclide in each of these compartments is described by first-order differential equations. Lake water was considered to consist of a single, well-mixed compartment, where it was assumed that the concentration of a given nuclide dissolved in the lake water and absorbed onto suspended solids were in equilibrium. In addition, it was assumed that the nuclide absorbed onto suspended solids would not to be taken up by biota.

[7] Parameter within RESRAD-BIOTA model.

Table 5.9. Organisms for which predictions have been made using the DOSE3D model and their assumed mass, dimensions and occupancy factors.

Common name	Scientific name	Mass (g)	Dimensions (cm)	Occupancy Factor (in water column)	Occupancy Factor (at water/sediment interface)
Creek chub	*Semotilus atromaculatus*	340	25×6×4	0.3	0.7
Pumpkinseed	*Lepomis gibbosus*	230	18×9×3	0.5	0.5
Brown bullhead	*Ameirus nebulosis*	1500	26×18×6	0.3	0.7
Yellow perch	*Perca falvescens*	235	21×11×6	0.5	0.5
Northern Pike	*Esox lucius*	2100	61×9×8	0.7	0.3
Star-nosed mole	*Condylura cristata*	60	9×4×3	0.2	0.2
American water shrew	*Sorex palustris*	16	6×2×2	0.2	0.2
Snail	*Amnicola spp.*	0.034	0.4×0.4×0.4	0	1

5.2.4. EPIC-DOSES3D

To estimate whole-body activity concentrations, CRs were selected from the underlying databases for freshwater environments from the ERICA Tool. These were subsequently used as inputs to the EPIC DOSES3D model to determine absorbed dose rates. However, the group applying this approach did not use the default (arithmetic mean) ERICA CR. Instead, they chose to determine median values from the ERICA database mean and standard deviation values assuming a log-normal distribution. In cases where no standard deviation was available within the ERICA database, the underlying distribution was assumed to be exponential. The calculations were performed based on 10 000 simulations utilising a Monte Carlo code. In the comparisons of predicted activity concentration in the subsequent text, outputs by this approach are referred to as ERICA (NRPA).

The computer code, DOSES3D, was employed to derive internal and external DCCs. In doing so, geometries for Perch Lake organisms were defined following a three-step process: first, the length of the animal of interest was defined through consultation of the scientific literature; second, a representative picture of the animal was selected on which the animal width could be measured; and third, the relationship between mass, volume and density was applied to estimate the third dimensions[8]. A list of the organisms for which DCC calculations have been carried out can be found in Table 5.10 along with the masses, derived dimensions and the assumed occupancy factors for each organism type.

5.2.5. D-Max

Relationships between fish-water *CR* for ^{90}Sr, and calcium in the surrounding water ([Ca] (mg l^{-1})), based on measurements made by Vanderploeg et al. [120] and quoted in Blaylock [81] have been determined:

$$CR_{(muscle)} = \exp(5.2\text{-}1.2 \ln[Ca]) \quad (5.1)$$
$$CR_{(bone)} = \exp(9.7\text{-}1.2 \ln[Ca])$$

Assuming that 20% of the wet weight of a fish is composed of bony parts (Ryabov, I.I., Severtsov Institute, Moscow, *pers. comm.*), this gives the whole-fish CR was used in this exercise:

[8] Note – this approach was used by most of the participants to derive organism specific geometries.

$$CR_{(\text{whole fish})} = \exp(9.13\text{-}1.2 \ln[Ca]) \quad (5.2)$$

The uncertainty range is estimated to be from 0.33 to 3 times the best estimate value (based on the variation of measured values) [81, 84].

5.2.6. ECOMOD

Equilibrium CRs for ^{60}Co were estimated on the basis of data collected from controlled aquarium experiments that were performed at the Ural Institute of Biology for different aquatic species by e.g. [121].

Equilibrium CRs for ^{90}Sr in aquatic plants, zooplankton and mollusc were evaluated on the basis of the data obtained in Latvia between 1971 and 1974 from observations of migration of this radionuclide in 6 lakes [122]. In addition, data collected from the Urals and from Chernobyl were also used [123–125].

An empirical formula was derived to estimate the equilibrium concentration ratio of ^{90}Sr in fish as a inverse relationship to water concentration Ca:

$$CR(^{90}Sr) = \frac{3940(1770 - 6110)}{[Ca^{2+}]} \quad (5.3)$$

where $[Ca^{2+}]$ is concentration of calcium (Ca^{2+}) in lake water (in mg l^{-1}).

The relationship between the equilibrium concentration ratio of ^{90}Sr in fish and the concentration of Ca in water is based upon data collected and analyzed by Kryshev [126]. The dataset contains 115 values of the concentration factors at different concentrations of Ca. The data included considered were based primarily on publications in the Russian literature and relate to the whole body concentration factors.

Equilibrium CRs for ^{137}Cs in aquatic plants, zooplankton and molluscs were evaluated on the basis of Chernobyl data [125].

Estimates of mass, size and growth rate of the species of fish were found on the internet. The methodology that was applied to evaluate the general metabolic rate of fish (W), assuming a balanced energy equality in relation to fish mass, growth rate and water temperature, have been previously described [55–58]. The ε_A parameter value for ^{137}Cs and ^{90}Sr was evaluated by Kryshev [56, 57], whereas for ^{60}Co, the value of this parameter was given as a first approximation.

Table 5.10. Parameter values for calculations of dynamic CRs using the ECOMOD semi-empirical model (see Section 2.10).

Species	M* (g)	μ**, year^{-1}	ε_A***, dimensionless
Blacknose shiner	20	0.5	
Pearl dace	60	0.4	^{137}Cs 0.3±0.1
Brown bullhead	100	0.3	^{90}Sr 0.04±0.02
Pumpkinseeds	300	0.2	^{60}Co 0.2±0.1
Yellow perch	500	0.2	
Pike	1000	0.2	

*Annual average fish mass; **annual average rate of increase in fish mass; ***radionuclide-specific coefficient of proportionality between the rate of bioelimination of radionuclide from fish and rate of metabolism.

5.2.7. CASTEAUR-EDEN

As the basic data provided within the Perch Lake scenario were mean annual concentrations in water, the dynamic approach used in the CASTEAUR model, was simplified in an equilibrium model. To estimate radionuclide activity concentrations in Perch Lake biota, CRs were then extracted from CASTEAUR as aggregated data, whose values depend on the code parameterisation. These CRs are combinations of ecological data, such as feeding rates, that are default values defined by a bibliographical research, and of radioecological data (kinetic parameters of radionuclide transfer). The kinetic parameters were derived from controlled laboratory experiments. Activity concentrations in biota were then obtained by the multiplication of the water concentrations by the derived CRs.

To provide absorbed dose rate estimates for Perch Lake biota, the EDEN dosimetric tool was applied following the options described below:

- Gamma DCCs were evaluated applying Monte Carlo calculation, considering a threshold organism approach for phytoplankton, *Daphnia* and *Gammarus*;
- Beta and alpha DCCs were evaluated with a local deposit hypothesis for all the other organisms (for one organism per "trophic" level, a Monte Carlo approach was also applied for comparison).

Organism geometries were derived from a literature review extended from the web-site list provided within the scenario. The location of each organism was defined having considered its usual habitat. Composition and densities were taken from the FASSET framework [26], except for plants and sediments. Aquatic plants were assumed to be the same as to water in terms of composition and density.

5.2.8. LAKECO-B

For plants, the CR values for ^{60}Co were derived from Coughtrey and Thorne [128], those for ^{90}Sr and ^{137}Cs were calculated by submodels within LAKECO-B, driven by calcium and potassium. For zooplankton, prey fish (non-piscivorous fish) and predatory fish (piscivorous fish) CRs are not used in the model, but calculated on the basis of the predicted concentrations.

In terms of the estimation of doses to Perch Lake biota, with the exception of benthic fish (i.e. brown bullhead), the external dose from sediments was assumed to be zero. In the case of ^{90}Sr, the dose through ingestion was the only exposure pathway that was considered; external dose from water and sediments were assumed to be negligible.

5.2.9. AECL

The objective of the AECL model run was to determine whether it is feasible to predict concentrations of key radionuclides in Perch Lake biota based on measurements taken in the water as part of AECL's routine radiological monitoring program. This involved tabulation of radionuclide transfer factors based on a subset of the 1996 values (a year during which an extensive sampling campaign of the lake was undertaken), and back-calculation to estimate radionuclide levels in biota in other years, as well as in the remaining 1996 samples.

5.2.10. EA R&D128

The DCCs and CR values applied were those contained within the freshwater (v1.15) and terrestrial (v1.20) spreadsheets released in 2003 [6].

The default CR values were derived from literature review (with a bias towards data collected in the UK) and using a guidance-derived approach to fill in gaps where no CRs exist for particular biota/radionuclide combinations. The guidance used to fill in the data gaps is described in Copplestone et al. [6].

5.3. Statistical methods

5.3.1. Activity Concentrations

Z-scores have been derived for each prediction by reference to the observed data and associated standard deviations, i.e.:

$$Z = \frac{Predicted\ activity\ concentration - Observed\ mean\ activity\ concentration}{Observed\ SD} \tag{5.4}$$

Statistical interpretation of inter-model results was performed on the basis of analyzing Z-scores for individual organisms, and additionally for groups or categories of organisms (including primary producers, invertebrates, fish, amphibians, reptiles and freshwater mammals), the latter of which is particularly useful for detecting broad data trends. The efficacy of predictions by organism and group has also been estimated as the percentage of Z-scores below a value of 3.

5.3.2. Dose Rates

The combined internal and combined external absorbed dose rates were analysed using the same approach as described in Chapter 3. A 'robust' mean and standard deviation were generated for each comparison by the removal of outlying predictions. Z-scores were then estimated using the robust mean and standard deviation as the reference value. The predictions of the D-Max model were not considered in this analysis, as its output is not comparable to that of the other participating models.

5.4. Results and Discussion

For subsequent presentation, predicted activity concentrations have been normalised to the observed data for each prediction. The range in observed values presented was estimated by normalising the highest and lowest values observed over the whole period considered to the overall mean for a given species-radionuclide combination.

5.4.1. Activity Concentrations

5.4.1.1. Modelled-to-Measured Comparisons for ^{3}H

Tritium activity concentrations were predicted by AECL, ERICA (CEH), EA R&D128, ERICA (NRPA), RESRAD-BIOTA (UK), RESRAD-BIOTA (NRPA), LAKECO-B and D-Max based on data that had been collected between 2003 and 2004 for zooplankton, freshwater mussels, emergent macrophytes, submergent macrophytes, brown bullheads and

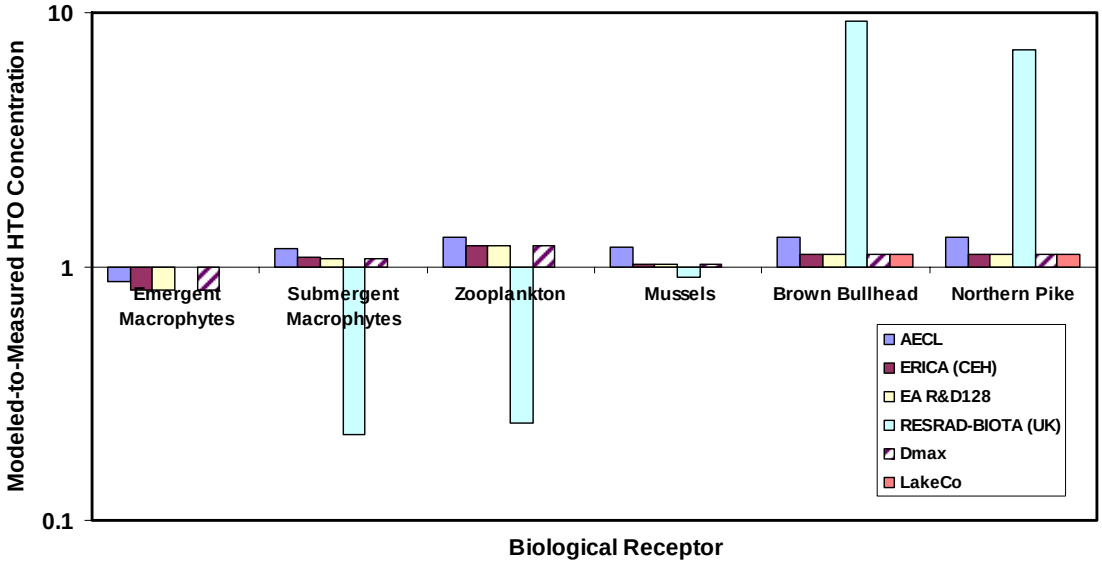

Fig. 5.1. Comparison of modelled-to-measured tritium (HTO) concentrations in Perch Lake freshwater species.

northern pike (Figure 5.1). In general, for the receptors considered, all modelled values fell within less than 1.3-fold of measured values, indicating that the various approaches are able to generate reasonable predictions of tritium (HTO) for a variety of freshwater biota species. Exceptions were predictions made by the two groups applying RESRAD-BIOTA. RESRAD-BIOTA (UK) over-estimated the tritium levels in fish by approximately 10-fold using an allometric approach, whereas RESRAD-BIOTA (NRPA) under-predicted the fish tritium activity concentration by approximately 5-fold following the application of a default CR of 0.2 for tritium. By comparison, other models typically applied a CR of approximately unity for tritium in aquatic biota, including fishes. RESRAD-BIOTA (UK) predictions for zooplankton and submergent macrophytes were also considerably lower than those generated by other models.

There were insufficient data to enable statistical evaluation of ^{3}H predictions.

5.4.1.2. Modelled-to-Measured Comparisons for Aquatic Primary Producers

Cobalt-60

Cobalt-60 concentrations were predicted in Perch Lake primary producers by AECL, ERICA (CEH), EA R&D128, CASTEAUR, D-Max, ECOMOD, LAKECO-B, LIETDOS-BIO and RESRAD-BIOTA (UK). Of these, CASTEAUR was only used to predict ^{60}Co levels in free-floating primary producers, whereas the AECL approach was applied to make predictions for all types of primary producers with the exception of emergent macrophytes.

Comparison of model predictions with measured data for emergent macrophytes revealed that with the exception of ECOMOD (which predicted within 3.6-fold of measured values), ^{60}Co levels in this type of macrophyte tended to be over-predicted by approximately 1- to 2-orders of magnitude (Figure 5.2). Such trends may be due to the relative importance of ^{60}Co uptake

via the water pathway in species that are submerged in the water compared to emergent species that are not; submergent species are known to absorb ions via pores, known as hydropoten, on the leaves, in addition to from roots (if present) [129]. Since in many cases, the same CR is being applied for all types of primary producers (e.g. Table 5.2), potential differences in radionuclide uptake pathways is not necessarily accounted for by many of the modelling approaches. In addition, emergent species were collected along the edges of the lake in areas where sediment radionuclide concentrations were relatively lower than those found in the lake itself. In the case of emergent macrophytes, which often have a relatively large proportion of their biomass in the air outside of the water column, the sediment pathway may be particularly important as a route of ^{60}Co uptake, potentially making sediment concentrations a key consideration. Since the CRs being applied are often being presented with respect to concentrations in the water (which is assumed to be the reference phase), the potential importance of the sediments in radionuclide transfer may be missed.

Similarly, ^{60}Co levels in floating-leafed macrophytes (such as water lilies and comparable species, which have upper leave surfaces open to the air) also tended to be over-predicted relative to values that had been measured in Perch Lake, but to a lesser extent than for emergent plants. For example, of the eight models that participated in the prediction of this type of primary producer, four models, including ERICA (CEH), LIETDOS-BIO, EA R&D128, D-Max and RESRAD-BIOTA (UK), produced predictions that fell outside of the measured range in Perch Lake.

By comparison, with the exception of the ERICA (CEH), LIETDOS-BIO and D-Max approaches (each of which tended to over-predict ^{60}Co in all types of Perch Lake primary producers) and the CASTEAUR model (which produced a 9-fold under-prediction), all other models produced predictions for free-floating primary producers that fell within the range found in the lake. The under-prediction by the CASTEAUR model may be due to the application of a CR derived for phytoplankton in the laboratory under conditions representative of French rivers to estimate ^{60}Co activity concentrations in Perch Lake free-floating primary producers.

In the case of submergent macrophytes, again, values that had been predicted using ERICA (CEH) and D-Max exceeded the range of values found in Perch Lake, although, two models (ECOMOD and LAKECO-B) under-predicted ^{60}Co concentrations in submergent species. Radionuclide uptake from the water, in addition to the sediments, can be enhanced for submergent species that are entirely immersed in the water and that often have leaves with relatively large surface areas [127, 133].

Statistical interpretation of model results was performed on the basis of analyzing Z-scores for individual organisms, as well as for groups or categories of organisms, as described in above. The results (Table 5.11) basically reinforce the observation made above. Z-scores showed efficacies of 67% for free-floating primary producers, of 30% for floating-leafed macrophytes and of 43% for submergent macrophytes, respectively, with an overall efficacy of 43% for predictions of ^{60}Co activity concentrations in aquatic primary producers as a group.

For application of the D-Max model a maximum CR value of 15000 l kg^{-1} (fw) for aquatic primary producers was assumed (from its database). As may have been anticipated, this leads to an over-estimation of ^{60}Co activity concentrations for all the types of primary producers that were considered as part of this scenario. Similarly conservative, screening-level CRs have also been selected for use in EA R&D128, with a number of over-predictions also occurring for the ERICA (CEH), LIETDOS-BIO and RESRAD-BIOTA (UK) approaches.

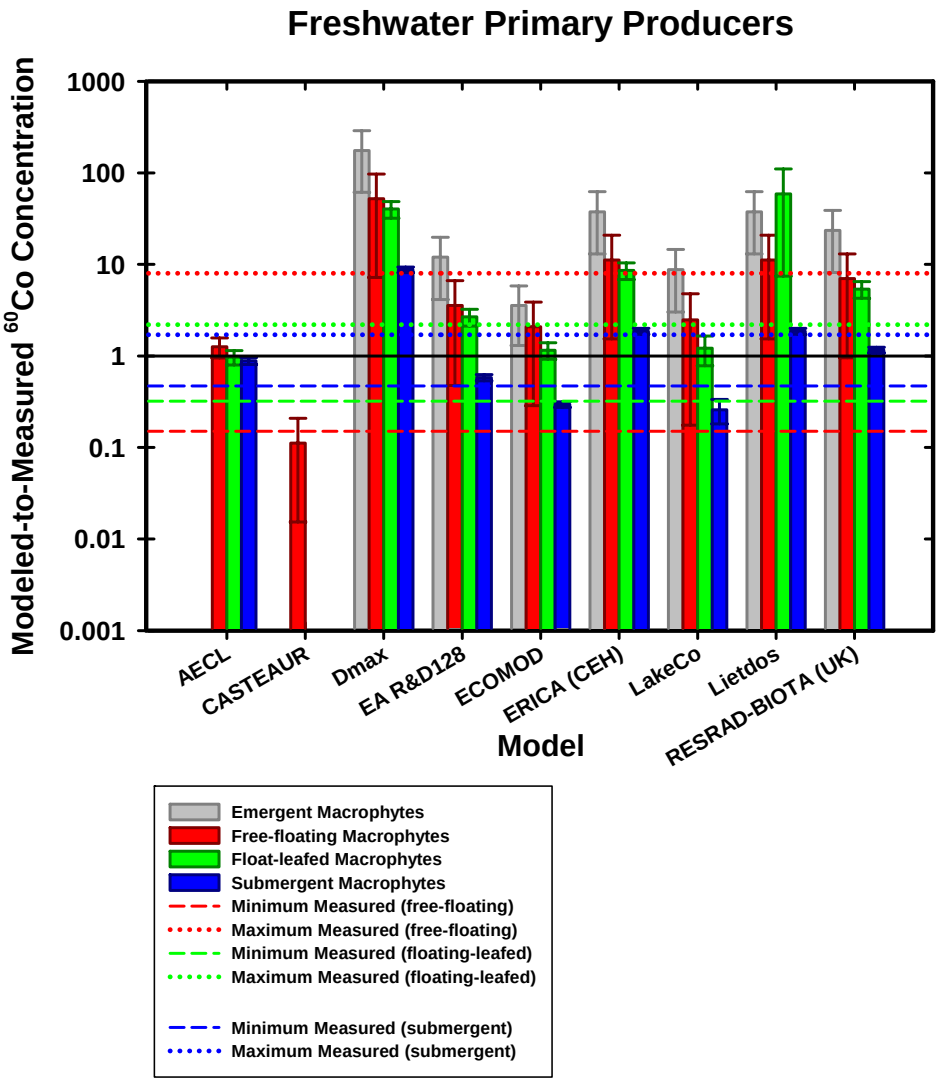

Fig. 5.2. Comparison of modelled-to-measured ^{60}Co concentrations in Perch Lake freshwater primary producers. Dashed and dotted horizontal lines represent minimum and maximum measured values in the lake for a given type of primary producer. Note that inadequate data were available to compare the modelled-to-measured range to minimum and maximum values for emergent macrophytes. Error bars represent the standard error in predicted values for a given species of primary producer by a given model.

Caesium-137

Caesium-137 activity concentrations were predicted in Perch Lake primary producers by: AECL, CASTEAUR, D-Max, EA R&D128, ECOMOD, ERICA (CEH), LAKECO-B, LIETDOS-BIO and RESRAD-BIOTA (UK). CASTEAUR was only used to predict ^{137}Cs levels in free-floating primary producers, whereas all other approaches were applied to make predictions for all types of the freshwater primary producers considered.

Table 5.11. Summary of Z-scores for the prediction of ^{60}Co activity concentrations in different types of freshwater primary producers. Values in shaded cells represent Z-scores that are greater than 3.

Type of Primary Producer	Year	Prediction No.	Z-scores for ^{60}Co Concentration Data (by Organism Type)										
			AECL	CASTEAUR	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	ERICA (NRPA)	LAKECO-B	LIETDOS-BIO	RESRAD-BIOTA (NRPA)	RESRAD-BIOTA (UK)
Emergent (rooted)	1994	PL33	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Emergent (rooted)	1995	PL42	n.a.	n.a.	39.3	1.3	0.5	7.2	n.a.	n.a.	7.2	n.a.	4.0
Emergent (rooted)	1996	PL48	n.a.	n.a.	117	6.8	1.3	24.0	n.a.	n.a.	24.0	n.a.	14.5
Free-floating (unrooted)	1968	PL1	1.5	1.6	17.3	0.4	0.9	2.4	n.a.	n.a.	2.4	n.a.	0.8
Free-floating (unrooted)	1969	PL6	0.2	1.5	7.5	0.9	1.2	0.4	n.a.	n.a.	0.4	n.a.	0.3
Free-floating (unrooted)	1994	PL31	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Free-floating (unrooted)	1995	PL39	0.7	3.8	13.3	2.7	3.2	0.2	n.a.	n.a.	0.2	n.a.	1.6
Floating-leafed (rooted)	1968	PL3	0.4	n.a.	26.2	0.6	n.a.	4.6	n.a.	n.a.	4.6	n.a.	2.4
Floating-leafed (rooted)	1969	PL8	0.2	n.a.	238	11.0	1.8	47.1	n.a.	n.a.	47.1	n.a.	27.6
Floating-leafed (rooted)	1970	PL14	0.3	n.a.	779	37.6	7.0	155	n.a.	n.a.	155	n.a.	91.4
Floating-leafed (rooted)	1994	PL32	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Floating-leafed (rooted)	1995	PL41	0.5	n.a.	30.7	0.9	0.3	5.6	n.a.	n.a.	5.6	n.a.	3.0
Floating-leafed (rooted)	1996	PL47	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Submergent (rooted)	1968	PL2	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Submergent (rooted)	1969	PL7	33.5	n.a.	625	55.1	78.0	53.0	n.a.	n.a.	53.0	n.a.	5.2
Submergent (rooted)	1970	PL13	0.3	n.a.	30.3	1.9	3.0	3.2	n.a.	n.a.	3.2	n.a.	0.5
Submergent (rooted)	1995	PL40	0.2	n.a.	5.5	0.4	0.6	0.5	n.a.	n.a.	0.5	n.a.	0.05
Submergent (rooted)	1996	PL46	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.

Table 5.12. Comparison of the efficacy of model predictions of ^{60}Co activity concentrations for different types of aquatic primary producers.

Radionuclide	Type of Freshwater Primary Producer	Number of predictions Z>3	Total n	[a] Efficacy Measure (%)
^{60}Co	Emergent (rooted)	9	12	25%
^{60}Co	Free-floating (unrooted)	8	24	67%
^{60}Co	Floating-leafed (rooted)	19	27	30%
^{60}Co	Submergent (rooted)	12	21	43%
^{60}Co	**ALL Primary Producers**	**48**	**84**	**43%**

[a] The Efficacy Measure represents the percentage of model predictions with Z-scores that fall below a value of 3 (which represents an under- or over-prediction relative to the predictions generated by all models).

As for ^{60}Co, it was not possible to determine whether modelled ^{137}Cs values fell within the range that could be found in Perch Lake for emergent macrophytes due to the relatively small amount of data that were available for this type of primary producer. However, all approaches produced predictions for ^{137}Cs in emergent macrophytes that fell within 10-fold of measured values. That said, it is interesting to note that the prediction by AECL was relatively lower than those of the other approaches (with a model-to-measured ratio of 0.11). By comparison, predictions produced by RESRAD-BIOTA (UK) and EA R&D128 fell at the higher end of the range of predicted values with modelled-to-measured ratios of 7.4 and 8.7 respectively (Figure 5.3).

For other types of freshwater primary producers, it was possible to evaluate whether model predictions fell within the range of values that had been measured in the lake. In the case of free-floating primary producers, comparison of modelled values with those measured in primary producers in the lake revealed a number of models that under-predicted ^{137}Cs activity concentrations for this type of organism. These included AECL, CASTEAUR and ECOMOD. In terms of the CASTEAUR approach, it is possible that this may have been due to the application of ^{137}Cs CRs that originated from laboratory experiments, conducted for phytoplankton under conditions representative of French rivers which are not applicable to the physico-chemical environment of Perch Lake. ERICA (CEH) slightly over-predicted ^{137}Cs levels in free-floating macrophytes. In general, the CRs that have been applied using the ERICA (CEH) tool represent a compilation of literature values, which may not reflect the conditions that are found in Perch Lake. In addition, the ERICA (CEH) CR applied a single CR value of 3200 l kg^{-1} (fw), regardless of the type of macrophyte being considered, and a relatively large amount of variability is known to occur for different types of freshwater primary producers [134].

By comparison, ^{137}Cs in submergent macrophytes, which varied by an order of magnitude in Perch Lake, were over-predicted by six of the eight approaches that were used to predict for this radionuclide-organism combination. These included D-Max, EA R&D128, ERICA (CEH), LAKECO-B, LIETDOS-BIO and RESRAD-BIOTA (UK) (Figure 5.3). Again, it is likely that the source of the ^{137}Cs CR values for each model, in addition to how CR values are being selected, drives these differences in modelled-to-measured-values.

In all cases, model predictions for floating-leafed primary producers fell within the range that is found in Perch Lake (Figure 5.3).

Z-scores were for individual models and estimates of the overall efficacy of the predicted values are presented in Tables 5.13 and 5.14, respectively. With the exception of emergent macrophytes (which had an efficacy of 100%), efficacy measures were relatively low for ^{137}Cs in most types freshwater primary producers.

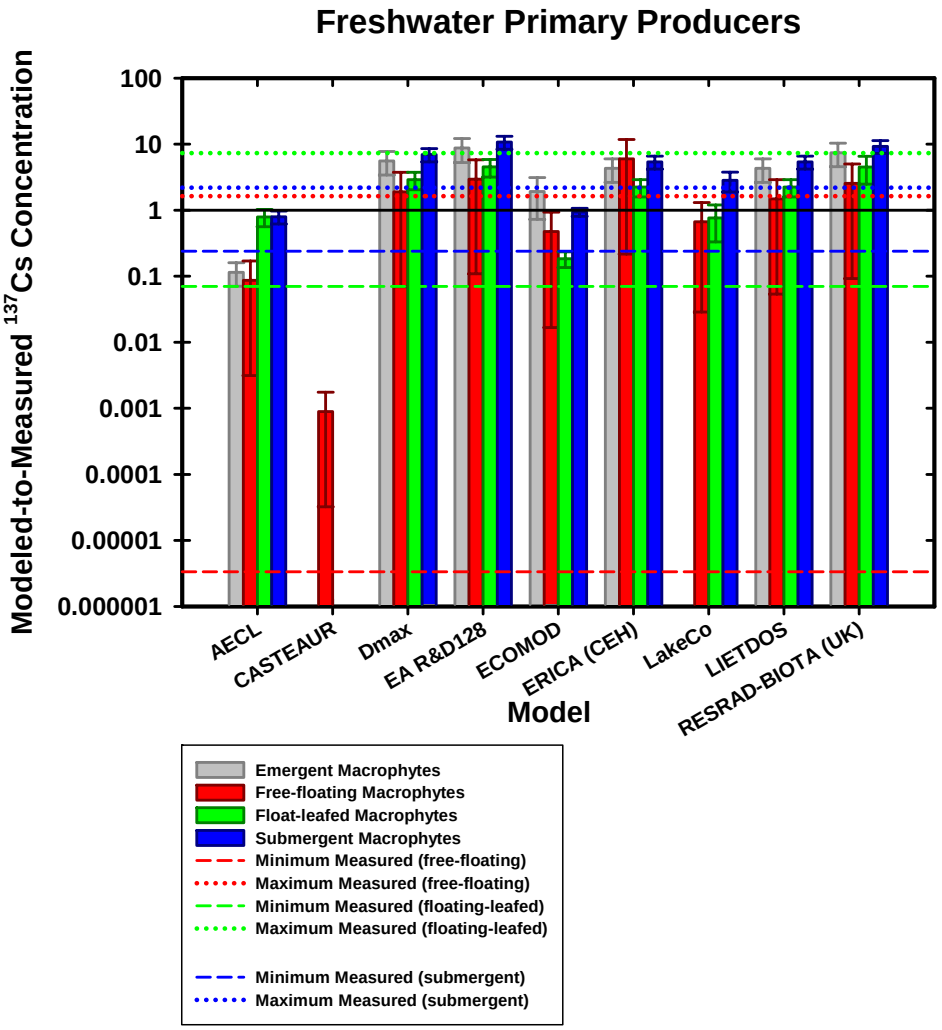

Fig. 5.3. Comparison of modelled-to-measured ^{137}Cs concentrations in Perch Lake freshwater primary producers. Dashed and dotted horizontal lines represent minimum and maximum measured values in the lake for a given type of primary producer Note that inadequate data were available to compare the modelled-to-measured range to minimum and maximum values for emergent macrophytes. Error bars represent the standard error in predicted values for a given species of primary producer by a given model.

Table 5.13. Summary of Z-scores for the prediction of ^{137}Cs activity concentrations in different types of freshwater primary producers. Values in shaded cells represent Z-scores that are greater than 3.

Type of Primary Producer	Year	Prediction No.	Z-scores for ^{137}Cs Concentration Data (by Organism Type)									
			AECL	CASTEAUR	D-Max	ECOMOD	EA R&D128	ERICA (CEH)	ERICA (NRPA)	LAKECO-B	LIETDOS-BIO	RESRAD-BIOTA (UK)
Emergent (rooted)	1994	PL33	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Emergent (rooted)	1995	PL42	0.01	n.a.	0.03	0.01	0.06	0.02	n.a.	n.a.	0.02	0.05
Emergent (rooted)	1996	PL48	0.02	n.a.	0.05	0.02	0.09	0.04	n.a.	n.a.	0.04	0.08
Free-Floating (unrooted)	1994	PL31	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Free-Floating (unrooted)	1995	PL39	1.7	2.0	5.6	0.1	9.7	21.9	n.a.	0.6	3.9	8.2
Floating-Leafed (rooted)	1968	PL3	1.0	n.a.	3.7	2.4	7.5	2.3	n.a.	2.4	2.3	n.a.
Floating-Leafed (rooted)	1969	PL8	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Floating-Leafed (rooted)	1970	PL14	2.4	n.a.	6.2	5.0	12.7	3.6	n.a.	5.0	3.6	n.a.
Floating-Leafed (rooted)	1971	PL26	1.7	n.a.	2.1	2.7	4.7	0.9	n.a.	2.7	0.9	3.8
Floating-Leafed (rooted)	1994	PL32	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Floating-Leafed (rooted)	1995	PL41	0.6	n.a.	2.8	1.7	5.4	1.8	n.a.	0.3	1.8	4.4
Floating-Leafed (rooted)	1996	PL47	1.3	n.a.	1.0	1.9	2.7	0.3	n.a.	0.7	0.3	2.0
Submergent (rooted)	1995	PL40	0.7	n.a.	7.8	0.3	13.0	5.6	n.a.	1.6	5.6	11.0
Submergent (rooted)	1996	PL46	0.1	n.a.	24.9	0.2	40.2	18.4	n.a.	9.2	18.4	34.1

Table 5.14. Comparison of the efficacy of model predictions of ^{137}Cs activity concentrations for different types of aquatic primary producers.

Radionuclide	Type of Freshwater Primary Producer	Number of predictions Z>3	No. of Predictions	[a] Efficacy Measure (%)
^{137}Cs	Emergent (rooted)	0	14	100%
^{137}Cs	Free-floating (unrooted)	6	9	33%
^{137}Cs	Floating-leafed (rooted)	23	38	39%
^{137}Cs	Submergent (rooted)	11	16	31%
^{137}Cs	**ALL Primary Producers**	**40**	**77**	**48%**

[a] The Efficacy Measure represents the percentage of model predictions with Z-scores that fall below a value of 3 (which represents an under- or over-prediction relative to the predictions generated by all models).

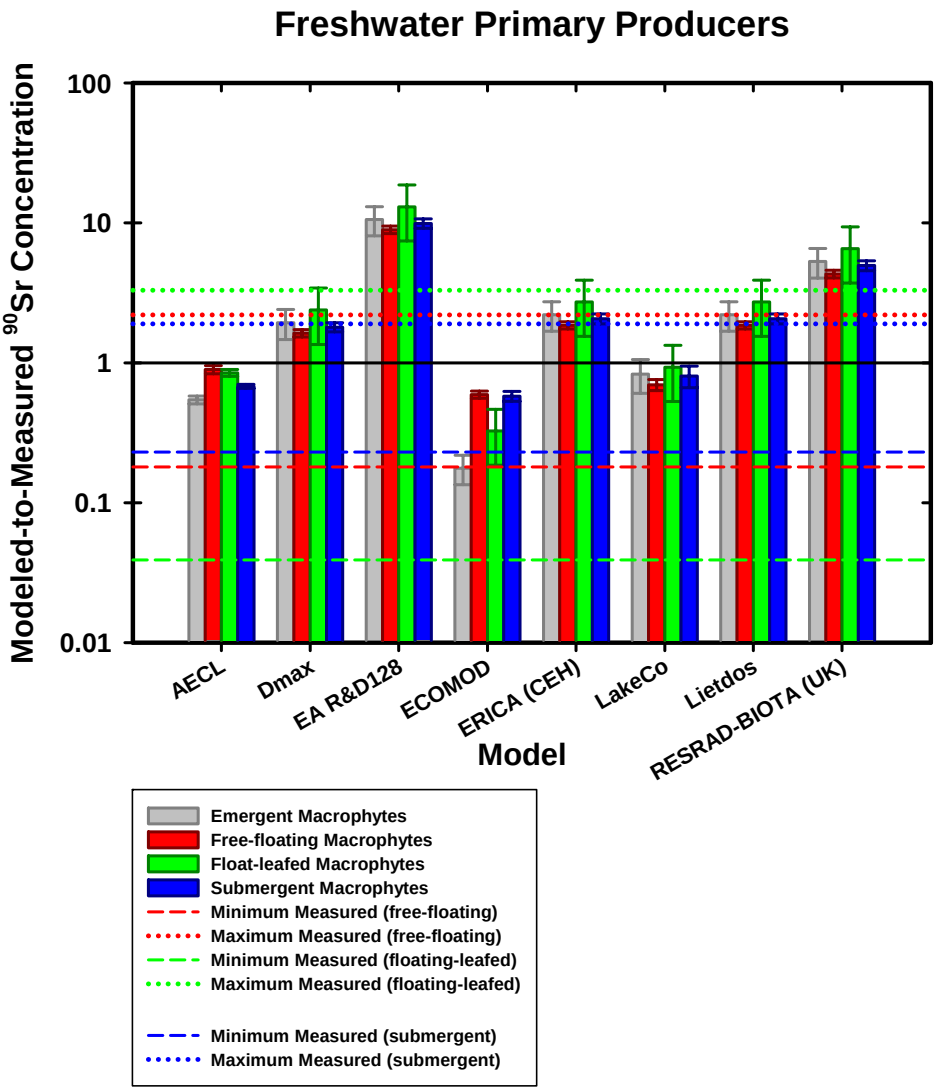

Fig. 5.4. Comparison of modelled-to-measured ^{90}Sr concentrations in Perch Lake freshwater primary producers. Dashed and dotted horizontal lines represent minimum and maximum measured values in the lake for a given type of primary producer. Note that inadequate data were available to compare the modelled-to-measured range to minimum and maximum values for emergent macrophytes. Error bars represent the standard error in predicted values for a given species of primary producer by a given model.

Strontium-90

Strontium-90 levels were predicted for Perch Lake primary producers by: AECL, ERICA (CEH), EA R&D128, D-Max, ECOMOD, LAKECO-B, LIETDOS-BIO and RESRAD-BIOTA (UK). Predictions using the AECL, D-Max, LAKECO-B, ECOMOD and LIETDOS-BIO models fell within the minimum-to-maximum ranges measured in the lake for all types of primary producers (Figure 5.4). However, predictions by ECOMOD, whilst in the observed range, were consistently low. By comparison, the EA R&D128 model tended to over-predict ^{90}Sr concentrations in all types of primary producers by an order of magnitude or more.

Whilst, RESRAD-BIOTA (UK) also consistently over-predicted ^{90}Sr activity concentrations, the degree of over-prediction was less than by EA R&D128.

Evaluation of Z-scores and the corresponding efficacies for ^{90}Sr in Perch Lake primary producers indicated that overall, efficacies tended to be relatively higher than those that had been generated for ^{137}Cs in floating-leafed and submergent, rooted primary producers (Tables 5.15 and 5.16).

Table 5.15. Summary of Z-scores for the prediction of ^{90}Sr activity concentrations in different tyeps of freshwater primary producers. Values in shaded cells represent Z-scores that are greater than 3.

Type of Primary Producer	Year	Prediction No.	Z-scores for ^{90}Sr Concentration Data (by Organism Type)								
			AECL	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	ERICA (NRPA)	LAKECO-B	LIETDOS	RESRAD-BIOTA (UK)
Emergent (rooted)	1970	PL15	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Emergent (rooted)	1971	PL27	3.9	0.7	40.2	7.2	2.1	n.a.	4.5	2.1	16.3
Emergent (rooted)	1994	PL33	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Emergent (rooted)	1995	PL42	1.6	4.6	40.4	2.7	5.7	n.a.	1.0	5.7	18.6
Emergent (rooted)	1996	PL48	0.3	0.9	8.3	0.7	1.1	n.a.	0.2	1.1	3.7
Free-Floating (unrooted)	1968	PL1	3.1	12.8	134.2	4.5	16.5	n.a.	5.4	16.5	n.a.
Free-Floating (unrooted)	1971	PL24	n.a.	2.3	33.9	2.0	3.4	n.a.	n.a.	3.4	14.7
Free-Floating (unrooted)	1994	PL31	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Free-Floating (unrooted)	1995	PL39	0.04	0.9	12.9	0.8	1.3	n.a.	0.3	1.3	5.6
Free-Floating (unrooted)	1996	PL45	0.1	1.7	19.8	0.9	2.3	n.a.	0.8	2.3	8.7
Free-Floating (unrooted)	1997	PL51	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Floating-Leafed (rooted)	1968	PL3	0.1	0.5	1.9	0.9	0.4	n.a.	0.8	0.4	0.4
Floating-Leafed (rooted)	1969	PL8	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Floating-Leafed (rooted)	1970	PL14	7.8	21.6	6.0	26.8	20.7	n.a.	25.7	20.7	10.9
Floating-Leafed (rooted)	1971	PL26	0.03	1.7	16.0	1.0	2.2	n.a.	0.2	2.2	7.4
Floating-Leafed (rooted)	1994	PL32	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Floating-Leafed (rooted)	1995	PL41	0.1	0.5	4.1	1.4	0.4	n.a.	1.0	0.4	1.3
Floating-Leafed (rooted)	1996	PL47	0.3	0.5	2.2	1.1	0.4	n.a.	0.9	0.4	0.5
Submergent (rooted)	1971	PL25	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Submergent (rooted)	1995	PL40	0.5	1.6	16.2	0.6	2.1	n.a.	0.1	2.1	7.3
Submergent (rooted)	1996	PL46	0.4	0.6	8.7	0.6	0.9	n.a.	0.5	0.9	3.7

Table 5.16. Comparison of the efficacy of model predictions of ^{90}Sr activity concentrations for different types of aquatic primary producers.

Radionuclide	Type of Freshwater Primary Producer	Number of predictions Z>3	Total n	[a] Efficacy Measure (%)
^{90}Sr	Emergent (rooted)	15	24	38%
^{90}Sr	Free-floating (unrooted)	19	29	34%
^{90}Sr	Floating-leafed (rooted)	14	41	66%
^{90}Sr	Submergent (rooted)	6	16	63%
^{90}Sr	**ALL Primary Producers**	**54**	**110**	**51%**

[a] The Efficacy Measure represents the percentage of model predictions with Z-scores that fall below a value of 3 (which represents an under- or over-prediction relative to the predictions generated by all models).

5.4.1.3. Modelled-to-Measured Comparisons for Freshwater Invertebrates

Cobalt-60

Cobalt-60 activity concentrations were predicted in Perch Lake invertebrates by: CASTEAUR, D-Max, EA R&D128, ECOMOD, ERICA (CEH), ERICA (NRPA), LAKECO-B, RESRAD-BIOTA (NRPA) and RESRAD-BIOTA (UK). LAKECO-B was only used to predict ^{60}Co levels in zooplankton and the RESRAD-BIOTA (NRPA) approach was only applied to make predictions for snails. Unfortunately, the available data were insufficient to allow comparison of predictions to a range in observed invertebrate activity concentrations.

In general, the ^{60}Co concentrations predicted by CASTEAUR and ECOMOD were under-estimated for zooplankton by up to approximately 17-fold, with an under-prediction by several orders of magnitude for the RESRAD-BIOTA (UK) approach (Figure 5.5). In the case of both the CASTEAUR and ECOMOD approaches, the ^{60}Co CRs originated from experiments that had been conducted under controlled laboratory conditions. By comparison, the D-Max model over-predicted ^{60}Co concentrations in zooplankton by an order of magnitude, probably because maximum (identified) literature values are often applied as part of the D-Max conservative screening approach, whereas the LAKECO-B model predicted a value that was equal to the measured concentration. The remaining models (EA R&D128, ERICA (CEH), LAKECO-B and RESRAD-BIOTA (UK)) produced predictions that fell within an order of magnitude of measured values, which is considered reasonable given the natural variability in invertebrate ^{60}Co activity that is expected to be inherent in aquatic ecosystems.

In the case of macroinvertebrates, the ^{60}Co values predicted by the CASTEAUR, EA R&D128, ECOMOD and RESRAD-BIOTA (UK) models were reasonably close to the measured value, falling within less than an order of magnitude of measured values (Figure 5.5). The remaining two models that were used to predict ^{60}Co in macroinvertebrates (D-Max and ERICA (CEH)) over-estimated activity concentrations by between one and two orders of magnitude. It would not be surprising for a relatively high variability to occur in macroinvertebrate CRs due to the large range of life-styles that exist for this group of organisms, the relative importance of water versus sediment in bioaccumulation of contaminants, such as ^{60}Co, and the heterogeneity in the distribution of ^{60}Co and other contaminants that typically occurs in sediments of natural ecosystems.

Estimated snail ^{60}Co concentrations based on the D-Max approach over-predicted the measured value by a factor of 21, whereas CASTEAUR and ECOMOD under-predicted ^{60}Co in snails by between one and two orders of magnitude (Figure 5.5). Again, the latter two approaches, have applied CRs generated under laboratory as opposed to field conditions, which appear not to reflect ^{60}Co transfer in Perch Lake; the CASTEAUR value was derived from studies of macroinvertebrates and assumed to be applicable to snails. By comparison, RESRAD-BIOTA (UK) under-predicted snail ^{60}Co levels by more than four orders of magnitude, applying the allometric option of the model, which estimates parameters (such as contaminant uptake rates) that are related to organism radionuclide exposure rates, on the basis of organism body mass, as described above. RESRAD-BIOTA (NRPA) applied the BiV (i.e. CR) option in the RESRAD-BIOTA tool and estimates of ^{60}Co activity concentrations in snails were close to the measured data.

There are insufficient data to consider Z-score statistics for ^{60}Co in individual types of Perch Lake invertebrates.

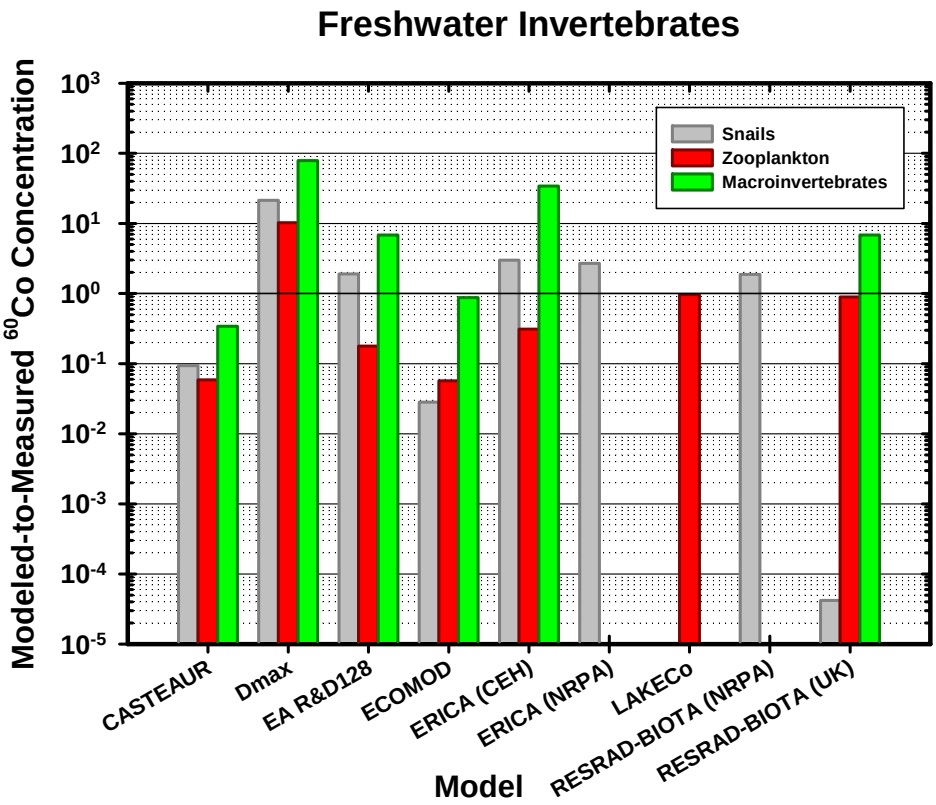

Fig. 5.5. Comparison of modelled-to-measured ^{60}Co concentrations in Perch Lake invertebrates. Predicted values could not be compared to minimum and maximum measured values in the lake for a given type of invertebrate due to the lack of historical data and the relatively large invertebrate biomass that was required to detect ^{60}Co in the sample, which made it necessary to composite samples.

Caesium-137

Caesium-137 concentrations were predicted in Perch Lake invertebrates by: CASTEAUR, D-Max, EA R&D128, ECOMOD, ERICA (CEH), RESRAD-BIOTA (UK) and LAKECO-B (zooplankton only).

In general, predicted ^{137}Cs concentrations in both zooplankton and macroinvertebrates varied widely between modelling approaches, with inter-model differences as great as three orders of magnitude between models for a given type of invertebrate receptor (Figure 5.6). Of the model predictions for ^{137}Cs in zooplankton, was under-predicted by CASTEAUR and EA R&D128 by approximately two orders of magnitude and by ECOMOD by almost one order of magnitude. Again, the CASTEAUR approach used a CR for invertebrates based on laboratory experiments (Table 5.3). The ECOMOD approach, which applied ^{137}Cs CRs that had been based on field measurements taken from Chernobyl [125], also under-estimated ^{137}Cs levels in both zooplankton and macroinvertebrates by approximately an order of magnitude.

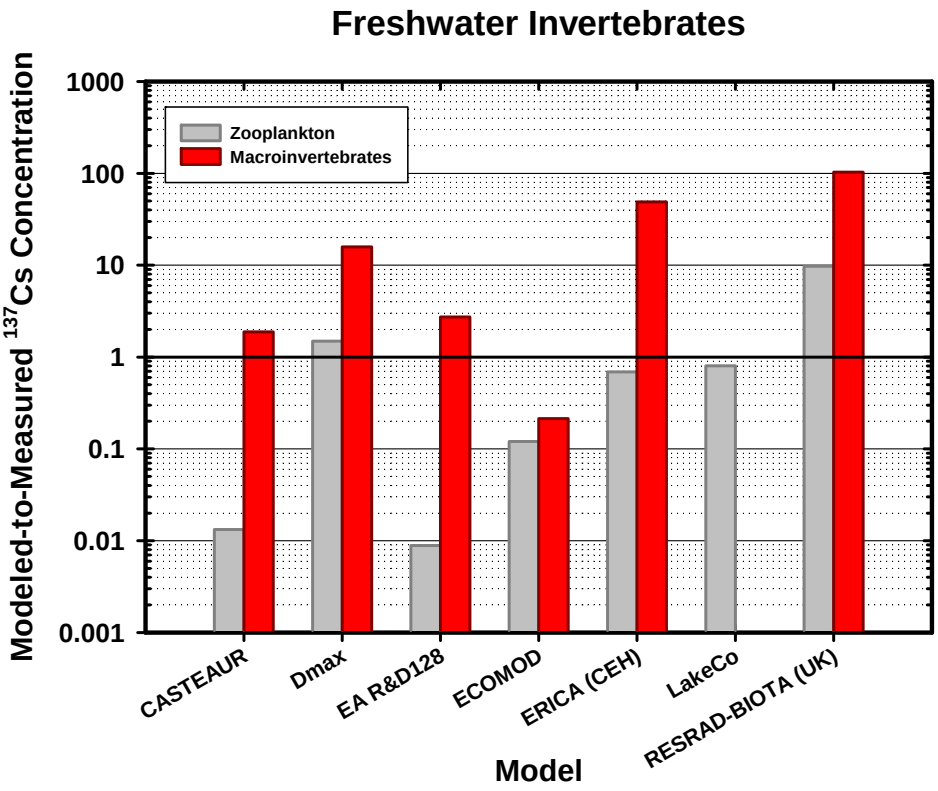

Fig. 5.6. Comparison of modelled-to-measured ^{137}Cs concentrations in Perch Lake invertebrates. Predicted values could not be compared to minimum and maximum measured values in the lake for a given type of invertebrate due to the lack of historical data and the relatively large invertebrate biomass that was required to detect ^{137}Cs in the sample, which made it necessary to composite samples.

By comparison, RESRAD-BIOTA (UK) over-predicted zooplankton by an order of magnitude and macroinvertebrates by two orders of magnitude using the RESRAD-BIOTA default BiVs (Figure 5.6). Over-estimates of ^{137}Cs in macroinvertebrates were also generated by D-Max (which applied the maximum CR found in the literature to estimate the ^{137}Cs activity concentration) and ERICA (CEH).

Model predictions that were generated for zooplankton by the D-Max, ERICA (CEH) and LAKECO-B approaches, as well as those produced for macroinvertebrates using CASTEAUR and EA R&D128, all fell relatively close to measured values (Figure 5.6).

As for ^{60}Co data were insufficient to enable comparison of Z-score statistics.

Strontium-90

Strontium activity concentrations were predicted by: D-Max, EA R&D128, ECOMOD, ERICA (CEH), LIETDOS-BIO and RESRAD-BIOTA (UK); LIETDOS-BIO was used to make predictions for mussels only.

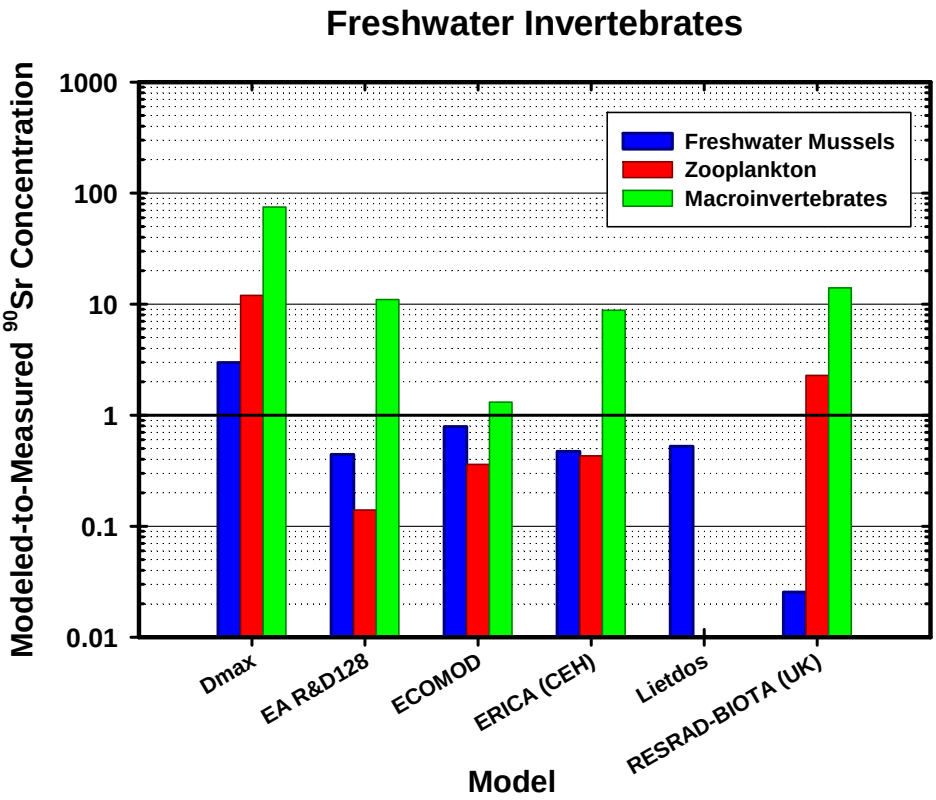

Fig. 5.7. Comparison of modelled-to-measured ^{90}Sr concentrations in Perch Lake invertebrates. Predicted values could not be compared to minimum and maximum measured values in the lake due to the lack of historical data and the relatively large invertebrate biomass that was required to detect ^{90}Sr in the sample, which made it necessary to composite samples.

Most predictions of ^{90}Sr activity concentrations in invertebrates were within an order of magnitude of the observed data (Figure 5.7). Notable exceptions were RESRAD-BIOTA (UK), which under-estimated the mussel ^{90}Sr activity concentration by almost two orders of magnitude, and D-Max which over-predicted ^{90}Sr activity concentrations in zooplankton and macroinvertebrates by between one and two orders of magnitude. The under-prediction of ^{90}Sr generated by RESRAD-BIOTA (UK) may have been due to the application of the allometric function to derive freshwater mussel activity concentrations. When applying the BiV option, RESRAD-BIOTA (UK) over-predicted ^{90}Sr in macroinvertebrates by slightly greater than an order of magnitude (Figure 5.7).

Estimated Z-scores for the predictions of ^{90}Sr concentrations in mussels ranged from 0.2 to 1.1 across the participating models.

5.4.1.4. Modelled-to-Measured Comparisons for Freshwater Fish

Cobalt-60

Cobalt-60 concentrations were predicted in Perch Lake fish using all of the eleven modelling approaches that participated in this scenario. With the exception of RESRAD-BIOTA

(NRPA) (which only provided predictions for brown bullheads and pumpkinseeds), participating models made predictions for the four types of fish under consideration.

In general, predictions for ^{60}Co concentrations in cyprinids, pumpkinseeds and brown bullheads that were generated by AECL and ECOMOD, both of which accounted for site-specific water chemistry (AECL because site specific measurements were used to generate CR values), fell within less than 2-fold of measured values (Figure 5.8). In addition, modelled values that were produced by LAKECO-B and ERICA (NRPA) for bullheads, cyprinids and pumpkinseeds; by ERICA (CEH), EA R&D128 and LIETDOS-BIO for cyprinids; and by RESRAD-BIOTA (UK) for all fish species, fell within the range of ^{60}Co activity concentrations that were present in the lake.

With the exception of yellow perch, fish ^{60}Co activity concentrations tended to be under-predicted using CASTEAUR. Again, the CR values that were applied to make the CASTEAUR predictions were generated from controlled laboratory experiments and assumptions with regard to fish physiology (e.g. diet, growth rate and feeding rate).

In many cases, ^{60}Co activity concentrations were over-predicted in yellow perch (Figure 5.8). Such over-predictions were prevalent when a given modelling approach applied the same fish ^{60}Co CR regardless of trophic position, not accounting for the tendency of ^{60}Co to decrease in concentration with increasing trophic level. Consequently, application of a ^{60}Co CR for a forage fish species to estimate the activity concentration of a top predator, such as yellow perch, will result in an over-estimation of ^{60}Co in the top predator. Similarly, application of a top predator CR to estimate ^{60}Co levels in a forage fish species would under-predict ^{60}Co in the forage species. If trophic position is taken into account through the selection of CRs that are specific to a trophic level, as was done within the AECL approach, predicted yellow perch ^{60}Co activity concentrations fall close to measured values (Figure 5.8). However, whilst the LAKECO-B model applied a 'dilution factor' to account for reducing ^{60}Co activity concentrations with increasing trophic level it tended to under-predict ^{60}Co concentrations in the top predator (yellow perch). In addition, whole-body ^{60}Co concentrations are estimated based on concentration data and compartment mass for target tissues into which it concentrates (where LAKECO-B assumes it is primarily found in the intestines). Since there can be variability in the relative concentration of ^{60}Co in various tissues within the body, it is possible that different values could be found depending upon the source literature that was used and also, whether the organisms had reached steady state with respect to the radionuclide levels in their various tissues.

Cobalt-60 was over-estimated by the RESRAD-BIOTA (NRPA) approach in both brown bullheads and pumpkinseeds, due to the application of the relatively conservative default CR that is recommended in the RESRAD-BIOTA tool.

Evaluation of the Z-scores supports the above observations with a relatively large number of Z-scores across fish species exceeded a value of 3, particularly for D-Max, EA R&D128, ERICA (CEH), ERICA (NRPA) and LIETDOS-BIO (Table 5.17). Based on these data, efficacies that ranged from 28% (for yellow perch) to 55% (for pumpkinseeds) were found, with an efficacy of 48% when the Z-score data for all fish species were pooled together (Table 5.18).

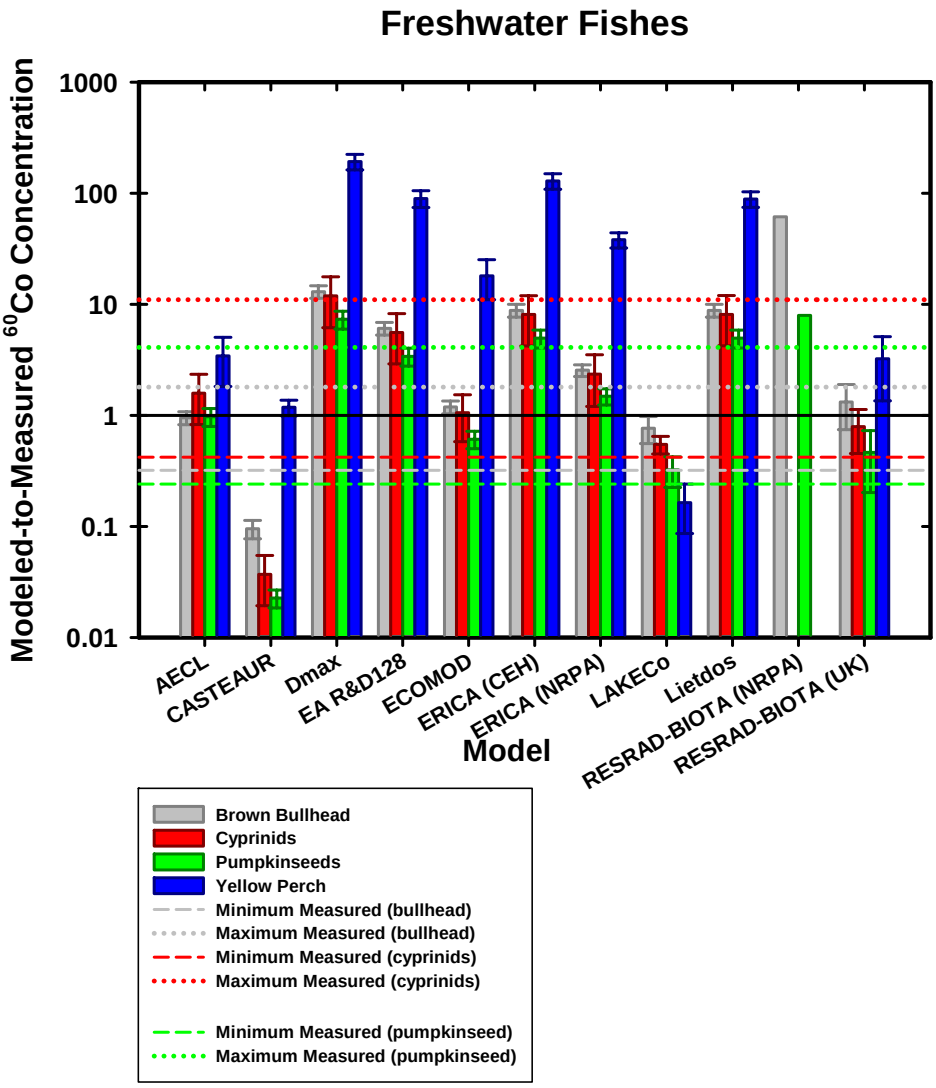

Fig. 5.8. Comparison of modelled-to-measured ^{60}Co concentrations in Perch Lake freshwater fish. Dashed and dotted horizontal lines represent minimum and maximum measured values in the lake for a given type of fish species. Note that inadequate data were available to compare the modelled-to-measured range to minimum and maximum values for yellow perch. Error bars represent the standard error in predicted values for a given fish species by a given model.

Table 5.17. Summary of Z-scores for the prediction of ^{60}Co activity concentrations in different types of freshwater fish. Values in shaded cells represent Z-scores that are greater than 3.

Type of Fish	Year	Prediction No.	Z-scores for ^{60}Co Concentration Data (by Organism Type)										
			AECL	CASTEAUR	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	ERICA (NRPA)	LAKECO-B	LIETDOS-BIO	RESRAD-BIOTA (NRPA)	RESRAD-BIOTA (UK)
Brown Bullhead	1968	PL4	0.2	2.0	25.2	10.7	0.8	16.2	3.2	0.9	16.2	n.a.	1.5
Brown Bullhead	1969	PL11	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Brown Bullhead	1970	PL20	0.03	1.1	15.7	6.6	0.1	10.2	2.2	0.8	10.2	n.a.	1.0
Brown Bullhead	1994	PL38	2.0	4.1	25.8	10.0	1.0	16.3	1.7	1.5	16.4	n.a.	1.3
Brown Bullhead	1995	PL44	1.6	2.9	65.2	28.7	2.6	43.6	9.4	1.4	43.7	211	7.2
Brown Bullhead	1996	PL50	n.a.	2.6	38.3	16.4	1.1	25.1	5.2	1.1	25.1	n.a.	6.2
Cyprinid species	1970	PL18	5.0	2.2	53.5	23.6	2.4	35.3	8.7	1.0	35.4	n.a.	1.9
Cyprinid species	1994	PL36	n.a.	1.7	11.1	4.4	0.5	7.1	0.8	0.5	7.1	n.a.	0.7
Cyprinid species	1995	PL43	0.8	2.5	10.0	3.3	1.4	6.1	0.2	1.6	6.1	n.a.	0.6
Pumpkinseed	1969	PL10	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Pumpkinseed	1970	PL19	1.9	4.2	43.1	17.7	0.7	27.7	5.1	3.1	27.7	n.a.	3.9
Pumpkinseed	1971	PL29	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Pumpkinseed	1994	PL37	n.a.	4.4	29.2	11.4	0.8	18.6	2.2	1.4	18.6	n.a.	1.9
Pumpkinseed	1997	PL53	0.4	0.6	1.0	0.1	0.5	0.4	0.2	0.4	0.4	4.2	0.2
Yellow Perch	1968	PL5	n.a.	0.8	505	237	54.7	338	97.7	1.7	231	n.a.	9.3
Yellow Perch	1969	PL12	n.a.	0.02	494	224	30.6	330	95.3	2.8	226	n.a.	1.1

Table 5.18. Comparison of the efficacy of model predictions of ^{60}Co activity concentrations for different types of freshwater fish.

Radionuclide	Type of Fish	Number of predictions Z>3	Total n	[a] Efficacy Measure (%)
^{60}Co	Brown bullhead	27	51	47%
^{60}Co	Cyprinids	14	30	53%
^{60}Co	Pumpkinseed	14	31	55%
^{60}Co	Yellow perch	13	18	28%
^{60}Co	**ALL Fish**	**68**	**130**	**48%**

[a] The Efficacy Measure represents the percentage of model predictions with Z-scores that fall below a value of 3 (which represents an under- or over-prediction relative to the predictions generated by all models).

Strontium-90

Of the eleven modelling approaches that participated in this scenario, only CASTEAUR did not report fish ^{90}Sr activity concentrations.

In general, ^{90}Sr concentrations predicted by AECL, D-Max and ECOMOD for cyprinids, pumpkinseeds and brown bullheads fell within approximately 2-fold of measured values (Figure 5.9). Of these predictions, only one (i.e. the prediction for cyprinids by D-Max) fell outside of the minimum-to-maximum range, although this prediction was only 2.3-fold higher than the measured value. It is important to note that each of these approaches accounts for Ca water concentrations in estimating ^{90}Sr bioaccumulation by Perch Lake fish (the AECL approach through using site-specific values and the other approaches by using predictive relationships that account for Ca levels in the lake). Since Sr and Ca compete during uptake by freshwater biota [130, 131], it is not surprising that predictions were improved with consideration of Perch Lake water chemistry. Predictions generated by the LAKECO-B, RESRAD-BIOTA (NRPA) and RESRAD-BIOTA (UK) models also fell fairly close to measured values, although in most cases, they were slightly below the range found in Perch Lake.

By comparison, the predicted ^{90}Sr concentrations for all fish species considered were under-estimated by the ERICA (CEH), EA R&D128, ERICA (NRPA) and LIETDOS-BIO models, with modelled-to-measured ratios of 0.02 to 0.10 for cyprinids, of 0.01 to 0.05 for pumpkinseeds and of 0.014 to 0.06 for brown bullheads, respectively. These under-estimations may be because the CRs that had been applied using these approaches were based on data that had been collected for edible fish tissue (i.e. muscle), as opposed to whole body values (N.B. data provenance is not clear in all instances). Strontium-90 levels may be expected to be approximately 1000-fold higher in bone than in flesh, and whole fish would contain approximately 5% bone [14, 132], this could account for the differences that were observed for these predictions. If the predictions for these models are 'corrected' to account for bone (which can be accomplished by multiplying model predictions by a factor of 23 [132], the resultant predictions fall much closer to measured values (Figure 5.10).

Z-scores for ^{90}Sr activity predictions in Perch Lake fish are presented in Table 5.19. The overall efficacy for ^{90}Sr in fish, when data for all species were pooled together, was 51% (Table 5.20).

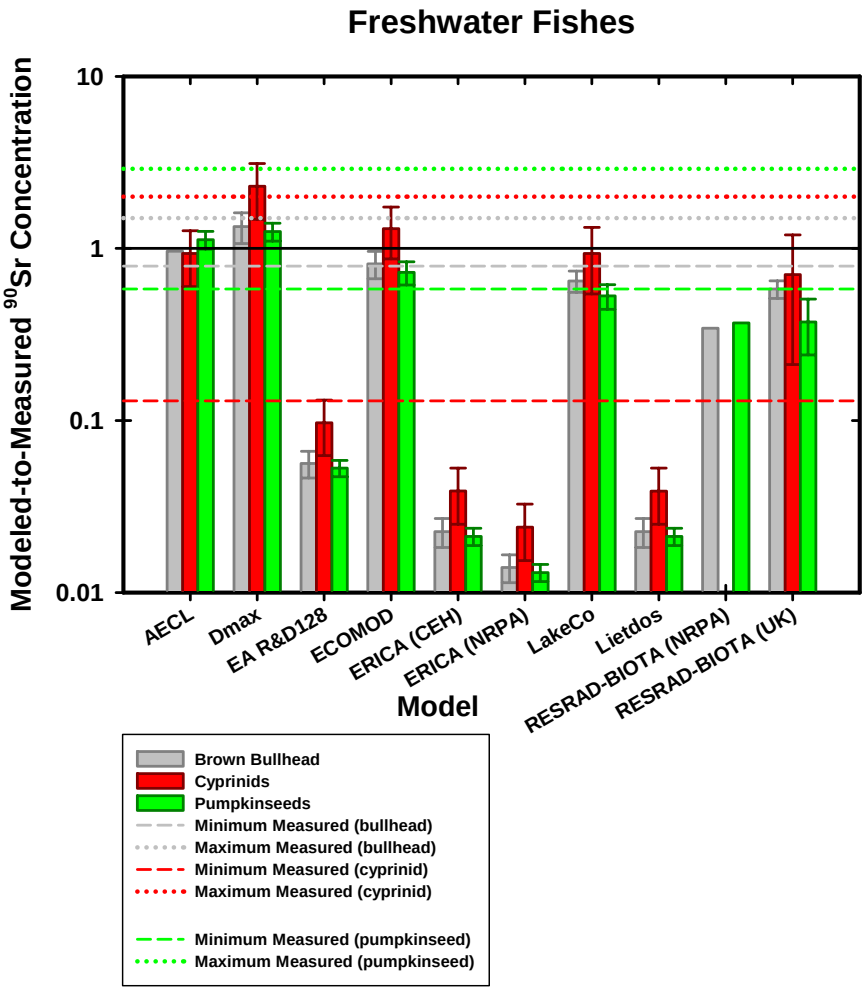

Fig. 5.9. Comparison of modelled-to-measured ^{90}Sr concentrations in Perch Lake freshwater fish. Dashed and dotted horizontal lines represent minimum and maximum measured values in the lake for a given type of fish species. Error bars represent the standard error in predicted values for a given fish species by a given model.

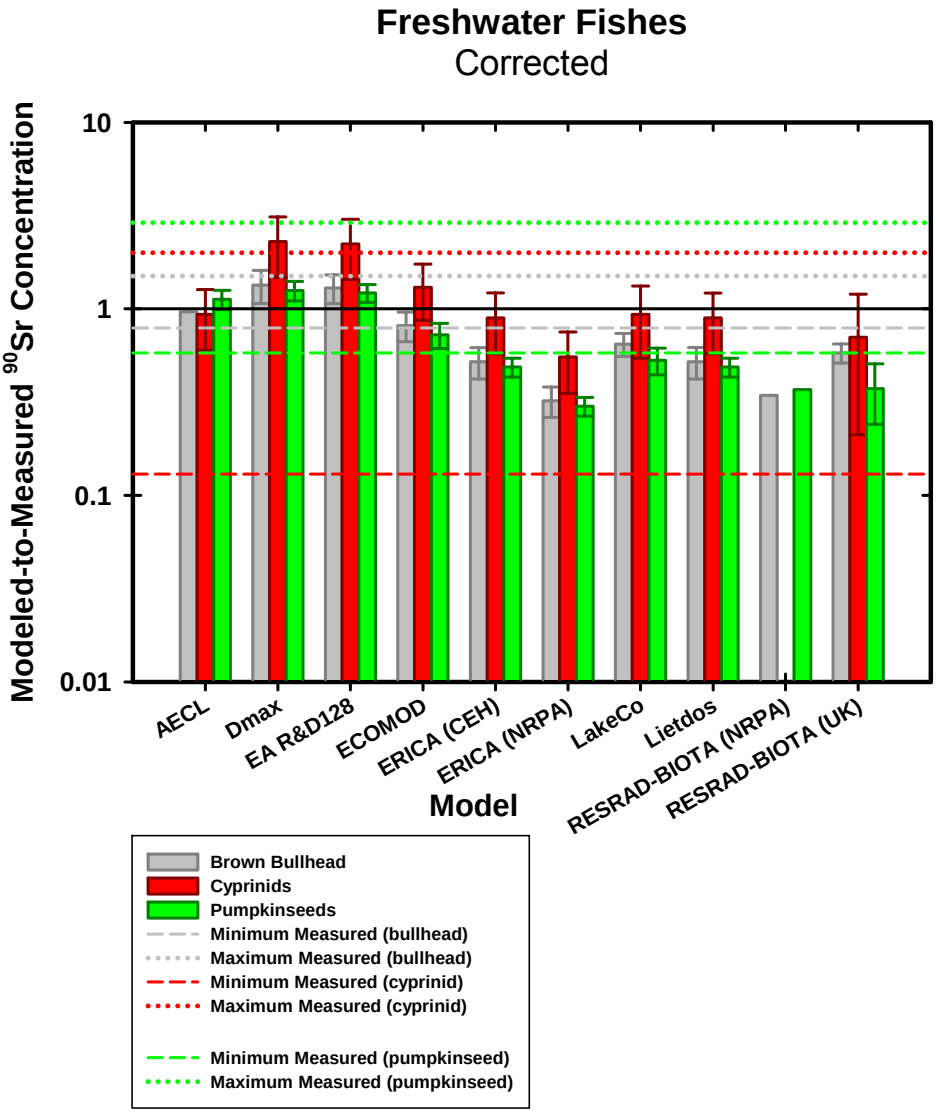

Fig. 5.10. Comparison of modelled-to-measured ^{90}Sr concentrations in Perch Lake freshwater fish following 'correction' of EA R&D128, ERICA and LIETDOS predictions for whole-body as opposed to fish muscle CRs. Dashed and dotted horizontal lines represent minimum and maximum measured values in the lake for a given type of fish species. Error bars represent the standard error in predicted values for a given fish species by a given model.

Table 5.19. Summary of Z-scores for the prediction of ^{90}Sr activity concentrations in different types of freshwater fish. Values in shaded cells represent Z-scores that are greater than 3.

Type of Organism	Year	Prediction No.	Z-scores for ^{90}Sr Concentration Data (by Organism Type)									
			AECL	D-Max	EA R&D128	ECOMOD	ERICA (CEH)	ERICA (NRPA)	LAKECO-B	LIETDOS	RESRAD-BIOTA (NRPA)	RESRAD-BIOTA (UK)
Brown Bullhead	1994	PL38	0.1	1.3	1.9	0.08	2.0	2.0	0.5	2.0	n.a.	0.7
Brown Bullhead	1995	PL44	n.a.	0.1	1.4	0.5	1.5	1.5	0.7	1.5	1.0	0.7
Cyprinid species	1970	PL18	1.5	4.0	4.9	0.03	5.2	5.2	2.8	5.2	n.a.	5.2
Cyprinid species	1971	PL28	3.0	0.3	5.1	1.7	5.2	5.3	3.1	5.2	n.a.	5.3
Cyprinid species	1994	PL36	7.4	14.7	21.0	1.5	21.9	22.2	5.4	21.9	n.a.	7.3
Cyprinid species	1997	PL52	2.5	10.0	2.2	4.3	2.5	2.6	2.9	2.5	n.a.	3.0
Pumpkinseed	1971	PL29	0.9	0.8	13.6	3.6	14.0	14.1	8.5	14.0	n.a.	14.2
Pumpkinseed	1994	PL37	3.6	4.8	6.4	0.2	6.7	6.8	1.5	6.7	n.a.	2.5
Pumpkinseed	1996	PL49	0.001	0.8	7.9	4.1	8.1	8.2	4.8	8.1	n.a.	5.2
Pumpkinseed	1997	PL53	0.08	0.4	2.3	0.9	2.4	2.4	1.2	2.4	1.5	1.3

Table 5.20. Comparison of the efficacy of model predictions of ^{90}Sr activity concentrations for different types of freshwater fish.

Radionuclide	Type of Fish	Number of predictions Z>3	Total n	[a] Efficacy Measure (%)
^{90}Sr	Brown bullhead	0	18	100%
^{90}Sr	Cyprinids	23	36	36%
^{90}Sr	Pumpkinseed	20	37	41%
^{90}Sr	**ALL Fish**	**43**	**91**	**51%**

[a] The Efficacy Measure represents the percentage of model predictions with Z-scores that fall below a value of 3 (which represents an under- or over-prediction relative to the predictions generated by all models).

5.4.1.5. Modelled-to-Measured Comparisons for Herpetofauna

Only four of the eleven models (ERICA (CEH), EA R&D128, RESRAD-BIOTA (UK) and D-Max) were used to predict radionuclide concentrations in Perch Lake herpetofauna (which included frogs and turtles). Of these, ERICA (CEH) provided predictions for frogs only. It was not possible to apply the AECL approach, as there were insufficient historical data.

Cobalt-60

In the case of AECL, it was, again, not possible to perform such predictions using the approach that was taken in participating in this scenario due to the lack of historical data. As a result, it was not feasible to establish reference lake values for these species against which to test whether herpetofaunal CRs were consistent in the lake over time. With the exception of the RESRAD-BIOTA (UK) predictions for turtles, which were under-estimated by two orders of magnitude or more using an allometric modelling approach, all participating models over-predicted ^{60}Co concentrations in both frogs and turtles by greater than one to greater than three orders of magnitude (Figure 5.11). For example, in the case of bullfrogs, ^{60}Co values were over-predicted by factors of 60, 2200, 270 and 31 using the ERICA (CEH), EA R&D128, D-Max and RESRAD-BIOTA (UK) models, respectively. In addition, green frog values were over-predicted by 40, 1400, 180 and 20 fold, respectively. By comparison, modelled-to-measured ratios of 507 and 65 were found for painted turtles, and of 250 and 32 for snapping turtles, based on predictions made by the EA R&D128 and D-Max models, respectively. With one exception, all Z-scores were much higher than a value of 3 with comparatively poor efficacies of (0-33%) (Tables 5.21- 5.22).

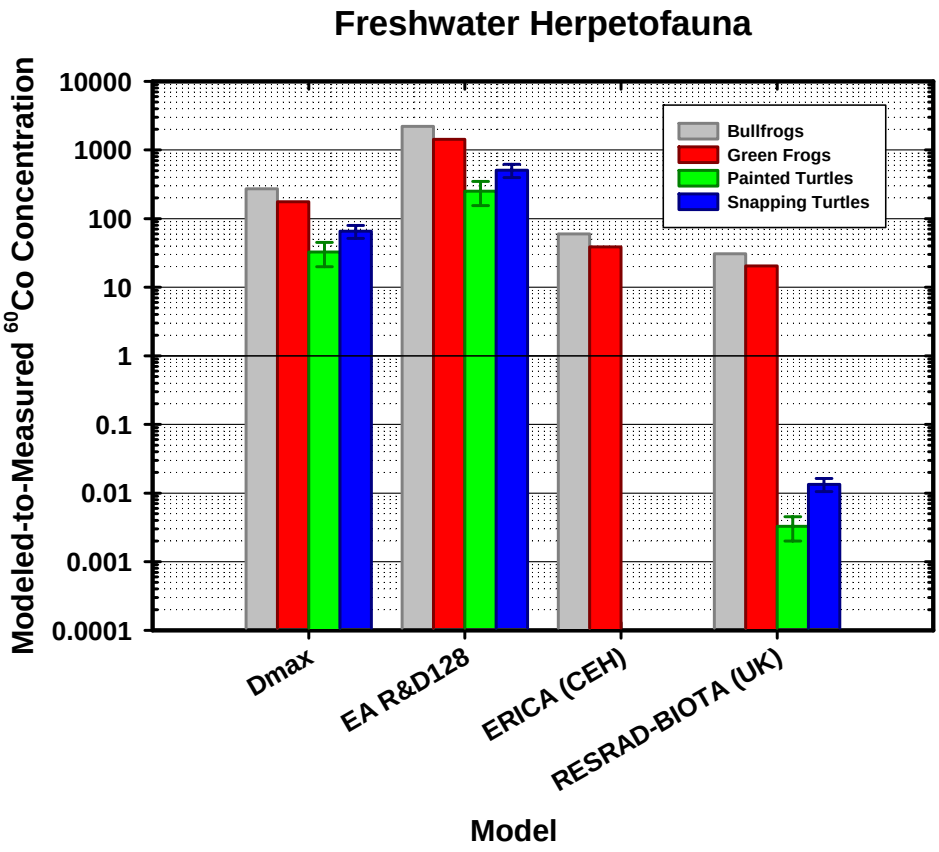

Fig. 5.11. Comparison of modelled-to-measured ^{60}Co concentrations in Perch Lake herpetofauna. Predicted values could not be compared to minimum and maximum measured values in the lake for a given species due to the lack of historical data and the relatively small sample sizes for such species. Error bars represent the standard error in predicted values for a given species by a given model.

Table 5.21. Summary of Z-scores for the prediction of ^{60}Co activity concentrations in different types of freshwater primary producers. Values in shaded cells represent Z-scores that are greater than 3.

Type of Organism	Year	Prediction No.	Z-scores for ^{60}Co Concentration Data (by Organism Type)			
			D-Max	EA R&D128	ERICA (CEH)	RESRAD-BIOTA (UK)
Bullfrogs	1998	PL59	404	3290	87.8	44.3
Green Frogs	1998	PL58	245	1999	52.9	27.2
Painted turtle	1971	PL30	n.a.	n.a.	n.a.	n.a.
Painted turtle	1997	PL54	15.5	125	n.a.	0.8
Common Snapping Turtle	1971	PL23	n.a.	n.a.	n.a.	n.a.
Common Snapping Turtle	1997	PL55	n.a.	n.a.	n.a.	n.a.

Table 5.22. Comparison of the efficacy of model predictions of ^{60}Co activity concentrations for different types of freshwater herpetofauna.

Radionuclide	Species	Number of predictions Z>3	Total n	[a] Efficacy Measure (%)
^{60}Co	Bullfrogs	4	4	0%
^{60}Co	Green frogs	4	4	0%
^{60}Co	Painted turtles	2	3	33%
^{60}Co	Common snapping turtles	n.a.	n.a.	n.a.
^{60}Co	**ALL Herpetofauna**	**10**	**11**	**9%**

n.a. – not available.
[a] The Efficacy Measure represents the percentage of model predictions with Z-scores that fall below a value of 3 (which represents an under- or over-prediction relative to the predictions generated by all models).

Caesium-137

Predicted ^{137}Cs concentrations fell approximately 1- to 2-orders of magnitude higher than measured values for the herpetofaunal species considered for all but the RESRAD-BIOTA (UK) (Figure 5.12). For example, bullfrog modelled-to-measured ratios ranged from 20 (for ERICA (CEH)) to 220 (for EA R&D128), with values of 11 (ERICA (CEH)) to 129 (EA R&D128) for green frogs. Predicted ^{137}Cs concentrations for painted turtles and snapping turtles were also much higher than measured values, with over-predictions of approximately 90-fold and 60-fold for these species, respectively, for these three modelling approaches. These over-predictions are likely due to application of highly conservative CRs for species that are less well-understood and the unrealistic assumption of 100 % occupancy in the aquatic environment.

As for predictions of ^{60}Co in Perch Lake herpetofauna, most of the Z-scores were in excess of 3 (Table 5.23) with ranging from 0 – 33% (Table 5.24).

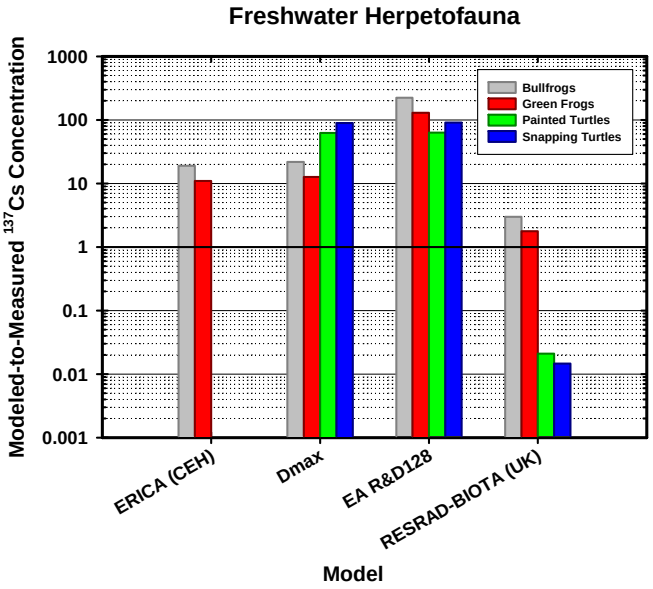

Fig. 5.12. Comparison of modelled-to-measured ^{137}Cs concentrations in Perch Lake herpetofauna. Predicted values could not be compared to minimum and maximum measured values in the lake for a given species due to the lack of historical data and the relatively small sample sizes for such species.

Table 5.23. Summary of Z-scores for the prediction of ^{137}Cs activity concentrations in different types of freshwater herpetofauna. Values in shaded cells represent Z-scores that are greater than 3.

Type of Organism	Year	Prediction No.	Z-scores for ^{137}Cs Concentration Data (by Organism Type)			
			D-Max	EA R&D128	ERICA (CEH)	RESRAD-BIOTA (UK)
Bullfrogs	1998	PL59	294	3137	253	28.0
Green frogs	1998	PL58	25.3	279	21.7	1.7
Painted turtles	1997	PL54	130	132	n.a.	1.4
Common snapping turtles	1997	PL55	n.a.	n.a.	n.a.	n.a.

Table 5.24. Comparison of the efficacy of model predictions of ^{137}Cs activity concentrations for different types of freshwater herpetofauna.

Radionuclide	Type of Fish	Number of predictions Z>3	Total n	[a] Efficacy Measure (%)
^{137}Cs	Bullfrogs	4	4	0%
^{137}Cs	Green frogs	3	4	25%
^{137}Cs	Painted turtles	2	3	33%
^{137}Cs	Common snapping turtles	n.a.	n.a.	n.a.
^{137}Cs	**ALL Herpetofauna**	**9**	**11**	**18%**

n.a. – not available.

[a] The Efficacy Measure represents the percentage of model predictions with Z-scores that fall below a value of 3 (which represents an under- or over-prediction relative to the predictions generated by all models).

Strontium-90

With the exception of the ERICA (CEH) approach, which generated predictions that fell within less than an order of magnitude of measured values, the three remaining models (EA R&D128, RESRAD-BIOTA (UK) and D-Max) largely over-predicted ^{90}Sr activity concentrations values in Perch Lake bullfrogs, with modelled-to-measured ratios of approximately 1- to 2-orders of magnitude (Figure 5.13). Both the D-Max and EA R&D128 approaches also over-predicted green frog ^{90}Sr levels by an order of magnitude, whereas predictions that had been generated using ERICA (CEH) and RESRAD-BIOTA (UK) fell within an order of magnitude of measured values. The default value within the ERICA Tool, as applied by ERICA (CEH) assumes the CR for benthic fish due to the lack of data for amphibians.

With the exception of RESRAD-BIOTA (UK), which applied an allometric approach, all other approaches used the same CR for the two frog species, suggesting that the differences in modelled to measured ^{90}Sr activity concentrations between these two frog species are likely to be due to inter-species differences in life mode or other factors.

Predictions of ^{90}Sr activity concentrations in turtles were over-predicted by approximately three-orders of magnitude by the D-Max and EA R&D128 models. Predictions by RESRAD-BIOTA (UK) were within an order of magnitude.

Strontium-90 Z scores all fell well above a value of three for Perch Lake herpetofauna, with the exception of those produced by the ERICA (CEH) approach for frog species. It was not possible to assess the efficacy of snapping turtle predictions because of insufficient data.

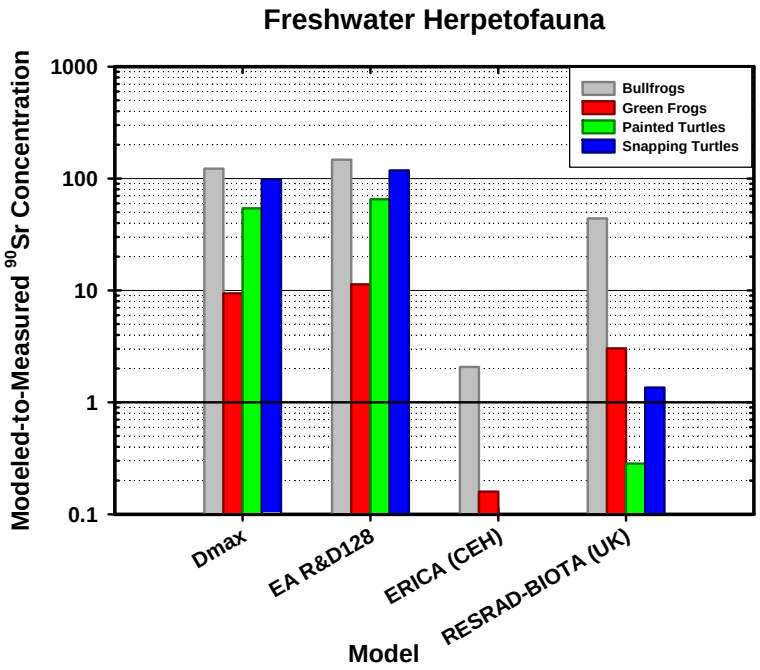

Fig. 5.13. Comparison of modelled-to-measured ^{90}Sr concentrations in Perch Lake herpetofauna. Predicted values could not be compared to minimum and maximum measured values in the lake for a given species due to the lack of historical data and the relatively small sample sizes for such species.

Table 5.25. Summary of Z-scores for the prediction of ^{90}Sr activity concentrations in idfferent types of freshwater herpetofauna. Values in shaded cells represent Z-scores that are grater than 3.

Type of Organism	Year	Prediction No.	Z-scores for ^{90}Sr Concentration Data (by Organism Type)			
			D-Max	EA R&D128	ERICA (CEH)	RESRAD-BIOTA (UK)
Bullfrogs	1998	PL59	145	175	1.3	51
Green Frogs	1998	PL58	7.3	8.9	0.7	1.8
Painted turtle	1997	PL54	71	86	n.a.	1.0
Common Snapping Turtle	1997	PL55	n.a.	n.a.	n.a.	n.a.

Table 5.26. Comparison of the efficacy of model predictions of ^{90}Sr activity concentrations for different types of freshwater herpetofauna.

Radionuclide	Type of Organisim	Number of predictions Z>3	Total n	[a] Efficacy Measure (%)
^{90}Sr	Bullfrogs	2	3	33%
^{90}Sr	Green frogs	2	3	33%
^{90}Sr	Painted turtles	2	2	0%
^{90}Sr	Common snapping turtles	n.a.	n.a.	n.a.
^{90}Sr	**ALL Herpetofauna**	**6**	**8**	**25%**

n.a. – not applicable.

[a] The Efficacy Measure represents the percentage of model predictions with Z-scores that fall below a value of 3 (which represents an under- or over-prediction relative to the predictions generated by all models).

5.4.1.6. Modelled-to-Measured Comparisons for Freshwater Mammals

Six models (EA R&D128, ERICA (CEH), ERICA (NRPA), RESRAD-BIOTA (NRPA), RESRAD-BIOTA (UK) and D-Max) estimated radionuclide concentrations in freshwater mammals, although RESRAD-BIOTA (NRPA) only made predictions for American water shrews. With the exception of ^{90}Sr, it was not possible to compare activity concentration Z-scores for predicted activity concentrations in freshwater mammals due to the limited number of data and corresponding model predictions.

Cobalt-60

In general, the six models over-estimated measured values for ^{60}Co in star-nosed moles by up to more than four orders of magnitude (Figure 5.14). This was probably a consequence of assuming the mammals spend 100 % of their time in the aquatic environment and there is typically less understanding of radionuclide transfer to aquatic mammalian species. However, three models (ERICA (CEH), ERICA (NRPA) and RESRAD-BIOTA (UK)) under-predicted by more than an order of magnitude of measured ^{60}Co activity concentrations for American water shrews. The default ERICA Tool CR value used to generate the ERICA (CEH) prediction was that identified for benthic fish given the lack of data specifically for aquatic mammals.

In addition, although an allometric model was applied by RESRAD-BIOTA (UK), which accounted for expected differences in uptake and loss rates based on body size, predictions for these two mammalian species compared to the available data varied by close to two orders of magnitude, suggesting that factors, such as habitat use and dietary intakes (contamination and/or dry matter quantities) are not being adequately modelled.

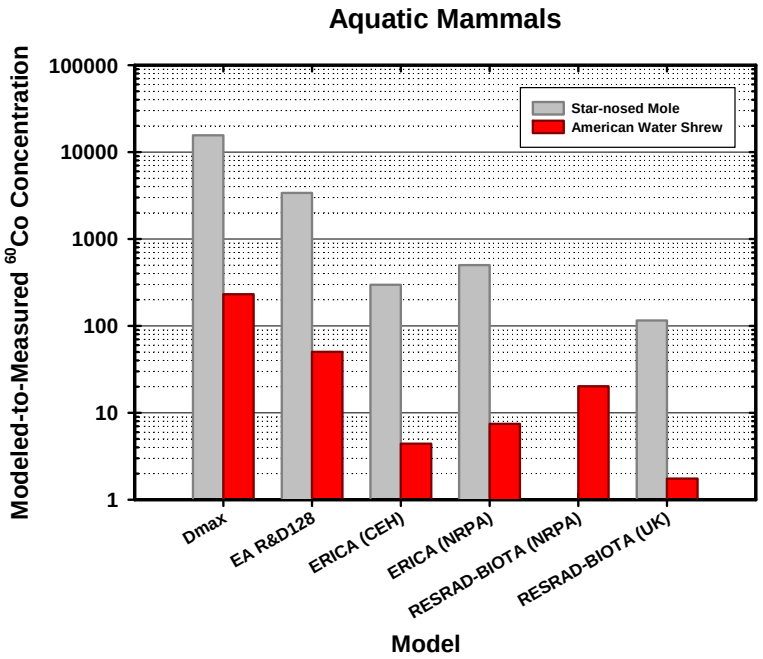

Fig. 5.14. Comparison of modelled-to-measured ^{60}Co concentrations in perch lake freshwater mammals. Predicted values could not be compared to minimum and maximum measured values in the lake for a given species due to the lack of historical data and the relatively small sample sizes for such species.

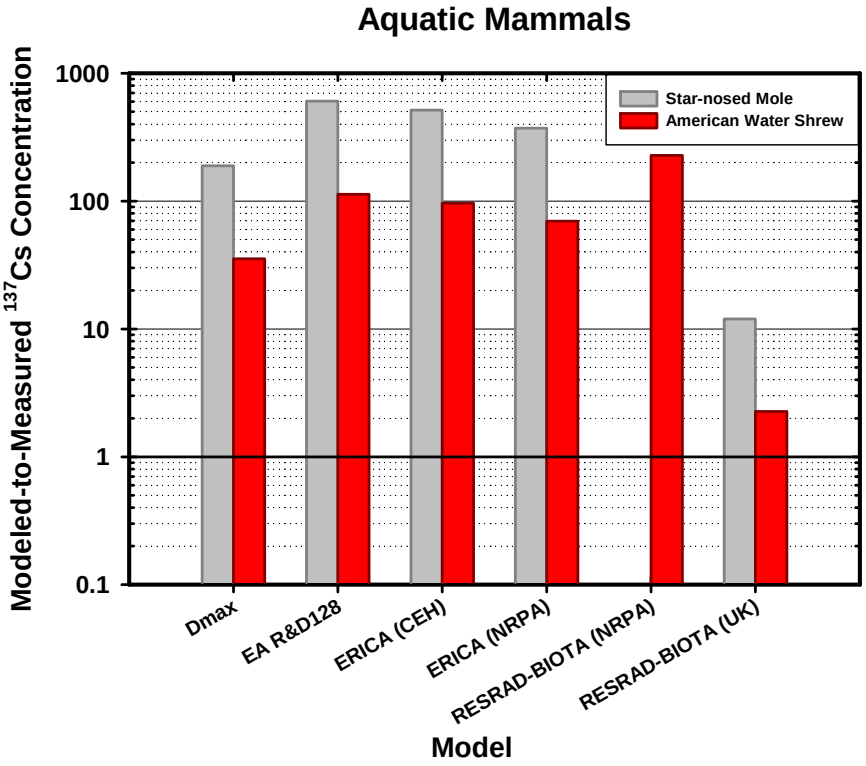

Fig. 5.15. Comparison of modelled-to-measured ^{137}Cs concentrations in Perch Lake freshwater mammalian species. Predicted values could not be compared to minimum and maximum measured values in the lake for a given species due to the lack of historical data and the relatively small sample sizes for such species.

Caesium-137

All models tended to over-predict the ^{137}Cs activity concentrations in freshwater mammals (Figure 5.15), although there was less difference in agreement between the species than observed for ^{60}Co and ^{90}Sr (Figure 5.16). There was also less inter-model variability in predictions for a given mammalian species. The least over-prediction was seen for RESRAD-BIOTA (UK), which applied an allometric approach.

Strontium-90

As for the other radionuclides considered, ^{90}Sr activity concentrations in star-nosed moles tended to be over-predicted by an order of magnitude or more (Figure 5.16). However, most models predicted ^{90}Sr in the American water shrews to be similar to the observed data The notable exception being the ERICA (CEH)-generated value, which was under-predicted by just over one-order of magnitude (Figure 5.16); this prediction was generated assuming the CR value for fish in the absence of a CR value for aquatic mammals.

In support of these observations, Z-scores for mammalian ^{90}Sr activity concentration predictions indicated that in all cases, those that had been generated for star-nosed moles were much higher than 3, whereas Z-scores for water shrews were lower (Table 5.27). Estimated efficacies for predictions of ^{90}Sr activity concentrations in star-nosed moles and American water shrews were 0 and 60%, respectively.

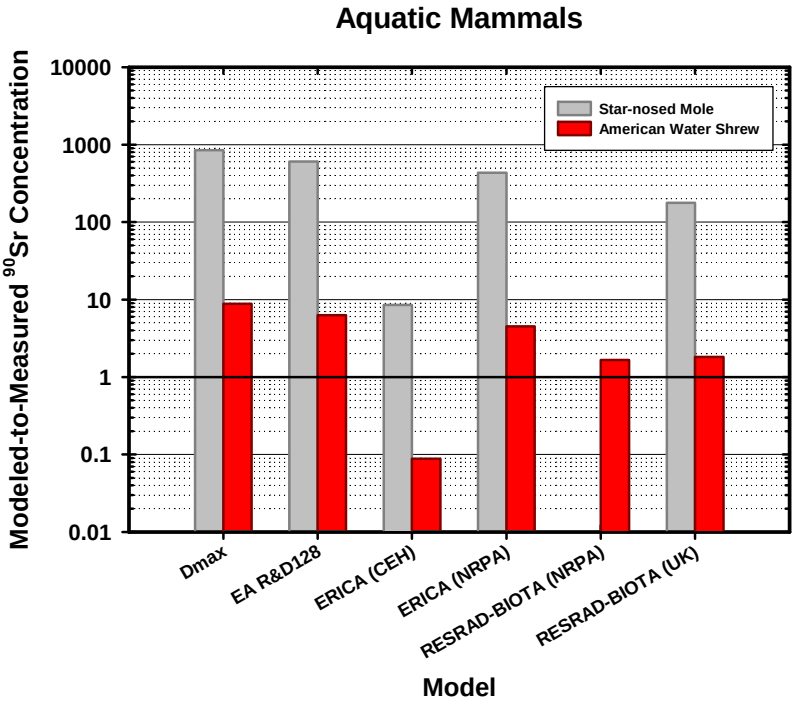

Fig. 5.16. Comparison of modelled-to-measured ^{90}Sr concentrations in Perch Lake freshwater mammalian species. Predicted values could not be compared to minimum and maximum measured values in the lake for a given species due to the lack of historical data and the relatively small sample sizes for such species.

Table 5.27. Summary of Z-scores for the prediction of ^{90}Sr activity concentrations in different types of freshwater mammals. Values in shaded cells represent Z-scores that are greater than 3.

Type of Organism	Year	Prediction No.	Z-scores for ^{90}Sr Concentration Data (by Organism Type)					
			D-Max	EA R&D128	ERICA (CEH)	ERICA (NRPA)	RESRAD-BIOTA (NRPA)	RESRAD-BIOTA (UK)
Star-nosed mole	1997	PL56	3242	2308	28.6	1654	n.a.	710
American Water Shrew	1997	PL57	6.0	4.0	0.7	2.7	0.5	0.6

5.4.2. Comment on interpretation of Z-scores

Z-scores represent a potentially useful tool to assess the dispersion of the model dataset relative to the measured dataset. However, if the standard deviation is sufficiently large (33% of the mean or more), an extreme under-prediction will record a Z-score of less than 3; hence over-predictions are better identified than under-predictions. An illustration of this can be seen in the results for ^{60}Co activity concentrations in fish (see Figure 5.8 and Table 5.17). CASTEAUR under-predicts concentrations in cyprinids by more than an order of magnitude (and below the observed data range), yet the Z-scores are below 3. Conversely, a high Z-score can be obtained if the measured dataset has a comparatively low standard deviation either because there is low inherent variability for a given parameter or because few data are available.

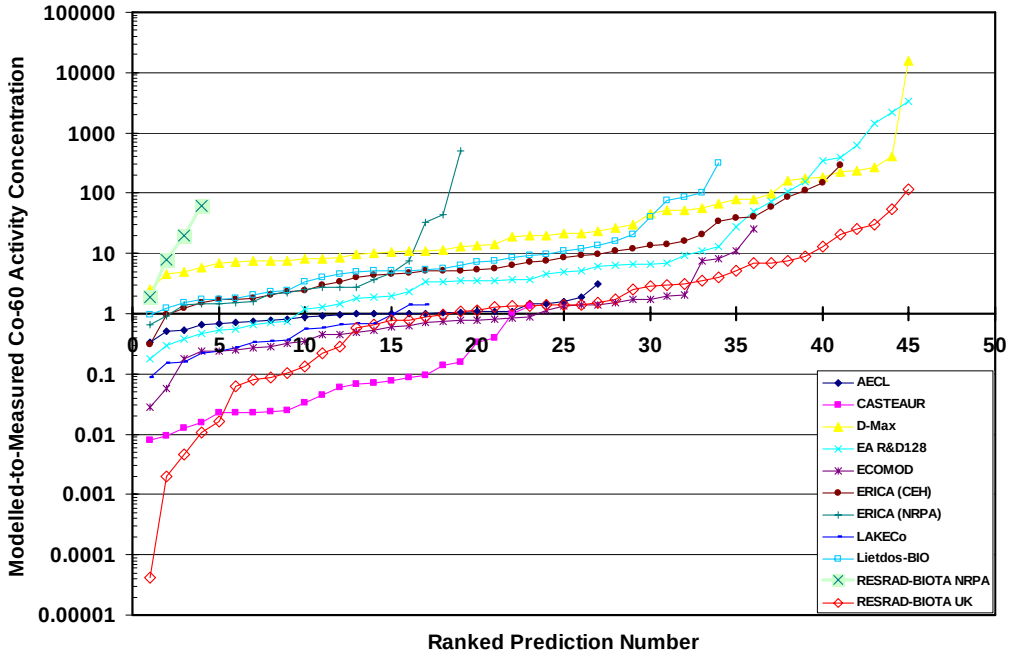

Fig. 5.17. Plots depicting modelled-to-measured ^{60}Co activity concentrations that were generated by each approach, where ratios have been ranked and sorted from smallest to largest. A modelled-to-measured ratio of one is desirable.

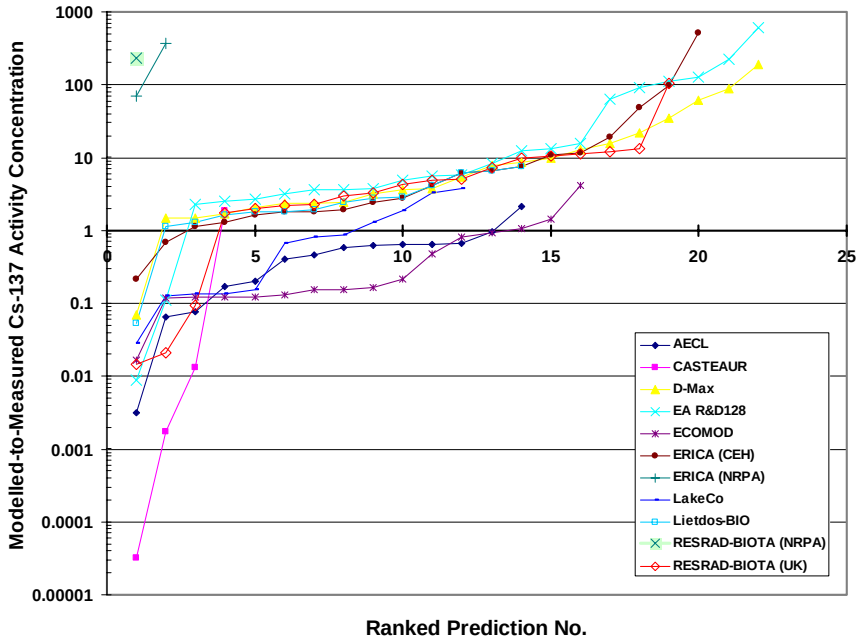

Fig. 5.18. Plots depicting modelled-to-measured ^{137}Cs activity concentrations that were generated by each approach, where ratios have been ranked and sorted from smallest to largest.

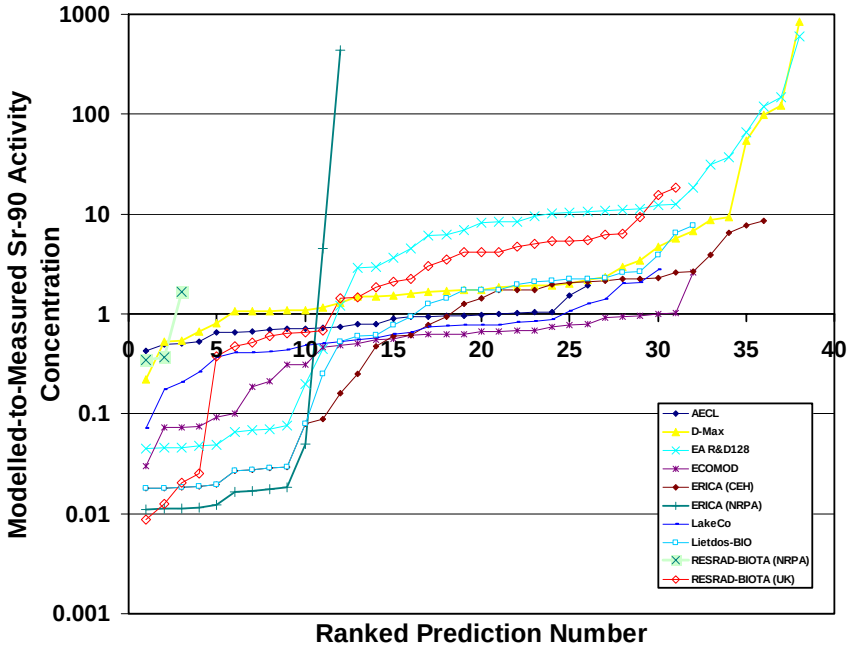

Fig. 5.19. Plots depicting modelled-to-measured ^{90}Sr activity concentrations that were generated by each approach, where ratios have been ranked and sorted from smallest to largest.

5.4.3. Overview of inter-model comparisons

For the purposes of the above comparisons of predicted and measured activity concentration values, focus was placed on models that either exceed or under-predict by an order of magnitude or more. To give an overview of the model predictions, Figures 5.17 to 5.19 present 'snake plots' depicting normalised values by model in increasing order for ^{60}Co, ^{137}Cs and ^{90}Sr; ^{3}H is not presented as predictions were largely close to the measured data (see Figure 5.1).

5.4.4. Doses to biota

Comparison of model-to-model predictions using Z-scores was undertaken to determine how consistently the various approaches were estimating internal and external unweighted dose rates from ^{60}Co, ^{137}Cs, ^{90}Sr and tritium (in the form of HTO).

From the results of Shapiro-Wilk tests, it was concluded that the raw data of estimated doses rates were neither normally- or log-normally distributed (Table 5.28). Hence, for the purposes of consistency with the concentration data, it was decided to treat both datasets as normal, since there was no clear advantage to assuming log-normal distributions. To further ensure consistency with the DCC evaluation described in Chapter 3, Z-scoring of Perch Lake dose data was carried out using a reference mean and standard deviation for the Z-score that was calculated using a robust data set after exclusion of outliers (as described in Section 3.3.3 above).

5.4.5. Internal dose rates

From Chapter 3 it is clear that the models produce similar internal unweighted dose rates assuming the same whole-body activity concentration. Variations in estimated internal dose rates in this exercise are therefore predominantly the consequence of the differing whole-body activity concentrations predicted by the different models. Few estimated Z-scores (which are not presented for internal dose rate estimates) were in excess of a value of 3. This is probably explained by the high variability amongst reported estimates (due to the variation in predicted activity concentrations). As a consequence, the standard deviation is much larger than the mean, and hence, low estimates of Z-scores are achieved.

Table 5.28. P-values for the Shapiro-Wilk Test for log-transformed versus raw data for radionuclide doses to Perch Lake biota.

Descriptionof Data	^{60}Co	^{137}Cs	^{90}Sr	Tritium (HTO)
Raw Data (untransformed)	< 2.2e-16	< 2.2e-16	< 2.2e-16	9.61E-06
Log-transformed Data	7.97E-05	0.00043	2.78E-07	2.98E-09

Table 5.29. Summary of Z-scores (with outliers removed) for the prediction of external dose from ^{90}Sr to different types of Perch Lake biota.

Species	Year	Prediction No.	Z-scores (outliers removed) for External Dose from ^{90}Sr (by Organism Type)						
			CASTEAUR +EDEN	EA R&D128	ERICA (CEH)	ERICA+Doses3D (NRPA)	LAKECO-B	LIETDOS-BIO	RESRAD-BIOTA (NRPA)
Primary Producers:									
Emergent	1996	PL48	0.80	1.12	0.33	n.a	n.a	1.05	n.a
Free-floating	1968	PL1	1.64	1.93	0.35	n.a	n.a	2.66	n.a
Floating-leafed	1968	PL3	0.50	1.38	0.24	n.a	n.a	0.78	n.a
Submergent	1996	PL46	0.33	1.35	0.38	n.a	n.a	0.47	n.a
Freshwater Invertebrates:									
Zooplankton	1994	PL34	0.23	0.19	0.22	n.a	n.a	n.a	n.a
Macroinvertebrates	1970	PL16	0.11	1.28	0.75	n.a	n.a	n.a	n.a
Freshwater Mussels	1969	PL9	0.32	1.41	0.45	n.a	n.a	0.9	n.a
Freshwater Fish:									
Brown bullheads	1995	PL44	0.71	0.64	0.06	0.28	n.a	0.57	0.79
Cyprinids	1970	PL18	0.46	1.31	1.12	0.03	0.39	0.08	n.a
Pumpkinseeds	1997	PL53	0.14	0.26	0.19	0.50	n.a	0.53	0.26
Amphibians:									
Bullfrogs	1971	PL22	1.10	1.54	0.20	n.a	n.a	n.a	n.a
Bullfrogs	1998	PL59	1.08	1.53	0.19	n.a	n.a	n.a	n.a
Green frogs	1998	PL58	0.28	1.16	0.24	n.a	n.a	n.a	n.a
Reptiles:									
Common snapping turtles	1997	PL55	0.83	1.26	n.a	n.a	n.a	n.a.	n.a.
Painted turtles	1997	PL54	0.76	1.19	n.a	n.a	n.a	n.a.	n.a.
Freshwater Mammals:									
Star-nosed moles	1997	PL56	0.27	1.02	0.57	0.68	n.a	n.a	n.a
American water shrews	1997	PL57	1.33	1.28	0.50	0.89	n.a.	n.a.	0.74

5.4.6. External dose rates

Overall, Z-scores for the external exposure from all radionuclides that have been considered as part of this scenario fall below a value of three (Tables 5.30–5.33). In addition, the distribution of the normalized values is tight, which is indicative of consensus between modelling approaches, and the assumptions made in applying the models to the scenario, in terms of how to evaluate this parameter.

In the case of external doses for tritium, extremely low values were reported with relatively high variability between the approaches; this is consistent with the findings described in Chapter 3.

Table 5.30. Summary of Z-scores (with outliers removed) for the prediction of external dose from ^{60}Co to different types of Perch Lake biota.

Type of Organism	Year	Prediction No.	Z-scores (outliers removed) for External Dose from ^{60}Co (by Organism Type)						
			CASTEAUR +EDEN	EA R&D128	ERICA (CEH)	ERICA+Doses3D (NRPA)	LAKECO-B	LIETDOS	RESRAD-BIOTA (NRPA)
Primary Producers:									
Emergent	1994	PL33	0.7	0.8	0.33	n.a.	n.a.	0.4	n.a.
Free-floating	1995	PL39	0.29	n.a.	1.1	n.a.	n.a.	0.5	n.a.
Floating-leafed	1968	PL3	1.6	n.a.	0.9	n.a.	n.a.	0.15	n.a.
Submergent	1968	PL2	0.6	n.a.	1.3	n.a.	n.a.	0.03	n.a.
Freshwater Invertebrates:									
Zooplankton	1994	PL34	0.2	0.4	0.4	n.a.	n.a.	n.a.	n.a.
Macroinvertebrates	1994	PL35	1.0	0.04	0.1	n.a.	n.a.	n.a.	n.a.
Snails	1970	PL17	1.7	n.a.	0.2	0.7	n.a.	n.a.	0.7
Freshwater Fish:									
Brown bullheads	1995	PL44	0.8	0.6	0.4	0.7	0.9	0.06	0.7
Cyprinids	1994	PL36	1.0	1.0	0.9	1.2	2.4	0.4	n.a.
Pumpkinseeds	1969	PL10	0.5	1.2	0.6	0.8	1.2	0.2	n.a.
Yellow perch	1969	PL12	n.a.	n.a.	0.5	1.2	0.8	0.6	n.a.
Amphibians:									
Bullfrogs	1998	PL59	n.a.	1.2	2.3	n.a.	n.a.	n.a.	n.a.
Green Frogs	1998	PL58	n.a.	1.2	2.3	n.a.	n.a.	n.a.	n.a.
Reptiles:									
Common Snapping Turtle	1997	PL55	0.8	0.1	n.a.	n.a.	n.a.	n.a.	n.a.
Painted turtle	1997	PL54	0.9	0.2	n.a.	n.a.	n.a.	n.a.	n.a.
Aquatic Mammals:									
American Water Shrew	1997	PL57	n.a.	0.34	2.5	0.8	n.a.	n.a.	0.8
Star-nosed mole	1997	PL56	n.a.	0.4	2.4	0.9	n.a.	n.a.	n.a.

Table 5.31. Summary of Z-scores (with outliers removed) for the prediction of external dose from ^{137}Cs to different types of Perch Lake biota.

Type of Organism	Year	Prediction No.	Z-scores (outliers removed) for External Dose from ^{137}Cs (by Organism Type)					
			CASTEAUR +EDEN	EA R&D128	ERICA (CEH)	ERICA+Doses3D (NRPA)	LIETDOS-BIO	RESRAD-BIOTA (NRPA)
Primary Producers:								
Emergent	1995	PL42	0.91	1.07	0.78	n.a.	2.04	n.a.
Free-floating	1994	PL31	0.33	n.a.	0.32	n.a.	0.64	n.a.
Floating-leafed	1994	PL32	1.17	n.a.	1.42	n.a.	1.15	n.a.
Submergent	1994	PL33	0.41	0.47	0.28	n.a.	0.44	n.a.
Freshwater Invertebrates:								
Zooplankton	1994	PL34	0.09	0.21	0.23	n.a.	n.a.	n.a.
Macroinvertebrates	1994	PL35	0.88	0.37	0.03	n.a.	n.a.	n.a.
Amphibians:								
Bullfrogs	1998	PL59	1.33	1.29	2.08	n.a.	n.a.	n.a.
Green frogs	1998	PL58	1.33	1.30	2.09	n.a.	n.a.	n.a.
Reptiles:								
Common snapping turtles	1997	PL55	0.23	0.13	n.a.	n.a.	n.a.	n.a.
Painted turtles	1997	PL54	0.21	0.15	n.a.	n.a.	n.a.	n.a.
Aquatic Mammals:								
American water shrews	1997	PL57	1.34	n.a.	1.42	0.27	n.a.	0.25
Star-nosed moles	1997	PL56	1.37	0.47	1.40	0.28	n.a.	n.a.

5.5. Summary and conclusions

Comparison of predicted and measured radionuclide activity concentrations in aquatic primary producers, invertebrates, fish, frogs, turtles and aquatic mammalian species highlighted a number of areas where additional work and understanding is required with respect to predicting radionuclide transfer to freshwater receptor biota. In particular, model predictive power was relatively poor for freshwater mammalian species and herpetofauna. For example, radionuclide concentrations in herpetofauna and freshwater mammals were generally over-predicted by orders of magnitude. This may largely be due to assuming that the organisms spent 100% of their time in the aquatic environment.

A relatively large inter-model range also occurred for predictions of radionuclide activity concentrations in Perch Lake primary producers. This variability was probably because many of the modelling approaches tended to apply the same CRs regardless of the type of primary producer being considered, although it is well documented that even visibly similar species can vary by orders of magnitude in terms of their propensities to accumulate radionuclides (e.g. [30, 134]). This observation also applies to some of the other organisms considered (e.g. fish and mammals)

In a number of cases, ^{60}Co and ^{137}Cs concentrations were under-estimated in zooplankton, possibly due to an exposure pathway that had not been considered and/or the incompatibility in terms of the concentration ratios obtained by modellers as source data to estimate radionuclide bioaccumulation data relative to transfer processes that are occurring in Perch Lake.

Transfer of Cs can be driven by potassium concentrations in surface waters, with Cs CRs tending to be lower in water bodies with relatively higher K levels and *vice-versa* [83]. Perch Lake has a mean water K concentration of 0.91±0.030 6 mg l^{-1}, which falls on the lower end of the global range of 1.6±2.6 mg l^{-1}. Therefore, Cs transfer in Perch Lake may have been expected to be comparatively high, potentially leading to some under-predictions.

In all cases assessed as part of this scenario, laboratory-generated CRs (used by the CASTEAUR and to a lesser degree ECOMOD approaches) under-predicted radionuclide activity concentrations in freshwater biota.

In general, models that accounted for water chemistry (e.g. calcium concentrations in water when estimating ^{90}Sr uptake by fish) tended to produce better predictions than models that applied a generic, CR-based approach.

Perhaps not surprisingly, the application of site-specific CRs by the AECL approach that had been measured in Perch Lake (and that would incorporate site-specific conditions in the lake) generated predictions that were usually similar to measured values. The implicit objective of this approach was to determine whether or not radionuclide bioaccumulation factors that had been measured at a given time point in the lake could be used to estimate radionuclide concentrations in Perch Lake biota over time based on surface water measurements. The outcome of this exercise indicated that for the most part, predictions that were made based on site-specific data fell quite close to values that had been estimated based on the independent site-specific measurements that had been taken in the lake. In general, this confirms that concentrations of key radionuclides in the lake water are representative of levels that can be found in resident flora and fauna species, making it possible to routinely monitor water, as opposed to biota, to demonstrate compliance with respect to doses to Perch Lake biota. This will ultimately reduce the need to euthanize biota at the site, resulting in more environmentally-friendly (or less destructive) sampling approaches, while facilitating more cost-effective environmental monitoring.

This scenario differed to other exercises conducted by the BWG in that one of the tools, RESRAD-BIOTA, was applied by two organisations who were not the developers. The two applications of RESRAD-BIOTA produced results of up to four-orders of magnitude difference between the activity concentrations derived using the default aquatic BiVs (CRs) provided in the RESRAD-BIOTA code versus those derived using allometry. The BiV-derived activity concentrations (RESRAD-BIOTA (NRPA)) generally being the higher and 'more accurate' (with respect to measured values) of the two. However, some caution must be expressed with regard to the applications of RESRAD-BIOTA within this scenario, as: (i) the application of the allometric approach to some of the organisms considered here (e.g. fish and invertebrates) is outside the scope of that envisaged by the tool developers, although this is not highlighted to users; and (ii) default BiV values are provided for screening purposes only.

Two approaches (D-Max and EA R&D128) often over-predicted radionuclide activity concentrations by more than an order of magnitude in Perch Lake biota. The D-Max model aimed to be conservative often applying maximum CR values(as identified by its developer). Similarly, the EA R&D128 approach aims to be conservative, especially when data for a specific species are lacking and guidance methodology is used to derive a default value.

The results of the *Perch Lake Freshwater Scenario* confirmed that, in general, most of the variability in dose-to-biota estimates lies with the prediction of radionuclide transfer to biota.

CHAPTER 6. TERRESTRIAL SCENARIO: CHERNOBYL EXCLUSION ZONE

6.1. Scenario description

A database of radionuclide activity concentrations in a range of biota was compiled from the open literature [135–138] and data holdings of the International Radioecology Laboratory (IRL) (including those described in Gaschak et al. [136] and other BWG members (see Beresford et al. [139, 140). By preference soil activity concentrations were collated from the same reference sources or were provided for the sampling sites by IRL. If this was not possible soil concentrations were derived using deposition maps within a geographical information system (see Beresford et al. [139]). This scenario was developed from case studies previously used to test the predictions of the FASSET framework [139] and ERICA Tool (Beresford et al. ([140]).

Available data covered a range of biota types including: graminaceous vegetation; invertebrates; birds; a wide range of mammal species (from small rodents to deer and carnivorous species) and amphibians (see Table 6.1). The majority of collated data were for ^{137}Cs and ^{90}Sr, although some data are available for actinide isotopes in small mammals and birds. Results from thermoluminescent dosimeters (TLD) attached to species of small mammals were also available for five data entries [135, 140]. The majority of data selected for inclusion within the scenario were for multiple measurements (i.e. an observed mean and standard deviation were available or could be estimated). However, for a few data (predominantly for birds) only one measurement was available.

Participants were provided with a spreadsheet containing all available soil concentrations and requested to predict the appropriate activity concentrations in biota. Internal, external and total unweighted absorbed dose rates were also requested for a subset of the data. Where results from TLDs attached to small mammal species were available participants were requested to estimate the TLD reading. Table 6.1 summarises the predictions which were requested by species.

The scenario description provided to participants can be found in Appendix VI. The description provided limited data for water activity concentrations for use in predictions of amphibian species whole-body activity concentrations if required and useful websites from which information on animal behaviour could be acquired. Manipulations used to generate the media and biota data sets are also detailed within the description. A table of the required predictions for each data entry has been added to Appendix VI (Table VI.1).

6.2. Application of models to the scenario

Seven approaches participated in this scenario: RESRAD-BIOTA, ERICA, EA R&D128, LIETDOS-BIOTA, DosDiMEco, FASTer-EPIC Doses 3D and D-Max. Of the participating models RESRAD-BIOTA, ERICA and EA R&D128 reported a complete set of predictions. Specifics of application to this scenario are described below; Chapter 2 should be consulted for general model descriptions. As many of the models use CR values to estimate whole-body activity concentrations in biota those used in this scenario have been collated for comparison in Tables 6.2-6.4.

Table 6.1. Number of predictions requested for the Chernobyl Scenario summarised by species.

Species	Common name (English)	Number of predictions					
		^{90}Sr	^{137}Cs	Pu[1]	^{241}Am	Dose rate	TLD[2]
Aegithalos caudatus	Long-tailed tit	-	-	1	-	1	-
Apodemus flavicollis	Yellow necked mouse	5	5	1	-	2	2
Apodemus sylvaticus	Wood mouse	1	1	-	-	1	-
Canis lupus	Wolf	2	2	-	-	1	-
Capreolus capreolus	Roe deer	7	7	-	-	1	-
Clethrionomys glareolus	Bank vole	7	6	2	1	2	2
Erithacus rubecula	Robin	2	2	-	-	1	-
Hirundo rustica	Barn swallow	1	1	-	-	1	-
Lactera agilis	Sand Lizard	1	1	-	-	1	-
Microtus arvalis	Common vole	2	2	-	-	1	-
Microtus oeconomus	Root vole	2	3	-	-	1	1
Microtus spp.	Vole species	1	1	1	-	1	1
Parus major	Great tit	2	2	1	-	1	-
Perdix perdix	Partridge	-	2	-	-	1	-
Rana esculenta	Edible frog	-	2	-	-	1	-
Rana terrestris	Brown frog	2	4	-	-	1	-
Sicista betulina	Northern birch mouse	1	1	-	-	1	-
Sorex araneus	Common shrew	5	5	-	-	1	-
Sturnus vulgaris	Starling	1	1	-	-	1	-
Sus scofa	Wild boar	9	9	-	-	1	-
	Beetles	1	1	-	-	1	-
	Grassy vegetation	4	4	-	-	1	-

[1]Pu isotopes varied between data sources; [2]Participants were requested to estimate the dose rate recorded by a TLD attached to these animals.

Table 6.2. Strontium-90 CR (biota:soil) values used by the participating models for the Chernobyl Scenario.

Species	RESRAD-BIOTA	EA R&D128	ERICA	LETDOS-BIOTA	DosDimEco	D-Max
Apodemus flavicollis	n/a	5.00	1.74	1.25	n/a	10.0
Apodemus sylvaticus	n/a	5.00	1.74	1.25	n/a	20.0
Canis lupus	n/a	5.00	1.74	1.30	n/a	20.0
Capreolus capreolus	1.74	5.00	1.74	1.96	n/a	10.0
Clethrionomys glareolus	n/a	5.00	1.74	1.25	n/a	10.0
Erithacus rubecula	n/a	5.00	0.55	0.49	n/a	20.0
Hirundo rustica	n/a	5.00	0.55	0.49	n/a	20.0
Lactera agilis	n/a	5.00	11.8	47.0	n/a	10.0
Microtus arvalis	n/a	5.00	1.74	1.25	n/a	10.0
Microtus oeconomus	n/a	5.00	1.74	1.25	n/a	10.0
Microtus spp.	n/a	5.00	1.74	1.25	n/a	10.0
Parus major	n/a	5.00	0.55	0.49	n/a	20.0
Perdix perdix	n/a	5.00	0.55	0.49	n/a	20.0
Rana terrestris	n/a	5.00	0.83	n/r	n/a	n/r
Sicista betulina	n/a	5.00	1.74	1.25	n/a	20.0
Sorex araneus	n/a	5.00	1.74	1.25	n/a	20.0
Sturnus vulgaris	n/a	5.00	0.55	0.49	n/a	20.0
Sus scofa	n/a	5.00	1.74	4.80	n/a	20.0
Beetles	0.06	5.00	0.41	n/r	n/a	10.0
Grass vegetation	0.21	5.00	0.21	0.21	0.03	10.0

n/r – not reported by this model; n/a – predictions made by approaches other than CR values (see text for details); shaded cells denote CR values which are derived from guidance approaches for use when data is lacking for EA R&D128 (see text for details).

Table 6.3. Caesium-137 CR (biota:soil) values used by the participating models for the Chernobyl Scenario.

Species	RESRAD-BIOTA	EA R&D128	ERICA	LETDOS-BIOTA	DosDimEco	D-Max
Apodemus flavicollis	n/a	0.01	2.87	11.4	n/a	10.0
Apodemus sylvaticus	n/a	0.01	2.87	11.4	n/a	20.0
Canis lupus	n/a	9.00	2.87	4.96	n/a	20.0
Capreolus capreolus	2.87	2.20	2.87	1.84	n/a	10.0
Clethrionomys glareolus	n/a	0.01	2.87	11.4	n/a	10.0
Erithacus rubecula	n/a	1.60	0.75	0.76	n/a	20.0
Hirundo rustica	n/a	1.60	0.75	0.76	n/a	20.0
Lactera agilis	n/a	9.00	3.59	23.2	n/a	10.0
Microtus arvalis	n/a	0.01	2.87	11.4	n/a	10.0
Microtus oeconomus	n/a	0.01	2.87	11.4	n/a	10.0
Microtus spp.	n/a	0.01	2.87	11.4	n/a	10.0
Parus major	n/a	1.60	0.75	0.76	n/a	10.0
Perdix perdix	n/a	1.60	0.75	0.76	n/a	20.0
Rana esculenta	n/a	9.00	0.54	0.43	n/a	10700*
Rana terrestris	n/a	9.00	0.54	0.43	n/a	10700*
Sicista betulina	n/a	0.01	2.87	11.4	n/a	20.0
Sorex araneus	n/a	0.01	2.87	11.4	n/a	20.0
Sturnus vulgaris	n/a	1.60	0.75	0.76	n/a	20.0
Sus scofa	n/a	9.00	2.87	2.41	n/a	20.0
Beetles	0.06	0.04	0.13	n/r	n/a	10.0
Grass vegetation	0.69	0.14	0.69	0.69	0.04	10.0

*CR biota:water; n/r – not reported by this model; n/a – predictions made by approaches other than CR values (see text for details); shaded cells denote CR values which are derived from guidance approaches for use when data is lacking for EA R&D128 (see text for details).

Table 6.4. Pu and ^{241}Am CR (biota:soil) values used by the participating models for the Chernobyl Scenario; RESRAD-BIOTA and DOSIMECO did not use any CR values for these radionuclides.

	EA R&D128	ERICA	LETDOS-BIOTA	D-Max
Pu isotopes				
Aegithalos caudatus	0.70	2.34×10^{-2}	1.00×10^{-5}	0.01
Apodemus flavicollis	5.0×10^{-4}	2.34×10^{-2}	5.67×10^{-3}	0.01
Clethrionomys glareolus	5.0×10^{-4}	2.34×10^{-2}	5.67×10^{-3}	0.01
Microtus spp.	5.0×10^{-4}	2.34×10^{-2}	5.67×10^{-3}	0.01
Parus major	0.70	2.34×10^{-2}	1.00×10^{-5}	0.01
^{241}Am				
Clethrionomys glareolus	2.7×10^{-4}	4.08×10^{-2}	7.49×10^{-3}	0.01

Shaded cells denote CR values which are derived from guidance approaches for use when data is lacking for either EA R&D128 or ERICA (see text for details).

Table 6.5. Parameters used within the RESRAD-BIOTA code for the Chernobyle Scenario.

Organism	Mass (g)	Geometry number[1]	Life span (y)	Fraction of time		Food ingestion (g/d)	Inhalation of soil (g/d)	Soil ingestion (g/d)	Food source[2]
				in soil	on soil				
Grassy vegetation	n/a	2	n/a	1	0	n/a	n/a	n/a	n/a
Beetle	0.1	1, 2	n/a	0	1	n/a	n/a	n/a	n/a
Long-tailed tit	8.5	3	2	0	0.5	2.61	1.00E-06	0.26	100% fi
Great-tit	18	3	3	0	0.5	4.25	1.90E-06	0.43	10% di, 10% si, 30% fi, 25% g&h, 25% t
Robin	19	3	2	0	0.5	4.41	1.90E-06	0.44	10% di, 10% si, 30% fi, 25% g&h, 25% t
Starling	75	3, 4	5	0	0.5	10.77	5.60E-06	1.08	5% di, 5% si, 20% fi, 35% g&h, 35% t
Barn swallow	19	3	3	0	0.5	4.41	1.90E-06	0.44	100% fi
Partridge	395	3, 4	3	0	0.5	31.77	2.00E-05	3.18	86% g&h, 14% fi
Sand lizard	12	3	3	0.75	0.25	2.31	1.67E-06	0.23	10% di, 30% si, 20% fi, 20% gas, 20% t
Edible frog[3]	47	3, 4	5	0	0.19	6.42	4.71E-06	0.64	10% si, 10% gas, 50% fi, 15% c, 15% m
Brown frog[4]	22.7	3, 4	5	0	0.58	3.72	2.71E-06	0.37	20% si, 30% gas, 50% fli
Mice species	30	3	1	0.5	0.5	6.3	2.50E-06	0.63	38% fi, 52% g&h, 10% s
Common shrew	10	3	1	0.75	0.25	4.9	2.30E-06	0.10	30% di, 20% gas 25% si, 20% g&h, and 5% fi
Root and Common vole	50	3	0.415	0.75	0.25	6.5	4.30E-06	0.65	100% g&h
Bank vole and *Microtus* spp.	23	3	1	0.75	0.25	7.4	5.20E-06	0.74	97% g&h, 3% si
Roe deer	n/a	5, 6	n/a	n/a	1	n/a	n/a	n/a	n/a
Wolf	41000	5, 6	7	0	1	1030	8.09E-04	0	90% deer, 10% mouse
Wild boar	250000	6, 7	10	0	1	4000	3.20E-03	400	86% s, 12% si, 2% di

n/a – not applicable, allometric approaches not used for these organisms.

[1]Where two numbers presented first is external geometry and second internal (see Table 6.6 for dimensions); [2]Food sources: fi – flying insects, di – detritivorous invertebrates, si – soil invertebrates, g&h – grass and herbs, t – tree, s – shrub, gas – gastropod, c – freshwater crustacean, m – freshwater mollusc; [3]Assumed to spend 0.39 and 0.42 of time in water and at the water-sediment interface respectively with a water ingestion rate of 6.32 g d^{-1}; [4]Assumed to spend 0.42 of time at the water-sediment interface and a water ingestion rate of 3.28 g d^{-1}.

Table 6.6. Dimensions of the default RESRAD-BIOTA default geometries used in the Chernobyl Scenario.

Default geometry number	Dimensions (cm)
1	0.2×0.2×0.2
2	2.5×1.2×0.6
3	10×2×2
4	45×8.7×4.9
5	50×26×13
6	100×42×33
7	270×66×48

6.2.1. RESRAD-BIOTA

The parameters used to apply RESRAD-BIOTA to the scenario are presented in Table 6.5. Predictions were made for the assumed maximum life-span of each animal. To estimate the radionuclide activity concentration in animal food sources default CR values from the ERICA Tool were used for a variety of terrestrial invertebrate and plant reference organisms. For green frog (for which water concentrations were presented in the scenario) freshwater mollusc and crustacean activity concentrations were estimated using CR values from the ERICA Tool. Results reported for grassy vegetation, beetles (assuming flying insect CR) and roe deer were also estimated using CR values from the ERICA Tool. The CR values used from the ERICA Tool were those contained within the first full release version of the ERICA Tool (April 2007). The activity concentration of the diet of wolf was estimated from the calculated activity concentrations in deer and mice. Ingestion of contaminated water was assumed for Edible and Brown frogs at rates of 6.32 and 3.28 g d^{-1} respectively.

Table 6.6 presents dimensions for those default geometries used from RESRAD-BIOTA in this scenario.

6.2.2. ERICA

The DCCs and CR values applied to the Chernobyl scenario were those contained within the first full release version of the ERICA Tool (April 2007) and as documented by Ulanovsky et al. [25] and Beresford et al. [23] respectively.

Default CR values, derived from literature review [23], were available for reference organisms appropriate to the species being considered with the exception of ^{241}Am and Pu CR values for birds (see Tables 6.2-6.4). Whole-body ^{241}Am and Pu activity concentrations were estimated assuming the same CR values as for mammals (the method used to derive the default bird CR values for the two radionuclides as presented within the ERICA Tool).

Dose conversion coefficients provided within the ERICA Tool for default reference organisms were used (i.e. species specific DCCs were not generated). The reference organism geometry and assumptions made concerning occupancy factors are presented in Table 6.7 (information to derive occupancy factors was obtained from the websites suggested within the scenario description (Appendix VI). To estimate dose rates to wolves underground it was necessary to generate underground DCC values using the ERICA Tool's <add organism> function. However, this function does not allow within soil DCCs for animals the size of a wolf to be generated therefore, the maximum allowed (within soil) organism mass of 6.6 kg was assumed.

Table 6.7. Occupancy assumptions and default reference organism geometries used for application of the ERICA Tool to the Chernobyl Scenarion.

Species	Reference organism geometry	Occupancy assumption (fraction of time)				
		in air	on soil	in soil	in water	in sediment
Aegithalos caudatus	Bird	0.2	0.8			
Apodemus flavicollis	Mammal (rat)		0.5	0.5		
Apodemus sylvaticus	Mammal (rat)		0.5	0.5		
Canis lupus	Mammal (deer)[1]		0.1	0.9		
Capreolus capreolus	Mammal (deer)		1			
Clethrionomys glareolus	Mammal (rat)		0.3	0.7		
Erithacus rubecula	Bird	0.5	0.5			
Hirundo rustica	Bird	0.65	0.35			
Lactera agilis	Reptile		0.6	0.4		
Microtus arvalis	Mammal (rat)		0.3	0.7		
Microtus oeconomus	Mammal (rat)		0.3	0.7		
Microtus spp.	Mammal (rat)		0.3	0.7		
Parus major	Bird	0.2	0.8			
Perdix perdix	Bird	0.1	0.9			
Rana esculenta	Amphibian		0.35		0.23	0.42
Rana terrestris[2]	Amphibian		0.54	0.46		
Sicista betulina	Mammal (rat)		1			
Sorex araneus	Mammal (rat)		0.9	0.1		
Sturnus vulgaris	Bird	0.7	0.3			
Sus scofa	Mammal (deer)		1			
Beetles	Detritivorous invertebrate		0.5	0.5		
Grass vegetation	Grass and Herb		1			

[1]An organism geometry was created to model underground dose rates (see text); [2]Water concentrations not available for the *R. terrestris* sample for which dose rates were required to be reported.

For four of the required frog predictions ^{137}Cs water activity concentrations were presented in the scenario description in addition to those for soil (see Appendix VI). Whole-body ^{137}Cs activity concentrations of 4150 Bq kg^{-1} (fw) and 35600 Bq kg^{-1} (fw) were estimated for the frogs at these two sites using a terrestrial ecosystem CR value and soil activity concentrations (N.B. the same activity concentrations were estimated for both species as the same CR value was applied). If the aquatic CR and water activity concentrations were used, whole-body activity concentrations of 3440 Bq kg^{-1} and 130000 Bq kg^{-1} respectively were predicted; for consistency the values estimated from soil activity concentrations were reported for analyses within the scenario. Dose rates were estimated assuming both frog species spent 23% of their time in water, 42% in sediment and 35% on soil which was identified as being typical for males (females spend less time in the aquatic environment and estimated dose rates were 4 to 16 times lower than those for males depending upon site).

6.2.3. EA R&D128

The DCCs and CR values applied to the Chernobyl scenario were those contained within the freshwater (v1.15) and terrestrial (v1.20) spreadsheets released in 2003 [6].

The default CR values were derived from literature review (with a bias towards data collected in the UK) and using a guidance derived approach to fill in gaps where no CR values exist for particular biota/radionuclide combinations (see Tables 6.2–6.4). The guidance used to fill in the data gaps is described in Copplestone et al [6]. For ^{90}Sr a CR value of 5 was used for all biota. This value was derived from the rodent reference organism geometry based on one set

of measurements of mice collected from Lady Wood near Sellafield, Cumbria, UK (unpublished data). For ^{137}Cs a CR value of 9.00 was used for the sand lizard, edible frog, and brown frog and this value was derived from the carnivorous mammal reference organism. The carnivorous mammal CR value, and also that used for birds (1.60), were derived from Lowe and Horrill [78] from measurements of samples collected soon after the fallout of the Chernobyl accident in the UK and this may have influenced the CR values.

Dose conversion coefficients provided within the R&D128 spreadsheets for default reference organisms were used (i.e. species specific DCCs were not generated). The reference organism geometry and assumptions made concerning occupancy factors are presented in Table 6.8.

6.2.4. LIETDOS-BIO

Dose conversion coefficients were estimated using the MCNPX software described above. Appropriate geometries were selected from documentation for the FASSET approach and the occupancy assumptions used are shown in Table 6.9.

Estimates of whole-body activity concentrations were made using CR values (see Tables 6.2–6.4) taken from FASSET, ERICA (grassy vegetation only), Macgee et al. [141] or derived from data obtained in Lithuania after the Chernobyl accident [142].

Table 6.8. Occupancy assumptions and default reference organism geometries used for application of the R&D128 to the Chernobyl Scenario.

Species	Reference organism geometry	Occupancy assumption (fraction of time)		
		in air	on soil	in soil
Aegithalos caudatus	Bird	0.5	0.5	
Apodemus flavicollis	Rodent		0.4	0.6
Apodemus sylvaticus	Rodent		0.4	0.6
Canis lupus	Carnivorous Mammal		0.6	0.4
Capreolus capreolus	Herbivorous Mammal		0.5	0.5
Clethrionomys glareolus	Rodent		0.6	0.4
Erithacus rubecula	Bird	0.5	0.5	
Hirundo rustica	Bird	0.5	0.5	
Lactera agilis	Reptile	0.1	0.4	0.5
Microtus arvalis	Rodent		0.4	0.6
Microtus oeconomus	Rodent		0.4	0.6
Microtus spp.	Rodent		0.4	0.6
Parus major	Bird	0.5	0.5	
Perdix perdix	Bird	0.5	0.5	
Rana esculenta[1]	Reptile	0.1	0.4	0.5
Rana terrestris[2]	Reptile	0.1	0.4	0.5
Sicista betulina	Rodent		0.4	0.6
Sorex araneus	Rodent		0.4	0.6
Sturnus vulgaris	Bird	0.5	0.5	
Sus scofa	Carnivorous Mammal		0.6	0.4
Beetles	Woodlouse		1.0	
Grass vegetation	Herb	0.5		1.0

[1]a freshwater assessment was also conducted using 0.4 in sediment, 0.3 sediment/surface and 0.3 in water and then the terrestrial and freshwater assessment results were combined assuming that an occupancy of 40% in terrestrial and 60% in the aquatic environment; [2]Water concentrations not available for the *R. terrestris* sample for which dose rates were required to be reported.

Table 6.9. Occupancy assumptions used in the application of LIETDOS-BIO to the Chernobyl Scenario.

Species		Occupancy assumption (fraction of time)		
		In air	On soil	In soil
Apodemus flavicollis	Yellow necked mouse	-	0.5	0.5
Apodemus sylvaticus	Wood mouse	-	0.5	0.5
Capreolus capreolus	Roe deer	-	1.0	-
Clethrionomys glareolus	Bank vole	-	0.5	0.5
Erithacus rubecula	Robin	0.5	0.5	-
Hirundo rustica	Barn swallow	0.5	0.5	-
Microtus arvalis	Common vole	-	0.5	0.5
Microtus oeconomus	Root vole	-	0.5	0.5
Microtus spp.	Vole species	-	0.5	0.5
Parus major	Great tit	0.3	0.7	-
Perdix perdix	Partridge	-	1.0	-
Rana terrestris	Brown frog	-	0.5	0.5
Sicista betulina	Northern birch mouse	-	1.0	-
Sorex araneus	Common shrew	-	1.0	-
Sturnus vulgaris	Starling	0.5	0.5	-

Table 6.10. Parameters used within the DOSDIMECO code for the Chernobyl Scenario.

Organism	Reference organism for DCC[1]	Reference organism for CR[2]	Occupancy (%)					Food ingestion (g d^{-1})	Body weight (g)
			Air	On Soil	In Under-ground	In air/ trees	On Water		
Bank vole	Mouse	Rodent	0	50	50	0	0	4.0	0.027
Barn swallow	Mouse	Bird egg	50	0	0	50	0	3.6	0.022
Common shrew	Mouse	Rodent	0	35	65	0	0	2.1	0.0085
Common vole	Mouse	Rodent	0	50	50	0	0	4.2	0.0295
Great tit	Mouse	Bird egg	25	25	0	50	0	4.9	0.0194
Long-tailed tit	Mouse	Bird egg	25	25	0	50	0	2.7	0.0093
Northern birch mouse	Mouse	Rodent	0	25	50	25	0	2.0	0.008
Partridge	Rat	Bird egg	50	50	0	0	0	32.6	0.41
Robin	Mouse	Bird egg	25	25	0	50	0	4.9	0.019
Roe deer	Deer	Herbivorous mammal	0	100	0	0	0	1160	35
Root vole	Mouse	Rodent	0	50	50	0	0	5.3	0.045
Starling	Mouse	Bird egg	25	25	0	50	0	16.9	0.0825
Vole species	Mouse	Rodent	0	50	50	0	0	4.8	0.0373
Wild boar	Wolf	Herbivorous mammal	0	100	0	0	0	2320	90.5
Wolf	Wolf	Carnivorous mammal	0	100	0	0	0	2520	80
Wood mouse	Mouse	Rodent	0	50	50	0	0	4.0	0.027
Yellow necked mouse	Mouse	Rodent	0	25	50	25	0	4.0	0.0275

[1] Reference organisms were taken from which the geometry relates closely to the Chernobyl species.
[2] Reference organisms were taken from which the life span and body mass relates closely to the Chernobyl species.

Table 6.11. Dimensions of the default DOSIMECO default geometries used in the Chernobyl Scenario.

Default geometry	Rat	Mouse	Deer	Wolf
a (cm)	20	5	70	80
b (cm)	6	1.6	22.5	20
c (cm)	5	1.6	22.5	20
Mass (g)	310	25	35000	80000
Area (cm^2)	250	110	22000	22000

6.2.5. DosDiMEco

The parameters used to apply DosDiMEco to the Chernobyl scenario are presented in Table 6.10. To estimate the radionuclide activity concentration in animal food sources CR values derived during the work described in Chapter 4 for a variety of terrestrial and plant reference organisms were used (see also Appendix IV). For the different species in the Chernobyl scenario, corrected CR values for the reference organisms (from Section 4.2.4; Appendix IV) were used. The different bodyweights (from internet searches) and the ingestion rates (from [41]) were derived and calculated to adapt the CR (using the ratio between reference organism and the Chernobyl species). Habitat assumptions used are presented in Table 6.10. Because the geometry dimensions of some Chernobyl species were not in the range of the geometry of the reference organisms considered in Chapter 3, some DCCs had to be recalculated. Table 6.11 presents the dimensions for the default geometries used for DosDiMEco in this scenario.

6.2.6. FASTer-EPIC DOSES3D

The FASTer transfer and EPIC DOSES3D dosimetry models were used in combination for this scenario.

6.2.6.1. Dosimetry

Table 6.12 presents the list of organisms for which DCC calculations were made. The assumed tissue density for birds was 0.8 g cm^{-3} and that of the remaining species considered was 1.0 g cm^{-3}. The dimensions of organisms have been back-calculated from their mass and density using the length of the organism from the literature (as this was normally the only reported dimension available) combined with an estimate of the relative width from representative photographs.

Table 6.12 also presents the assumed occupancy factors. The time a Brown frog spends in water or the terrestrial environment depends on its sex: the literature suggests that males spend 40% of their time in water and 60% on land, whereas females spend 10% of their time in water and 90% on land. As an average value for both sexes the following occupancy factors were assumed: 0.25 in water and 0.75 on soil; the same assumptions were made for Edible frog as no species specific information could be identified. Except for the size and mass it was assumed that other parameters are the same for both frog species.

6.2.6.2. Transfer

The input data used for the FASTer model runs were selected to be the mean activity concentrations in soil reported for the scenario.

In the original FASTer model (see Brown et al. [26]), a correction factor was applied in situations where biological half-lives were equal to, or longer than, the life expectancy of the animal. This approach was not applied directly here as there were concerns with regard to the validity of the equations as defined in Brown et al. [26]. Instead a pragmatic approach was taken whereby the model predictions were selected at 50% of the life expectancy of the animal. This was considered to represent an average individual in the population. Information was therefore required in relation to the life expectancy of studied animals. Relevant life history information was collated from internet websites and life-span values are provided below (Table 6.13).

Table 6.12. Assumed mass, dimensions and occupancy factors for organisms included in the Chernobyl Scenario which were modelled using FASTer-EPIC DOSES3D.

Common name	Scientific name	Mass (kg)	Dimensions (cm)	Occupancy Factor	
				In soil*, air** or water***	On ground
Wolf	*Canis lupus*	45	86x42x24	0*	1
Roe Deer	*Capreolus capreolus*	17.5	70x32x15	0*	1
Common shrew	*Sorex araneus*	0.0095	5.6x1.8x1.8	0.5*	0.5
Bank vole	*Clethrionomys glareolus*	0.0235	7.4x2.8x2.2	0.5*	0.5
Root vole	*Microtus oeconomus*	0.05	10.6x3x3	0.5*	0.5
Wood mouse	*Apodemus sylvaticus*	0.02	6.8x2.4x2.4	0.5*	0.5
Yellow necked mouse	*Apodemus flavicollis*	0.034	7.2x3x3	0.5*	0.5
Great tit	*Parus major*	0.0181	6x3x2.4	0.8**	0.2
Long-tailed tit	*Aegithalos caudatus*	0.0093	4.8x2.2x2.1	0.8**	0.2
Robin	*Erithacus rubecula*	0.0181	6x3x2.4	0.6**	0.4
Brown frog	*Rana terrestris*	0.0226	6x3x2.4	0.25***	0.75
Edible frog	*Rana esculenta*	0.056	7.6x4.4x3.2	0.25***	0.75

Table 6.13. Life expectancy of the animals considered in this study.

Animal	Life expectancy	Reference
Roe Deer	11–12 years	http://www.answers.com/topic/capreolus-capreolus-1
Wolf	6-8 years	http://www.mnstate.edu/regsci/eyes/Natural%20History%20of%20the%20Gray%20Wolf.htm
Bank voles	2 years	http://www.wildkids.org.uk/woodland/mammals3.htm
Common shrew	18 months	http://www.bbc.co.uk/nature/wildfacts/factfiles/642.shtml
Root vole	3 years	http://www.southeastwater.co.uk/otters.asp
Wood mouse	20 months	http://www.britishwildlifecentre.co.uk/animals/woodmouse.htm
Yellow-necked mouse	1 year	http://www.abdn.ac.uk/mammal/yellow_necked_mouse.shtml
Great tit	3 years	http://www.arkive.org/species/ARK/ and http://www.bto.org/birdfacts/indexa_all.htm
Long-tailed tit	2 years	http://www.rspb.org.uk/birds/guide/index.asp and http://www.bto.org/birdfacts/indexa_all.htm
Robin	2 years	http://www.arkive.org/species/ARK/ and http://www.bto.org/birdfacts/indexa_all.htm
Brown frog	up to 8 years	http://www.arkive.org/species/ARK/
Edible frog	assumed same as Brown frog	http://en.wikipedia.org/wiki/Main_Page

Table 6.14. Animal masses and dietary compositions assumed in the configuration of FASTer.

Animal	Mass (kg)	Dietary composition
Roe deer	17.5	Grasses and herbs (33%); shrub (33%); tree (33%)
Wolf	45	Roe deer = 100%
Bank voles	0.0235	Grasses and herbs (60%); lichen and bryophytes (20%); soil invertebrate (20%)
Common shrew	0.0095	Soil Invertebrate (25%); detritivorous invertebrate(25%); flying insects (25%); gastropod (25%).
Root vole	0.05	Grasses and herbs (100%)
Wood mouse	0.02	Grasses and herbs (40%); soil invertebrate (40%); lichen and bryophytes (20%)
Yellow-necked mouse	0.035	Grasses and herbs (40%); soil invertebrate (40%); lichen and bryophytes (20%)
Great tit	0.018	Flying insects (40 %); soil invertebrates (20 %); detrivorous insects (20%); grass and herbs (20%)
Long-tailed tit	0.009	Flying insects (40 %); soil invertebrates (20 %); detrivorous insects (20%); grass and herbs (20%)
Robin	0.018	Soil invertebrates (40%); detrivorous insects (40%); grass and herbs (20%)
Brown frog	0.0023	Soil invertebrates (25%); gastropod (25%); detrivorous insects (25%); flying insects (25%)
Edible frog	0.0055	Soil invertebrates (25%); gastropod (25%); detrivorous insects (25%); flying insects (25%)

Assumed masses and dietary compositions are presented in Table (6.14) these were predominantly derived from the sources identified in Table 6.13.

To derive the activity concentrations in the dietary components identified for the different animals in Table 6.14, biota-soil CR values were used from a development version (28/03/2006) of the ERICA Tool database. For the biota-radionuclide combinations of interest the only difference in CR values to the final version of the ERICA Tool database [23] was the CR value for Pu to tree (a value of 2.07×10^{-5} being used whereas the final database version was 3.15×10^{-2}).

Allometric parameters used in the derivation of fresh and dry matter ingestion rates (as described in Section 2.13) are presented in Table 6.15. No soil ingestion has been assumed for rodents. Allometric parameters used to derive radionuclide loss rates from the animals are in Table 2.3. The FASTer model and the allometric parameters in Table 2.3 were derived for mammals, their applicability to birds and especially poikilothermic amphibians and reptiles may be questionable.

6.2.7. D-Max

Tables 6.2–6.4 presents the CR values applied by this approach with the Chernobyl scenario; the derivation of CR values used are described in Section 2.15. The CR value used for frog species related whole-body activity concentrations to those in water (see Section 2.14).

6.3. Results

There were considerable ranges in the activity concentrations in the observed data; for instance, the range (across all biota) in the mean ^{137}Cs activity concentrations of datasets available was 10^{1} to 10^{7} Bq kg^{-1} (fw). To allow easy comparison of predicted and observed activity concentrations in the subsequent text predictions have been normalised to the observed data mean (for each prediction). Similarly, estimated total absorbed dose rates across all models and species ranged from 4.9×10^{-4} μGy h^{-1} to 39000 μGy h^{-1}. To simplify comparisons, predicted dose rates have been normalised to the soil activity concentrations provided in the scenario for each prediction (Section 6.3.4).

Table 6.15. Allometric parameters used in derivation of ingestion rates taken from Nagy [41].

Animal Group	Ingestion	a	b
Rodentia	FMI (kg d^{-1})	0.2296	0.864
Rodentia	DMI (kg d^{-1})	0.0697	0.774
Herbivores	FMI (kg d^{-1})	0.1995	0.628
Herbivores	DMI (kg d^{-1})	0.0658	0.628
Carnivores	FMI (kg d^{-1})	0.1641	0.848
Carnivores	DMI (kg d^{-1})	0.0486	0.834
Omnivorous birds	FMI (g d^{-1})	0.159	0.627
Omnivorous birds	DMI (g d^{-1})	0.051	0.627
All reptiles*	FMI (g d^{-1})	0.021	0.932
All reptiles*	DMI (g d^{-1})	0.0064	0.920

*Poikilothermic reptile values have also been used for amphibians as Nagy [41] presents no values for the latter.

6.3.1. Caesium-137

Predicted ^{137}Cs activity concentrations for each model are compared to observed values in Figures 6.1-6.3. Note in these, and subsequent figures, where the observed data mean does not have an associated standard deviation, the sample size was one.

The lowest predicted ^{137}Cs activity concentrations were generally the DosDimEco model which often predicted values more than one-order of magnitude below the observed mean. The exception was for rodent species for which the EA R&D128 model predicted values 1-2 orders of magnitude below the observed mean (Figure 6.3). Predictions by the FASTer model for both frog species (not modelled by DosDiMEco) were more than one order of magnitude below the observed data (Figure 6.1). The highest predicted ^{137}Cs activity concentrations were generally made by the D-Max model, most notably for bird species which D-Max over-predicted by more than two orders of magnitude (Figure 6.1). Exceptions were for amphibian species which were most over predicted by the EA R&D128 model (Figure 6.1) and rodents for which the LETDOS-BIOTA model gave predictions comparable to those of D-Max (Figure 6.3).

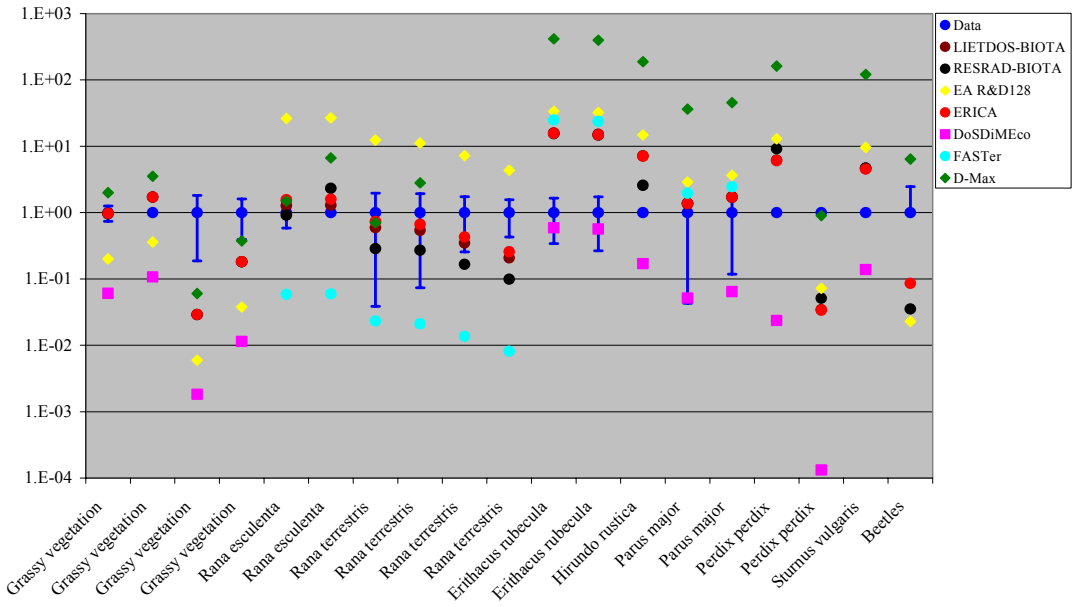

Fig. 6.1. A comparison of predicted and observed (mean±SD) ^{137}Cs activity concentrations grassy vegetation, frog species, bird species and a sample of beetles. For each comparison, predictions and data are normalised to the observed data mean.

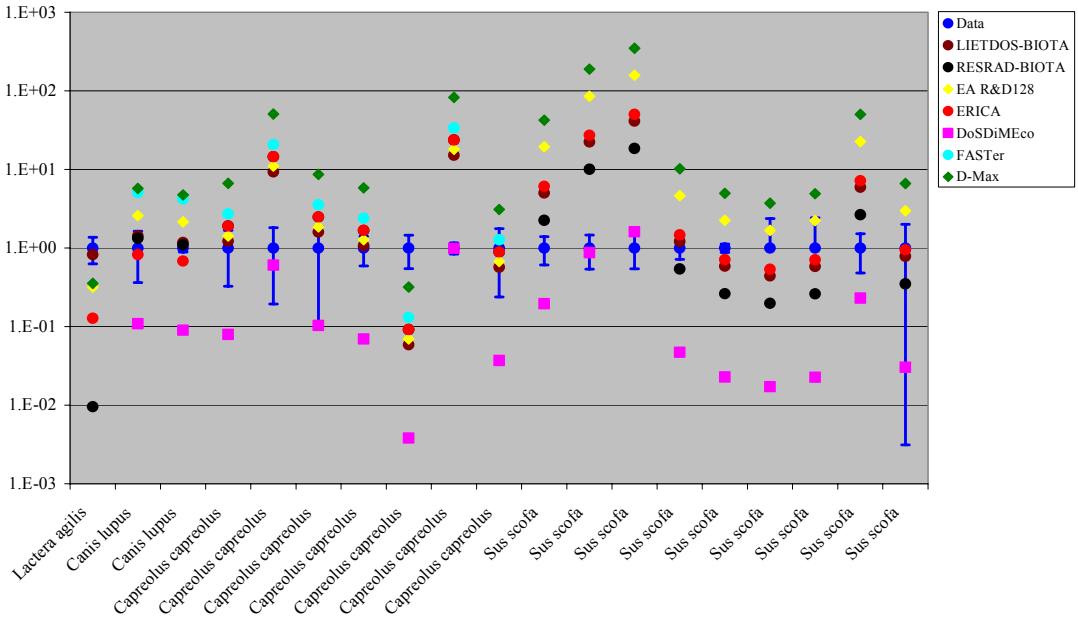

Fig. 6.2. A comparison of predicted and observed (mean±SD) ^{137}Cs activity concentrations in a reptile and large mammal species. For each comparison, predictions and data are normalised to the observed data mean.

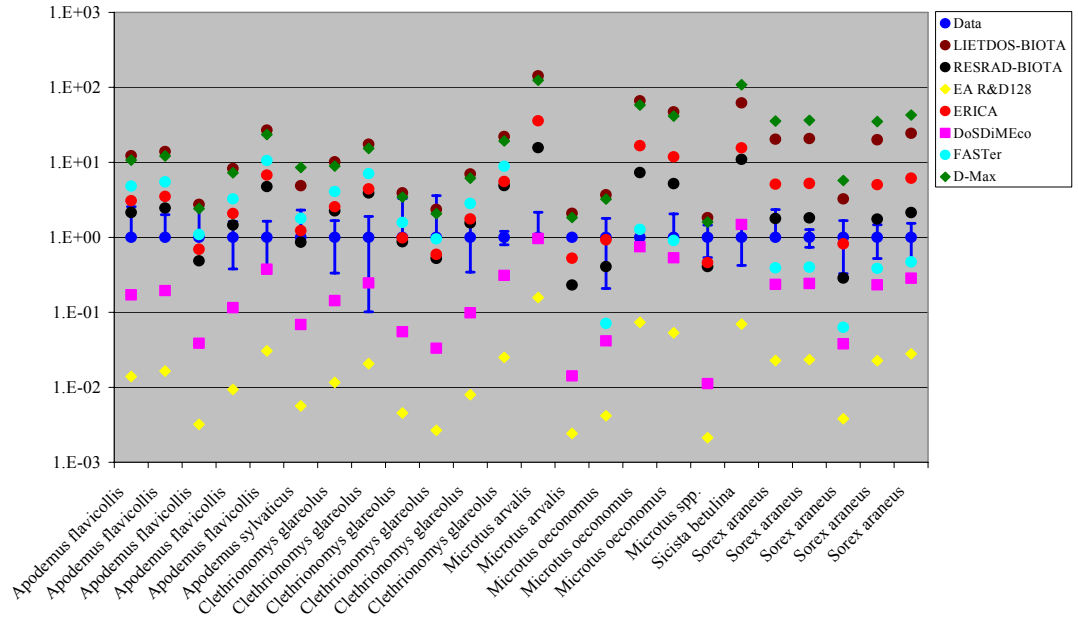

Fig. 6.3. A comparison of predicted and observed (mean±SD) ^{137}Cs activity concentrations in rodent species. For each comparison, predictions and data are normalised to the observed data mean.

6.3.2. Strontium-90

Predicted ^{90}Sr activity concentrations for each model are compared to observed values in Figures 6.4-6.6.

As for predictions of ^{137}Cs, the DosDiMEco model tended to predict the lowest ^{90}Sr activity concentrations, however, predictions from this model tended to be in better agreement with the observed data than was the case for ^{137}Cs. Whilst the D-Max model tended to predict comparatively high ^{90}Sr activity concentrations, this was less markedly so than for ^{137}Cs. For grassy vegetation and some rodent species notable over predictions were made by the EA R&D128 and FASTer models respectively (Figures 6.4 and 6.6).

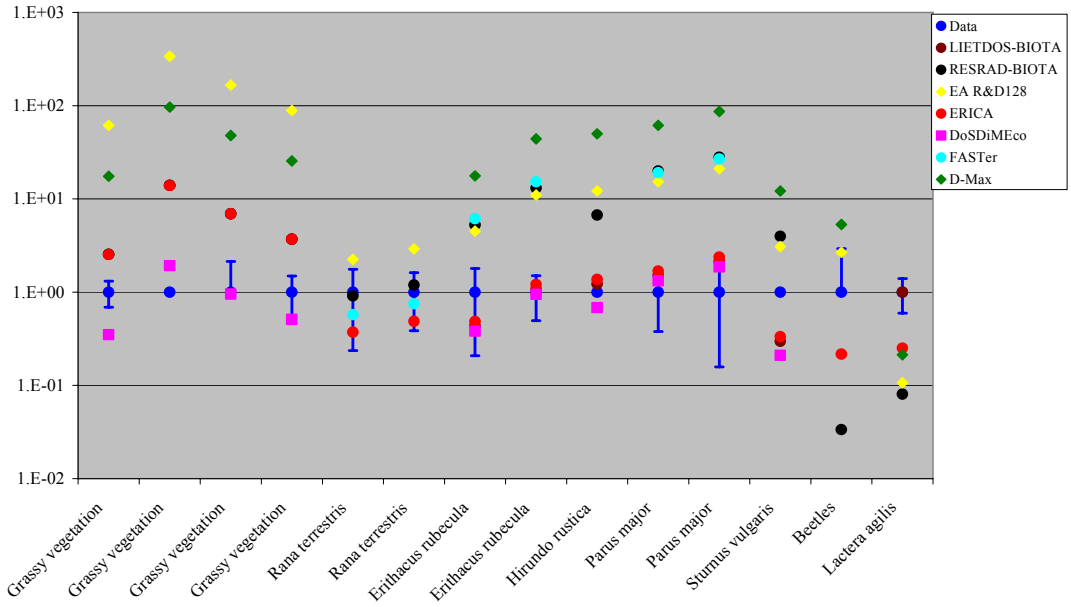

Fig. 6.4. A comparison of predicted and observed (mean±SD) ^{90}Sr activity concentrations grassy vegetation, frog species, bird species, a sample of beetles and a reptile. For each comparison, predictions and data are normalised to the observed data mean.

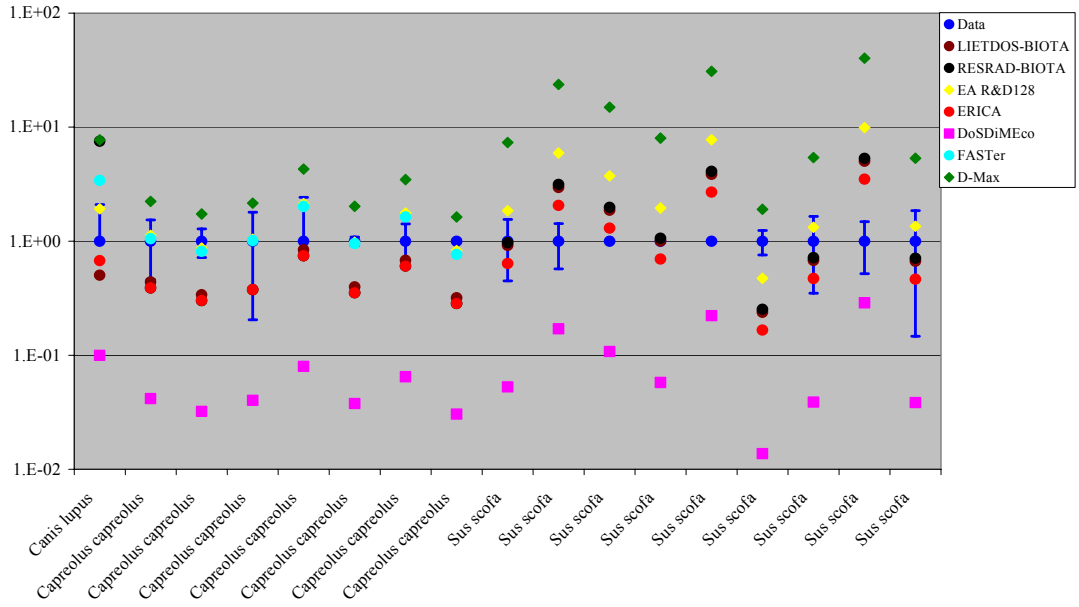

Fig. 6.5. A comparison of predicted and observed (mean±SD) ^{90}Sr activity concentrations in large mammal species. For each comparison, predictions and data are normalised to the observed data mean.

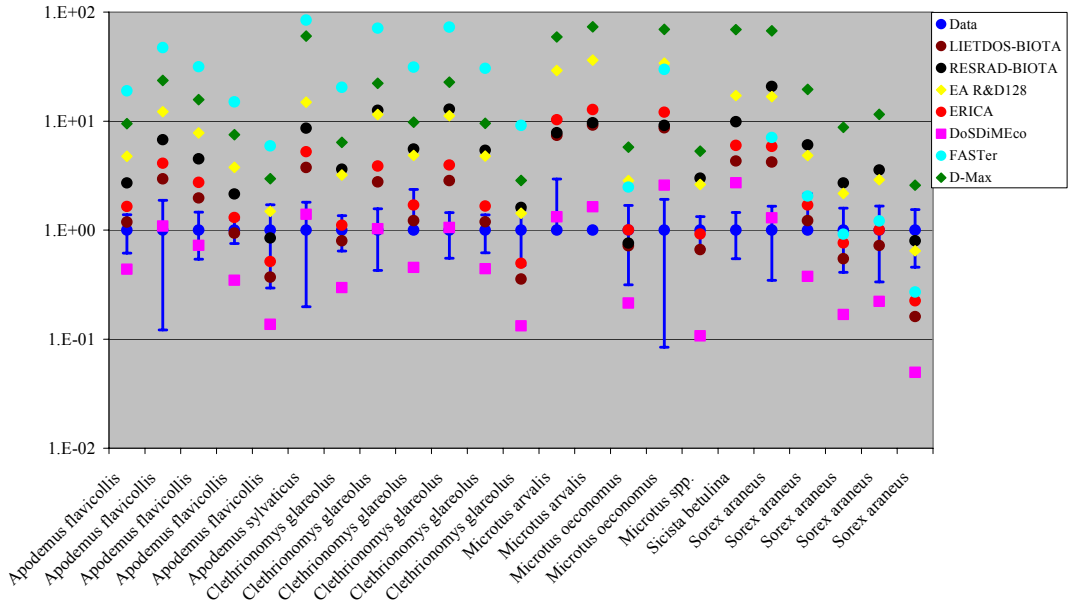

Fig. 6.6. A comparison of predicted and observed (mean±SD) ^{90}Sr activity concentrations in rodent species. For each comparison, predictions and data are normalised to the observed data mean.

6.3.3. *Actinides*

Predicted activity concentrations of Pu-isotopes in a range of bird and rodent species, and ^{241}Am in *C. glareolus* are compared to observed data in Figure 6.7. For birds, Pu comprises $^{238+239+240}$Pu whilst for mammals it is $^{239+240}$Pu, reflecting differences in the available data for soil within the scenario.

Unlike for ^{137}Cs and ^{90}Sr, D-Max and DosDiMEco do not make extreme predictions compared to the other models and or data. There is a considerable range in the predicted concentrations of Pu for the two bird samples available: EA R&D128 predicted values in excess of 100 times greater than the mean of the observed data and LIETDOS-BIOTA predicted values more than 100 times lower than the mean.

6.3.4. *Absorbed dose rates*

The combined internal dose rates and total dose rates predicted by each model are compared in Table 6.16. In virtually all cases the internal dose was predicted to dominate the total dose rate. The D-Max model predicts 'maximum dose rates' (see Section 2.14) and does not report results in the same manner as the other models considered. As would be expected, absorbed dose rate predictions by this model are in most instances (20 of the 24 comparisons) higher than the total dose rate estimates by the other models. As this model is not directly comparable with the others participating in the scenario it is not considered further in this section.

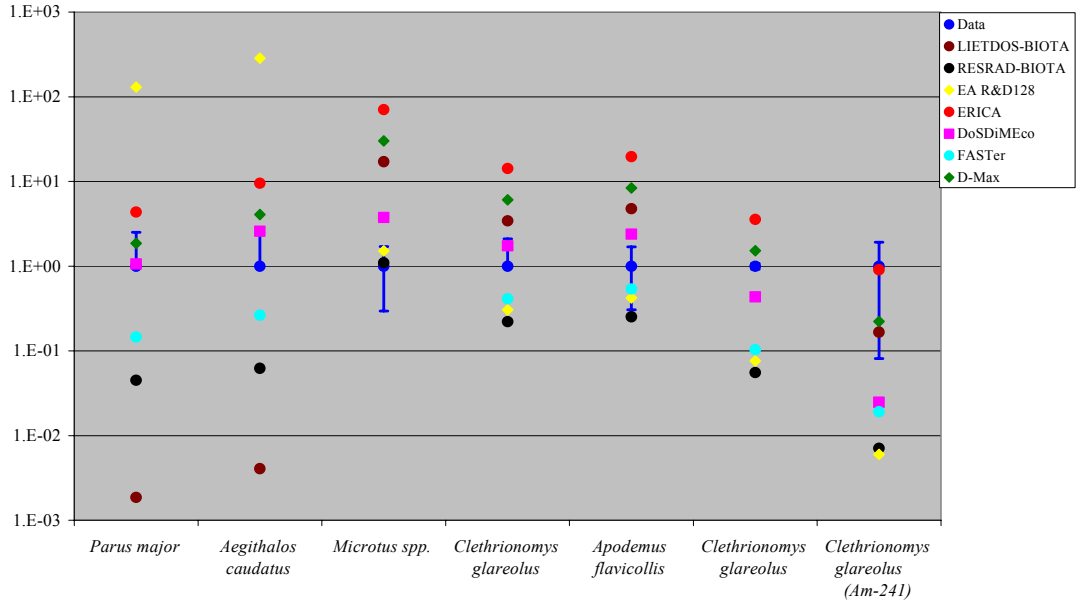

Fig. 6.7. A comparison of predicted and observed (mean±SD) Pu activity concentrations in bird and rodent species, and ^{241}Am activity concentrations in C. glareolus. For each comparison, predictions and data are normalised to the observed data mean.

For more than 50% of the comparisons, the variability in internal dose rate between the participating models was less than an order of magnitude. The most variable predictions were for long-tailed tit (*A. caudatus*), edible frog (*R. esculenta*) and partridge (*Perdic perdix*). With two exceptions, all models predicted external dose rates to within an order of magnitude of each other. The exceptions were for barn swallow (*H. rustica*) (input soil concentrations ^{137}Cs and ^{90}Sr) and long-tailed tit (input soil concentrations Pu-isotopes) for which DosDiMEco predicted dose rates lower than the other participating models by approximately one and three orders of magnitude respectively. Total dose rate predictions were again most variable for the long-tailed tit, partridge and edible frog.

To enable illustrative comparisons of the different contributions to the estimated dose rates, Figures 6.8-6.15 present the absorbed dose rate for each radionuclide normalised to the activity concentration of that radionuclide in soil; internal and external contributions are shown separately for each radionuclide. Note that not all radionuclides were considered in all comparison (see Table A4.1). There are some notable instances where the predicted contributions to the total dose rate varied considerably between models:

- Root vole (*M. oeconomus*) (Figure 6.13) greater contribution of ^{137}Cs predicted by LIETDOS-BIOTA
- Grassy vegetation (Figure 6.13) greater contribution of ^{90}Sr predicted by EA R&D128; lack of contribution of ^{90}Sr to external dose rate predicted by the ERICA Tool
- Beetle (Figure 6.13) dominance of external dose rate contribution to total dose rate predicted by RESRAD-BIOTA; lack of contribution of ^{90}Sr to external dose rate predicted by the ERICA Tool
- Relative contributions of ^{90}Sr and ^{137}Cs to internal dose rate for rodent species between different models (see Figures 6.10-6.13)
- Greater contribution of Pu-isotopes and ^{241}Am predicted by the ERICA Tool (Figures 6.8, 6.10-6.12).

Table 6.16. Comparison of the predicted combined internal and total (internal+external) dose rates predicted by each participating model.

Species	Absorbed dose rate (μGy h^{-1})													
	RESRAD		R&D128		ERICA		Lietdos		DosDiMEco		FASTer		D-Max	
	Internal Dose	Total Dose	Internal Dose	Total Dose	Internal Dose	Total Dose	Internal Dose	Total Dose	Internal Dose	Total Dose	Internal Dose	Total Dose	Internal Dose	Total Dose
Grassy vegetation	5.2	19.8	10.2	24.4	4.8	15.4	n/r	n/r	n/r	n/r	n/r	n/r	N/A	35.0
Lactera agilis	22.7	34.2	90.0	96.0	91.7	93.4	n/r	n/r	n/r	n/r	n/r	n/r	N/A	245
Beetles	0.6	10.1	16.2	22.1	2.0	10.4	n/r	n/r	n/r	n/r	n/r	n/r	N/A	245
Sicista betulina	2994	3590	3904	4371	2587	2827	5390	6187	3304	3713	n/r	n/r	N/A	38725
Rana esculenta	1.9	3.9	100	110	261	277	n/r	n/r	n/r	n/r	0.2	9.4	N/A	70.0
Hirundo rustica	7.2	8.3	17.7	18.5	3.3	4.6	n/r	n/r	3.3	3.4	n/r	n/r	N/A	175
Perdix perdix	9.5	10.4	19.1	19.8	2.8	3.9	n/r	n/r	0.1	0.6	n/r	n/r	N/A	146
Sturnus vulgaris	24.6	26.1	24.1	25.2	4.5	6.3	n/r	n/r	5.7	6.3	n/r	n/r	N/A	241
Canis lupus	6.5	6.7	3.9	4.1	1.8	1.9	n/r	n/r	0.6	0.7	9.3	9.3	N/A	18.3
Capreolus capreolus	2.0	2.3	2.3	2.5	2.3	2.4	1.4	1.6	0.1	0.2	3.3	3.4	N/A	11.6
Sus scofa	1.2	1.3	3.9	4.1	1.8	1.9	1.5	n/r	0.2	0.3	n/r	n/r	N/A	18.3
Apodemus sylvaticus	223	276	270	312	203	241	464	535	140	179	1651	1716	N/A	3127
Microtus arvalis	113	174	270	312	203	248	464	535	67.4	106	n/r	n/r	N/A	1563
Microtus spp.	223	254	160	181	109	132	236	271	34.3	54.5	n/r	n/r	N/A	824
Clethrionomys glareolus	77.4	90.6	51.1	59.9	41.4	51.3	99.6	115	26.8	34.8	359	372	N/A	323
Apodemus flavicollis	43.8	55.1	51.1	59.9	41.4	49.9	99.6	115	26.6	33.5	242	255	N/A	323
Clethrionomys glareolus	10.0	12.2	6.1	7.5	6.0	7.7	16.2	18.8	3.3	4.6	44.0	46.3	N/A	48.7
Apodemus flavicollis	5.9	7.7	6.1	7.5	6.0	7.4	16.2	18.8	3.3	4.4	30.2	32.4	N/A	48.7
Sorex araneus	9.0	11.2	6.1	7.5	6.0	7.9	16.2	18.8	2.8	4.2	3.2	5.3	N/A	97.4
Parus major	97.3	101	93.4	96.1	14.4	18.8	n/r	n/r	29.3	30.7	62.0	62.8	N/A	699
Aegithalos caudatus	4.4×10^{-4}	4.9×10^{-4}	3.4	3.4	0.1	0.1	n/r	n/r	4.1×10^{-4}	4.1×10^{-4}	2.2×10^{-3}	2.2×10^{-3}	N/A	0.1
Erithacus rubecula	112	115	87.1	94.5	15.4	18.7	n/r	n/r	37.2	38.4	85.8	86.9	N/A	703
Rana terrestris	114	123	390	405	47.3	65.0	n/r	n/r	n/r	n/r	61.0	74.9	N/A	n/r
Microtus oeconomus	12.8	19.5	31.0	35.6	22.8	27.8	51.4	59.2	12.8	17.1	28.5	35.4	N/A	175

n/r – not reported; N/A – no applicable for this model (see text).

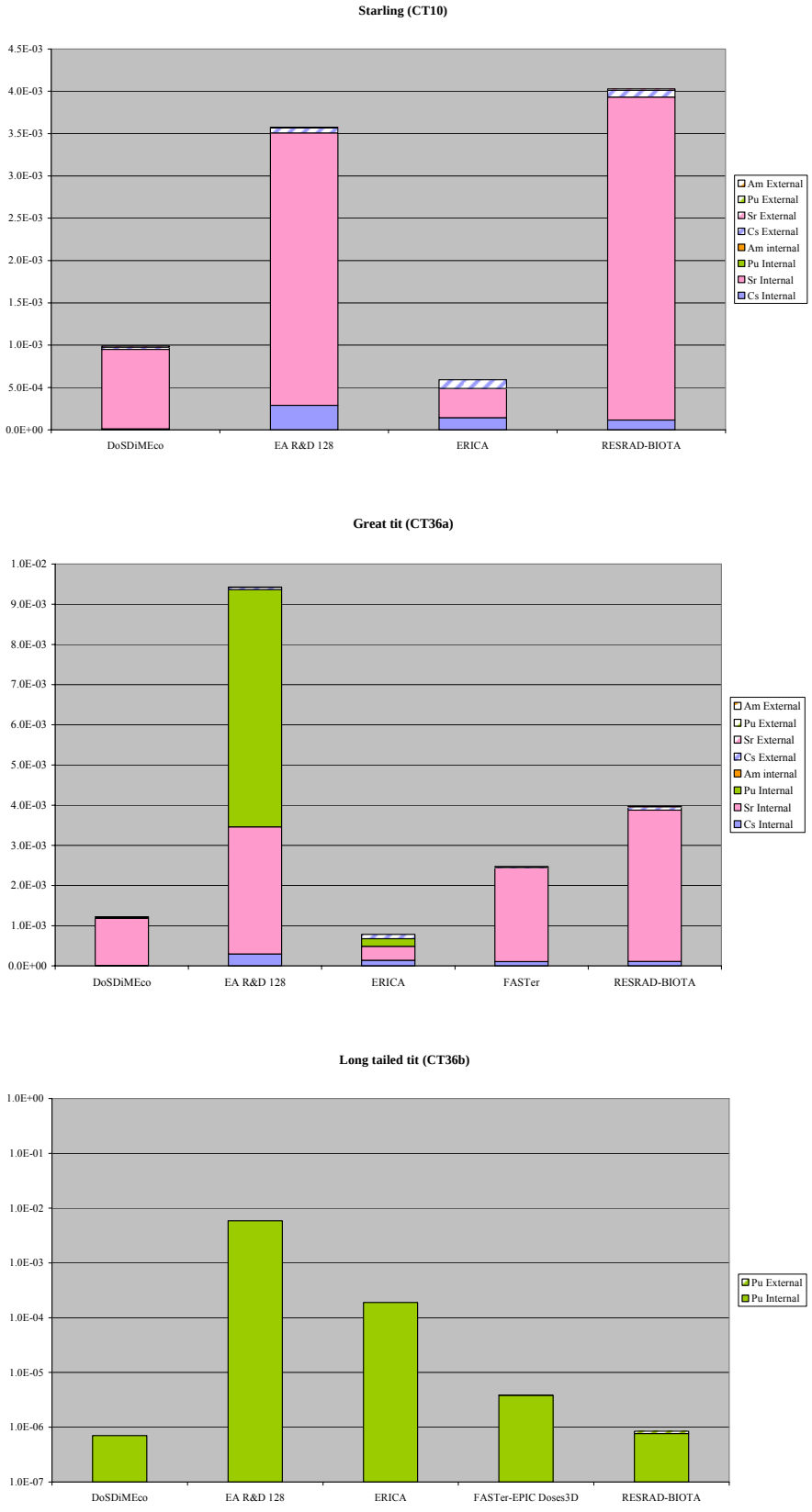

Fig. 6.8. An illustration of the relative contributions to total absorbed dose for three bird species (note only Pu was included within the scenario for long tailed tit (bottom)).

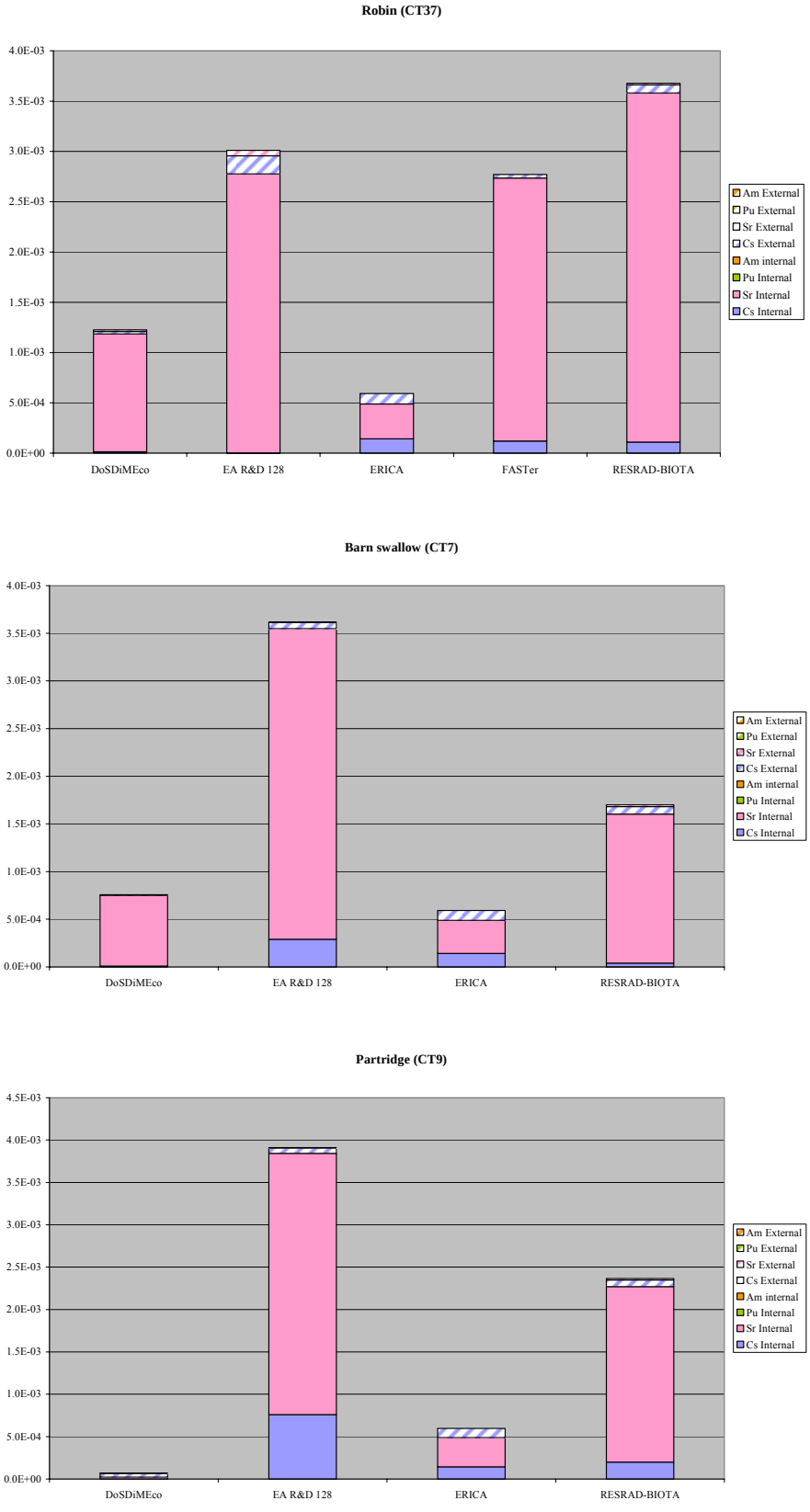

Fig. 6.9. An illustration of the relative contributions to total absorbed dose for three bird species.

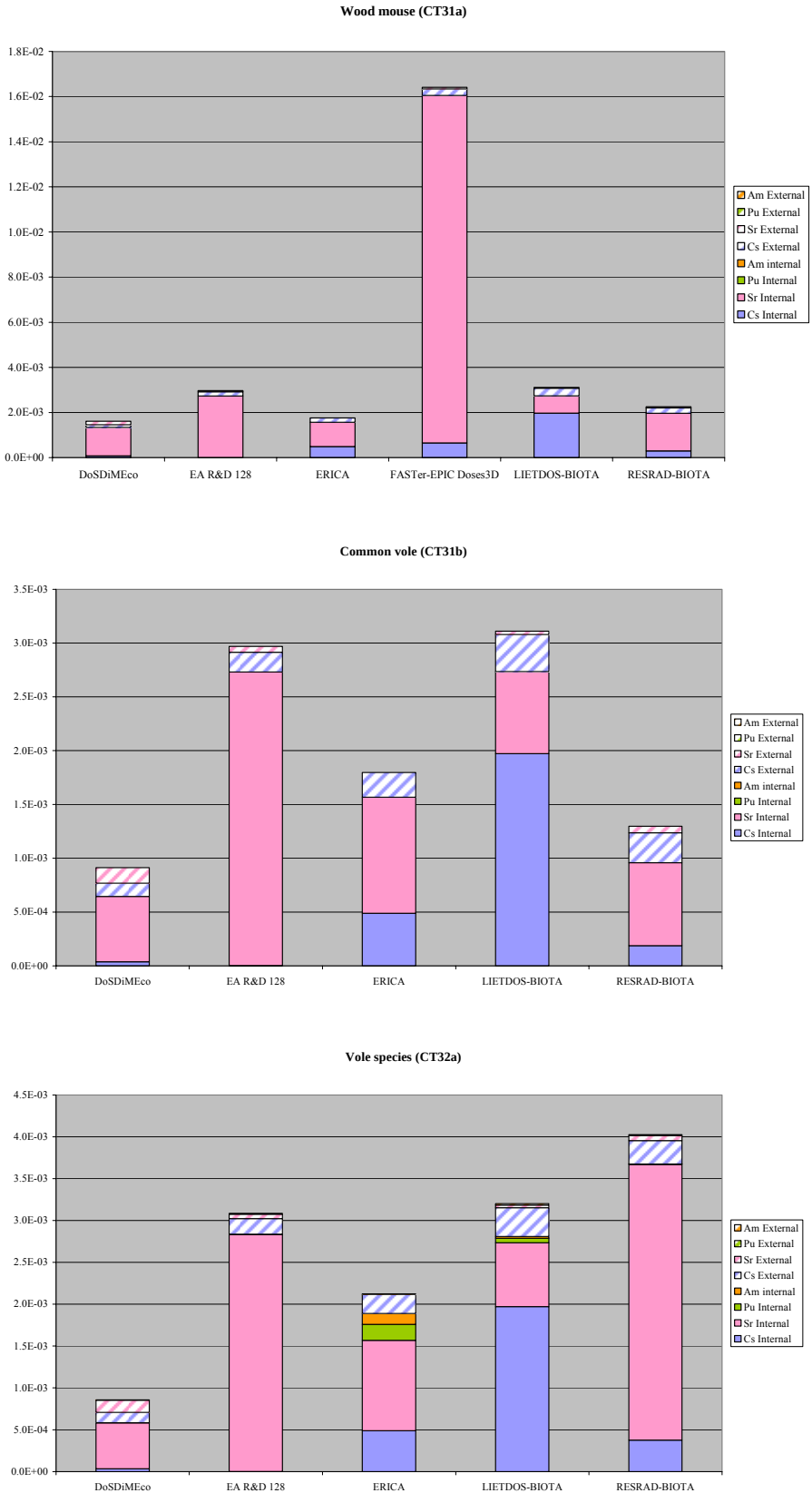

Fig. 6.10. An illustration of the relative contributions to total absorbed dose for three rodent species.

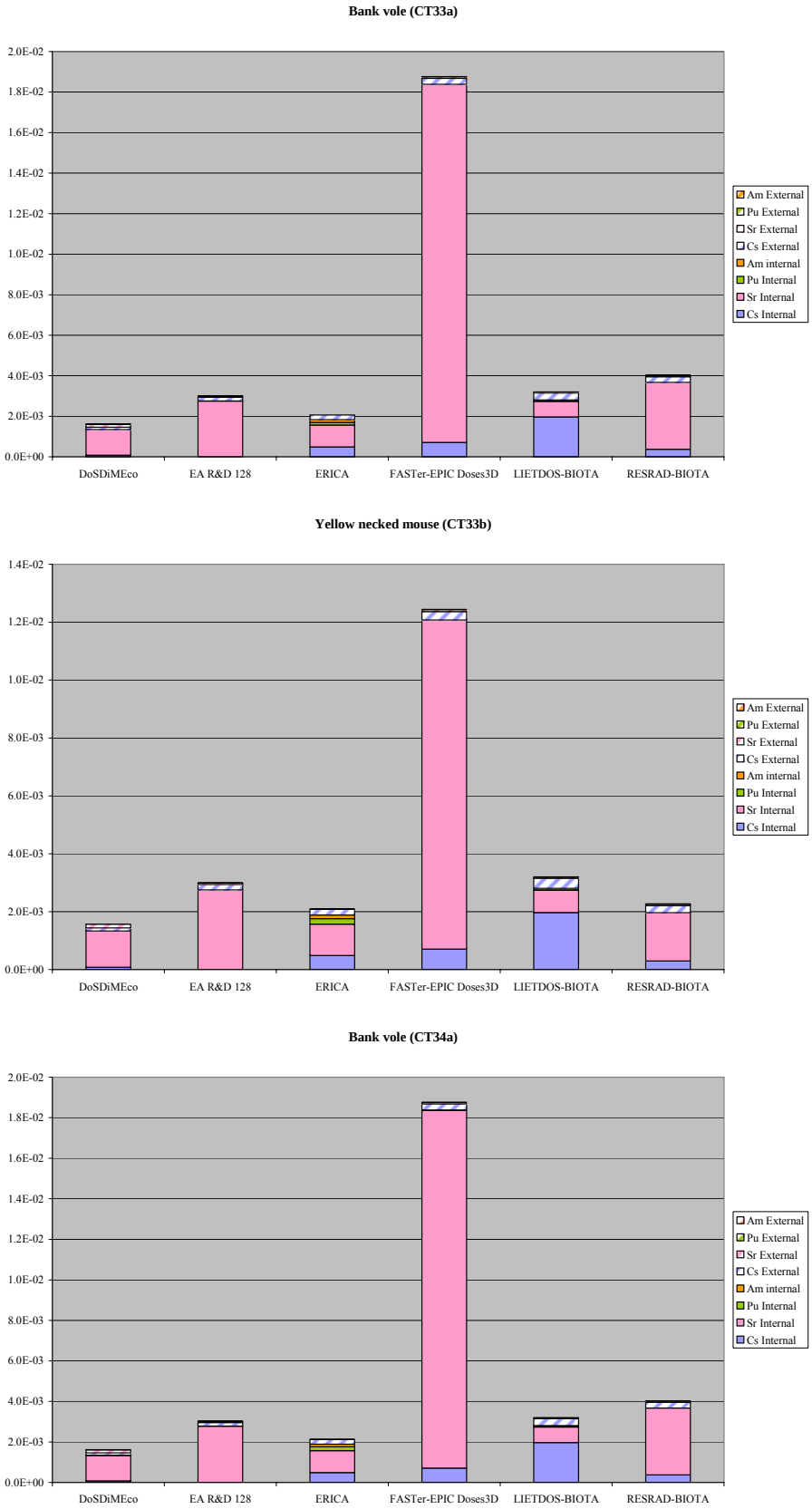

Fig. 6.11. An illustration of the relative contributions to total absorbed dose for three rodent species.

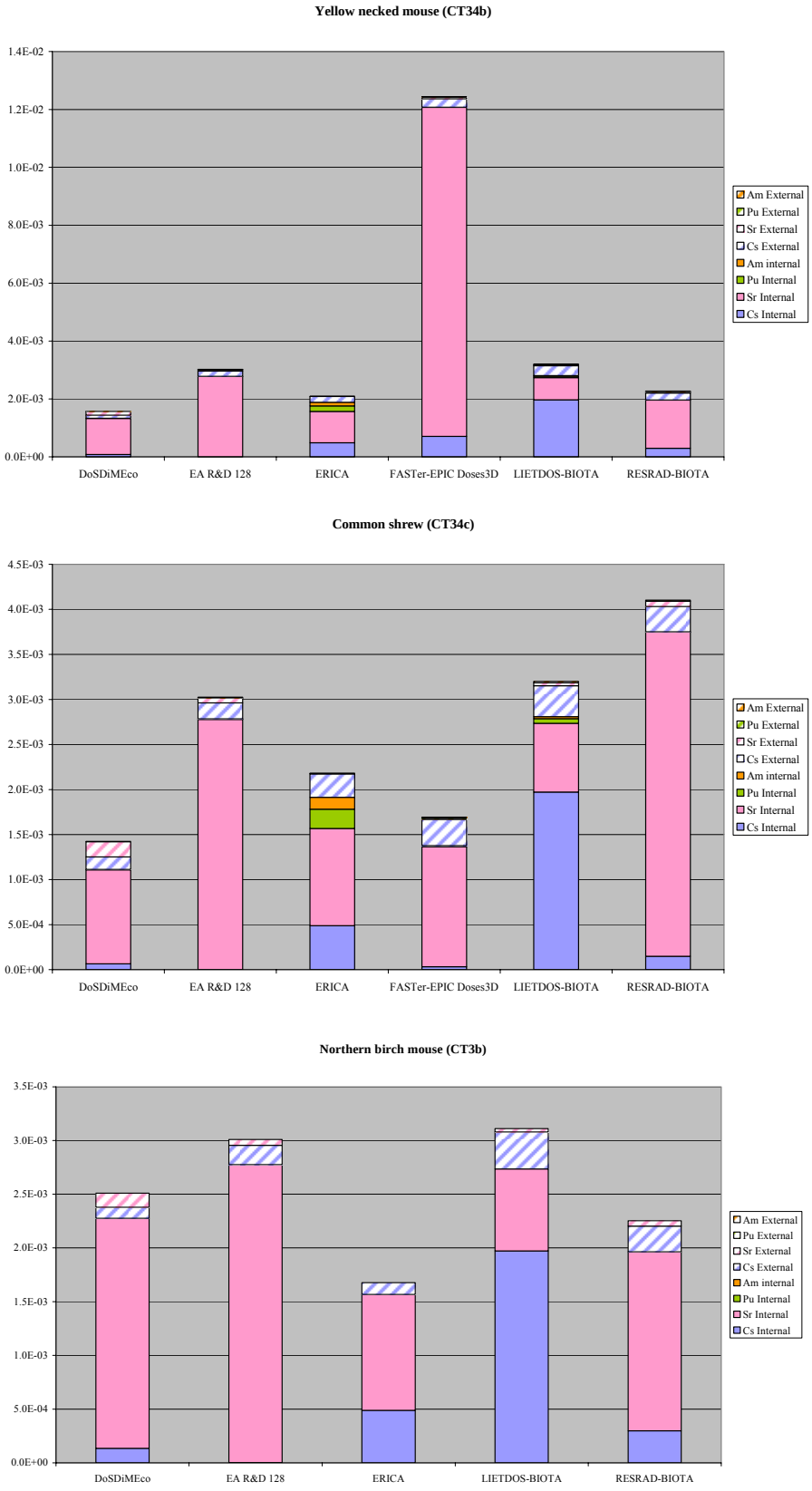

Fig. 6.12. An illustration of the relative contributions to total absorbed dose for three rodent species.

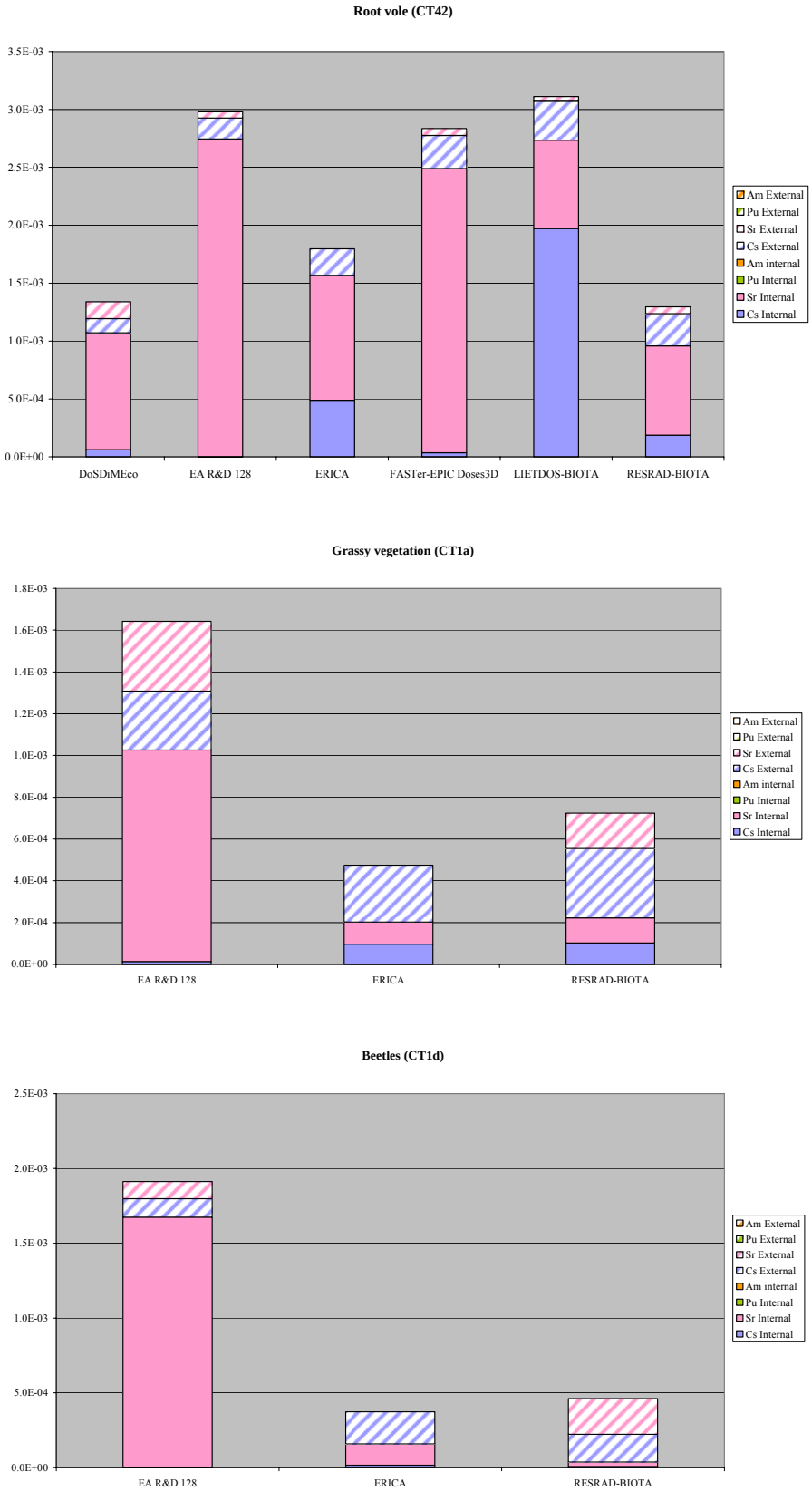

Fig. 6.13. An illustration of the relative contributions to total absorbed dose for root vole, grassy vegetation and beetles.

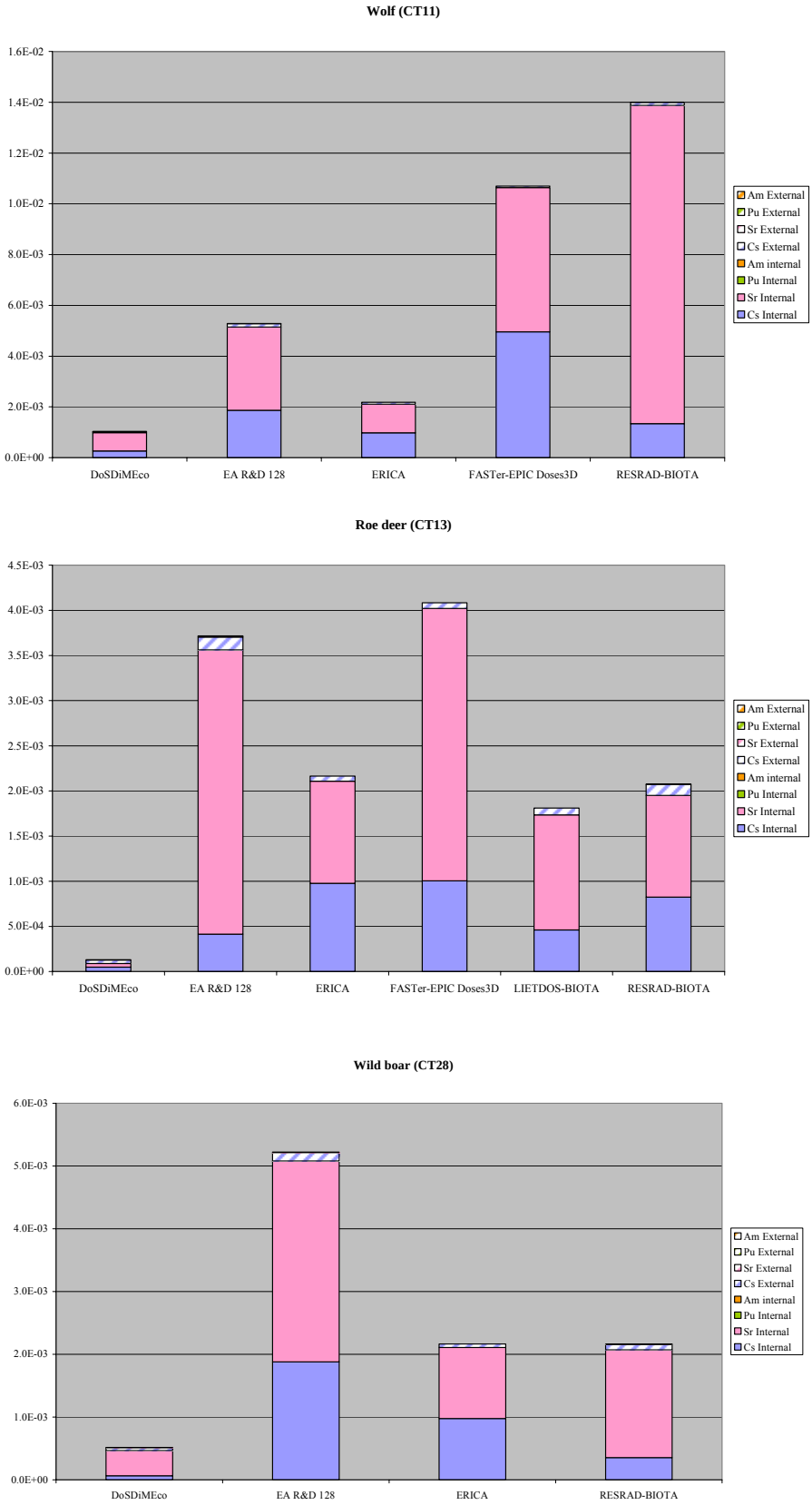

Fig. 6.14. An illustration of the relative contributions to total absorbed dose for three species of large mammals.

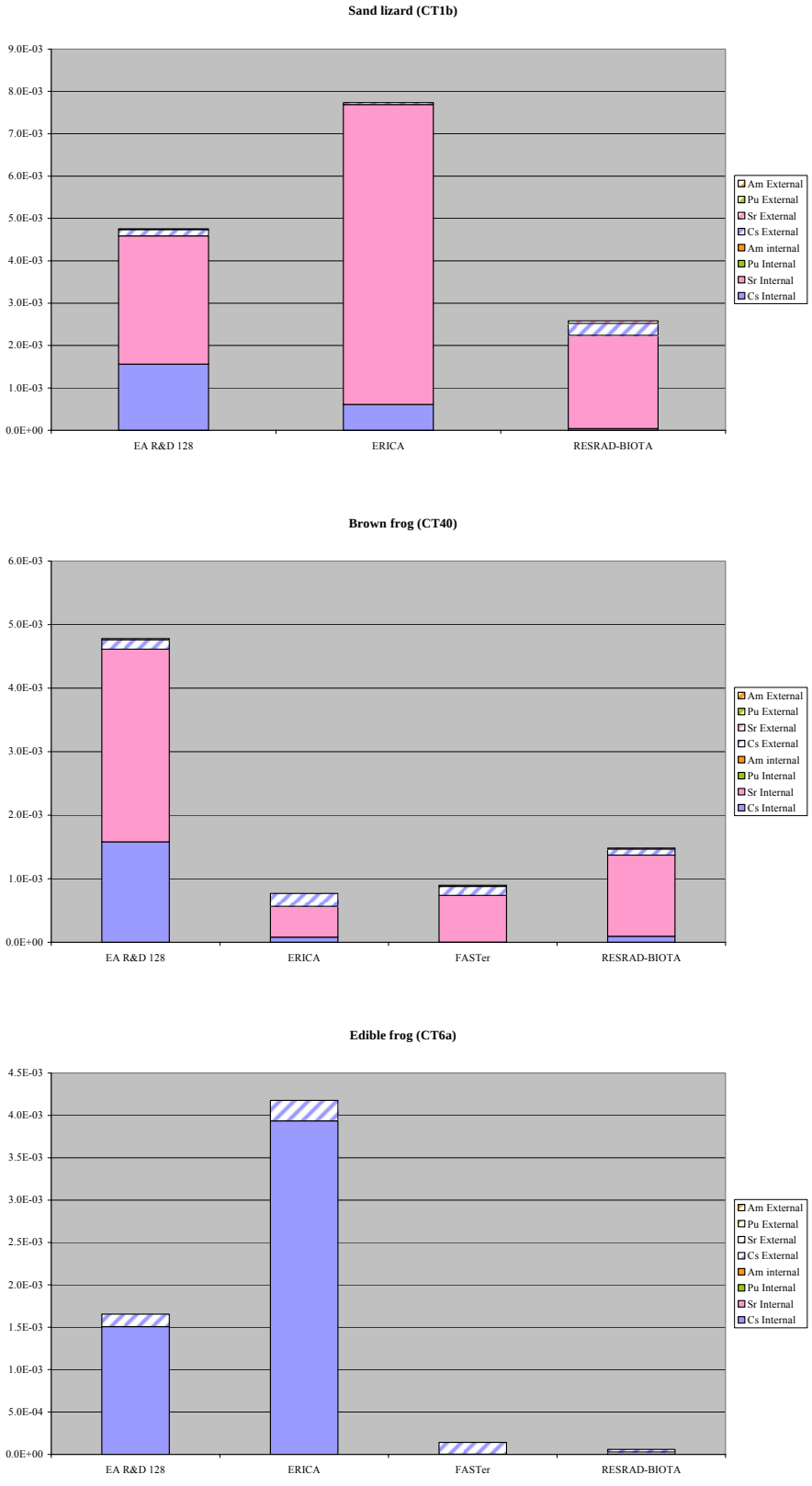

Fig. 6.15. An illustration of the relative contributions to total absorbed dose for a lizard and two species of amphibians.

6.3.5. Thermoluminescent dosimeter predictions

With the exception of the *M. oeconumus* sample, the data for dose rates estimated by TLDs attached to rodent species came from the study described by Beresford et al. [140] conducted at three forest sites during the summer of 2005. In addition to the TLDs attached to animals, TLDs were placed at various heights above and below the soil surface at the three study sites. These TLDs were paired, one being prepared in the same manner as those attached to the animals and one encased in 2 cm of Perspex. The dose rates recorded by TLDs prepared in the same manner as those attached to the study animals and placed at various heights above and below the soil surface were on average 1.95 times higher than the dose rates recorded by TLDs situated in the same location but shielded by 2 cm of Perspex. It was assumed that this 'additional' dose was the result of exposure to beta radiation (excluded by the Perspex) and that it was representative of beta dose rates recorded by the TLDs on the animal collars. However, this would not be representative of the whole-body external beta dose rates, therefore the results of the TLDs attached to the animals were corrected (i.e. dividing by 1.95) to derive the external gamma dose rate. These corrected results are compared to the model predictions here. The results for the *M. oeconumus* TLDs (from Chesser et al. [135] were similarly 'corrected'. Figure 6.16 compares the ^{137}Cs external dose rates predicted by the participating models to the available TLD measurements (N.B. Beresford et al. [140] reported that ^{137}Cs contributed ≥99 % of the total external dose rate at all three of their study sites).

Predictions were all within an order of magnitude of the observed data mean, with the majority being within the standard deviation of the data. The *M. oeconumus* sample was less well predicted. The D-Max model was used to predict dose rates to the TLD in soil; although these predictions are not directly comparable to those of the other models they were similar to estimates of the LIETDOS-BIOTA model.

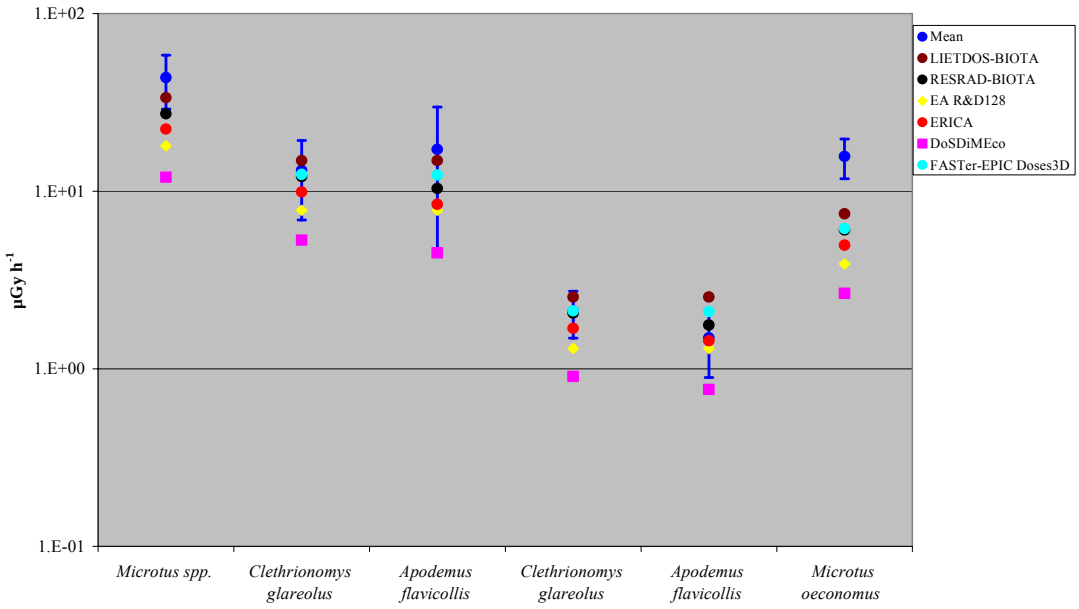

Fig. 6.16. Comparison of predicted external dose rates from ^{137}Cs to measurements from TLDs attached to collars fitted to rodents.

6.4. Statistical analyses

6.4.1. Activity concentrations

Z-scores have been derived for each prediction by reference to the observed data and associated standard deviations as for the Perch Lake results discussed in Chapter 5, i.e.:

$$Z = \frac{Predicted\ activity\ concentration - Observed\ mean\ activity\ concentration}{Observed\ SD} \quad (6.1)$$

Tables 6.17 -6.19 present a summary by radionuclide of the resultant Z-scores for each model including the overall number of Z-score ≥3 (indicating a comparatively poor prediction relative to the available data) and the average Z-score by organism group. The model with the greatest number of Z-scores ≥3 was D-Max (>70% of predictions for both ^{137}Cs and ^{90}Sr) reflecting the tendency of this model to overestimate. Predictions with the least number of high Z-scores were those of the DosDiMEco model (<10%). However, as noted in Chapter 5 caution is needed when interpreting the Z-scores as when the standard deviation of the observed data is relatively large (compared to the mean) they better identify those models which overestimate than those which underestimate. For instance, although DosDiMEco had the least number of predictions with a Z-score ≥3, the total number of predictions by DosDiMEco which were in agreement with, or in excess of, the data mean was only 2 and 12 for ^{137}Cs and ^{90}Sr respectively (i.e. as noted above this model tends to underestimate).

6.4.2. Dose rates

The combined internal, combined external and total dose rates were analysed using the same approach as described in Chapter 3. A 'robust' mean and standard deviation was generated for each comparison by the removal of outlying predictions. Z-scores were then estimated using the robust mean and standard deviation as the reference value. The predictions of the D-Max model were not considered in this analyses as its output is not comparable to that of the other participating models. The resultant Z-scores are provided in Tables 6.20–6.23; those results removed as outliers had resultant Z-scores ≥3. There were relatively few predictions which had Z-scores ≥3 reflecting the relative similarities in dose rate estimates apparent in Figures 6.8-6.15 above.

Table 6.17. Summary of Z-score by model and organism type for predicted ^{137}Cs activity concentrations.

Organism type	RESRAD-BIOTA	EA R&D128	ERICA	LIETDOS-BIOTA	DosDiMEco	FASTer	D-Max
Overall summary							
Number of predictions	56	56	56	55	48	39	54
Z-scores <3	39	34	37	28	43	26	15
Z-scores ≥3	17	22	19	27	5	13	39
By organism type							
Vegetation	0.8	0.8	0.8	0.8	0.8	N/A	0.7
Invertebrate	0.7	0.7	0.6	N/A	N/A	N/A	3.7
Amphibian	0.3	13.8	0.1	0.1	N/A	0.8	1.7
Reptile	2.7	1.8	2.3	0.5	N/A	N/A	1.7
Bird	0.2	0.4	0.2	0.2	0.7	0.7	13.8
Rodent	0.4	0.3	1.0	5.2	0.3	0.02	8.5
Large mammal	0.3	1.5	0.1	0.1	0.9	0.1	4.9

N/A – not applicable (prediction not made); Shaded cells denote a z-score ≥3

Table 6.18. Summary of Z-score by model and organism type for predicted ^{90}Sr activity concentrations.

Organism type	RESRAD-BIOTA	EA R&D128	ERICA	LIETDOS-BIOTA	DosDiMEco	FASTer	D-Max
Overall summary							
Number of predictions	50	50	50	47	46	34	48
Z-scores <3	18	22	32	31	39	14	8
Z-scores ≥3	32	28	18	16	7	20	40
By organism type							
Vegetation	2.3	65.4	2.3	2.3	0.1	N/A	18.4
Invertebrate	0.5	0.9	0.4	N/A	N/A	N/A	2.2
Amphibian	0.01	1.7	0.7	N/A	N/A	0.5	N/A
Reptile	2.3	2.2	1.9	<0.01	N/A	N/A	2.0
Bird	10.9	8.6	2.2	1.3	0.1	6.2	36.8
Rodent	5.3	9.0	2.4	1.5	0.2	9.2	34.8
Large mammal	0.2	1.7	0.4	0.1	1.5	0.4	9.8

N/A – not applicable (prediction not made); Shaded cells denote a z-score ≥3

Table 6.19. Summary of Z-score by model and organism type for predicted Pu and ^{241}Am activity concentrations.

Organism type	RESRAD-BIOTA	EA R&D128	ERICA	LIETDOS-BIOTA	DosDiMEco	FASTer	D-Max
Pu							
Overall summary							
Number of predictions	6	6	6	5	6	5	6
Z-scores <3	5	3	1	3	4	4	2
Z-scores ≥3	1	3	5	2	2	1	4
By organism type							
Bird	0.6	110.9	3.1	0.6	0.3	0.5	1.0
Rodent	1.5	1.1	58.1	16.5	3.1	1.2	23.5
^{241}Am Rodent	1.1	1.1	0.1	0.9	1.1	1.1	0.8

Shaded cells denote a z-score ≥3

Table 6.20. Z-scores for combined internal dose rate estimates.

Species	RESRAD -BIOTA	ERICA	EA R&D128	LIETDOS-BIOTA	DosDiMEco	FASTer-EPIC Doses3D
Grassy vegetation	0.39	0.45	0.45	N/A	N/A	N/A
Beetles	1.01	0.46	5.14	N/A	N/A	N/A
Rana esculenta	1.05	25.79	9.11	N/A	N/A	1.22
Rana terrestris	0.12	0.78	2.60	N/A	N/A	0.64
Lactera agilis	0.83	0.44	0.41	N/A	N/A	N/A
Aegithalos caudatus	1.20	10.40	320.62	N/A	1.20	1.03
Erithacus rubecula	0.92	0.95	0.43	N/A	0.53	0.41
Hirundo rustica	0.00	0.68	1.79	N/A	0.68	N/A
Parus major	0.91	0.92	0.82	N/A	0.59	0.13
Perdix perdix	2.73	0.09	6.75	N/A	1.21	N/A
Sturnus vulgaris	1.07	0.82	1.02	N/A	0.70	N/A
Apodemus flavicollis	0.51	0.55	0.39	0.43	0.80	2.81
Apodemus flavicollis	0.51	0.49	0.48	0.78	0.83	2.53
Apodemus sylvaticus	0.54	0.60	0.39	0.23	0.80	3.98
Clethrionomys glareolus	0.14	0.65	0.51	0.18	0.86	3.87
Clethrionomys glareolus	0.17	0.60	0.59	0.49	0.88	3.47
Microtus arvalis	0.59	0.07	0.32	1.44	0.85	N/A
Microtus oeconomus	0.67	0.23	0.14	1.05	0.67	0.03
Microtus spp.	0.61	0.33	0.09	0.72	0.96	N/A
Sicista betulina	0.35	0.47	0.08	0.36	0.26	N/A
Sorex araneus	0.33	0.20	0.18	1.58	0.76	0.69
Capreolus capreolus	0.37	0.59	0.54	0.10	1.16	1.34
Sus scofa	0.27	0.30	2.06	0.01	1.04	N/A
Canis lupus	0.98	0.62	0.08	N/A	1.02	1.90

N/A – not applicable (prediction not made); shaded cells denote a z-score ≥3 (and removed outlying prediction).

Table 6.21. Z-scores for combined external dose rate estimates.

Species	RESRAD-BIOTA	ERICA	EA R&D128	LIETDOS-BIOTA	DosDiMEco	FASTer-EPIC Doses3D
Grassy vegetation	0.08	0.59	0.01	N/A	N/A	N/A
Beetles	0.31	0.01	0.72	N/A	N/A	N/A
Rana esculenta	1.79	2.42	0.56	N/A	N/A	0.36
Rana terrestris	0.81	0.46	0.04	N/A	N/A	0.16
Lactera agilis	2.77	1.62	0.30	N/A	N/A	N/A
Aegithalos caudatus	8.78	14.58	6.99	N/A	2.38	0.64
Erithacus rubecula	0.12	0.34	3.70	0.47	1.34	1.48
Hirundo rustica	1.73	2.60	0.70	N/A	2.02	N/A
Parus major	1.27	1.99	0.31	N/A	1.00	1.53
Perdix perdix	0.20	0.92	0.44	N/A	0.98	N/A
Sturnus vulgaris	0.55	1.15	0.19	N/A	1.36	N/A
Apodemus flavicollis	0.02	0.59	0.51	0.92	0.92	0.50
Apodemus flavicollis	0.05	0.52	0.54	1.01	1.02	0.53
Apodemus sylvaticus	0.06	0.68	0.54	0.77	0.67	0.48
Clethrionomys glareolus	0.21	0.44	0.66	0.66	0.82	0.33
Clethrionomys glareolus	0.24	0.36	0.68	0.74	0.92	0.35
Microtus arvalis	0.30	0.40	0.56	0.74	0.69	N/A
Microtus oeconomus	0.21	0.47	0.65	0.63	0.73	0.25
Microtus spp.	0.28	0.44	0.55	0.70	0.63	N/A
Sicista betulina	0.43	1.25	0.18	1.39	0.45	N/A
Sorex araneus	0.15	0.14	0.74	0.63	0.79	0.13
Capreolus capreolus	1.21	0.75	1.88	0.17	1.25	0.59
Sus scofa	0.29	0.72	1.61	N/A	1.07	N/A
Canis lupus	1.03	0.27	1.31	N/A	1.16	0.81

N/A – not applicable (prediction not made); shaded cells denote a z-score ≥3 (and removed outlying prediction).

Table 6.22. Z-scores for total dose rate estimates.

Species	RESRAD-BIOTA	ERICA	EA R&D128	LIETDOS-BIOTA	DosDiMEco	FASTer-EPIC Doses3D
Grassy vegetation	0.15	0.43	0.15	N/A	N/A	N/A
Beetles	0.46	0.43	0.67	N/A	N/A	N/A
Rana esculenta	1.27	9.26	2.61	N/A	N/A	0.77
Rana terrestris	0.18	0.77	2.65	N/A	N/A	0.67
Lactera agilis	0.78	0.32	0.37	N/A	N/A	N/A
Aegithalos caudatus	1.36	12.24	376.01	N/A	1.37	1.17
Erithacus rubecula	2.55	0.78	1.83	1.30	0.10	1.57
Hirundo rustica	0.07	0.59	1.90	N/A	0.81	N/A
Parus major	1.06	0.96	0.94	N/A	0.67	0.12
Perdix perdix	1.40	0.36	3.93	N/A	1.26	N/A
Sturnus vulgaris	1.15	0.79	1.07	N/A	0.80	N/A
Apodemus flavicollis	0.48	0.57	0.40	0.54	0.85	2.92
Apodemus flavicollis	0.47	0.51	0.50	0.89	0.88	2.55
Apodemus sylvaticus	0.53	0.64	0.41	0.31	0.84	4.13
Clethrionomys glareolus	0.10	0.67	0.55	0.25	0.91	3.99
Clethrionomys glareolus	0.13	0.61	0.62	0.57	0.94	3.47
Microtus arvalis	0.50	0.11	0.23	1.40	0.86	N/A
Microtus oeconomus	0.60	0.25	0.08	1.08	0.70	0.07
Microtus spp.	0.63	0.35	0.05	0.78	0.98	N/A
Sicista betulina	0.29	0.53	0.05	0.52	0.25	N/A
Sorex araneus	0.32	0.19	0.25	1.50	0.77	0.59
Capreolus capreolus	0.43	0.56	0.64	0.13	1.27	1.34
Sus scofa	0.20	0.39	2.46	N/A	1.13	N/A
Canis lupus	1.19	0.67	0.17	N/A	1.14	2.21

N/A – not applicable (prediction not made); shaded cells denote a z-score ≥3 (and removed outlying prediction).

Table 6.23. A comparison of Z-scores for all predicted TLD readings.

Species Latin	RESRAD-BIOTA	EA R&D128	ERICA	LIETDOS-BIOTA	DosDiMEco	FASTer-EPIC Doses3D
Microtus spp.	1.12	1.75	1.45	0.69	2.16	N/A
Clethrionomys glareolus	0.16	0.85	0.51	0.29	1.25	0.11
Apodemus flavicollis	0.54	0.75	0.70	0.18	1.01	0.38
Clethrionomys glareolus	0.08	1.30	0.68	0.68	1.94	0.01
Apodemus flavicollis	0.46	0.32	0.09	1.75	1.21	1.03
Microtus oeconomus	2.43	2.98	2.71	2.08	3.29	2.43

N/A – not applicable (prediction not made); Shaded cells denote a z-score ≥3

6.4.3. TLD predictions

The same approach used for predicted activity concentrations (Section 6.4.1) was used to derive Z-scores for predictions of the dose rates derived from the TLDs (Table 6.23). The highest Z-scores were recorded for predictions for the *M. oeconomus* sample which originated from a different reference source [135] than the other data [140]. However, only DosDiMEco had a Z-score in excess of 3 for this prediction.

6.5. Discussion

6.5.1. Whole-body activity concentrations

The Chernobyl scenario has allowed predictions of the seven participating models to be compared to available ^{90}Sr and ^{137}Cs whole-body activity concentrations in a wide range of vertebrate species (and a more limited comparison for lower organisms and actinide elements). In many instances, the majority of predictions are within an order of magnitude of

the measured data. However, there was considerable variation in predicted whole-body activity concentrations between the participating models with 3 to 4 orders of magnitude difference in predictions being common. Relative to the observed data there were predictions more than an order of magnitude either side of the data mean in most instances. In this section we attempt to understand some of the reasons for poor predictions by some models (in part identified by high Z-scores) and the variation between predictions. To aid this discussion, Figures 6.17 and 6.18 present the average normalised predicted ^{137}Cs and ^{90}Sr activity concentrations respectively grouped by organism types.

6.5.1.1. D-Max

The D-Max model generally over-predicted whole-body activity concentrations, often by more than an order of magnitude. This model is designed to be conservative and hence this outcome was to be anticipated (if it was to meet its conservative aim). However, there were a few instances (e.g. ^{241}Am activity concentrations in *C. glareolus*, ^{137}Cs and ^{90}Sr concentration in *L. aglis*, ^{137}Cs activity concentrations in grassy vegetation) when it predicted values below the data mean although the degree of under prediction was usually less than for most other models.

6.5.1.2. DosDiMEco

The DosDiMEco model tended to under predict whole-body activity concentrations generally having the lowest prediction of any of the models. To estimate the intake rates of herbivorous/omnivorous species the model takes CR values for grass and (agricultural) grain from those recommended for human food chain modelling in IAEA [34]. It is unlikely that these values will accurately model the transfer of radionuclides to the diet of wild animals in the Chernobyl exclusion zone. The model was subsequently re-run using CR values for 'grass&herb' from the ERICA Tool instead of the IAEA CR values for both grain and grass. Predicted ^{90}Sr activity concentrations for all mammals and birds increased by approximately six times over the initial estimates. Those for ^{137}Cs increased by approximately: 15 times for herbivorous and carnivorous mammals; 60 times for rodent species and; 120 times for bird species. Consequently, on average, with the revised parameters the model would not underestimate ^{90}Sr and ^{137}Cs activity concentrations for any animal type included within the scenario for which it was used to provide predictions.

6.5.1.3. EA R&D128 – ^{137}Cs activity concentrations in rodent species

The EA R&D128 model consistently under predicted ^{137}Cs activity concentrations in rodent species by typically 1-2 orders of magnitude. This is consistent with observations in Chapter 4 where it was noted that the CR value used in EA R&D128 was based upon a single study of a coastal sand dune ecosystem close to the Sellafield reprocessing plant and it is hence unlikely to be applicable to many other ecosystems.

6.5.1.4. EA R&D128 – ^{137}Cs activity concentrations in frog species

The EA R&D128 approach consistently over predicted ^{137}Cs activity concentrations in frog species by approximately an order of magnitude (Z-score >13). The CR value of 9 (also applied for wolves, wild boar and sand lizard in this scenario application) used was derived from values derived in 1986 for red fox (*Vulpes vulpes*) following the Chernobyl accident [78]; it is likely that these data were not in equilibrium and should not have been used to derive CR values (Table 6.3).

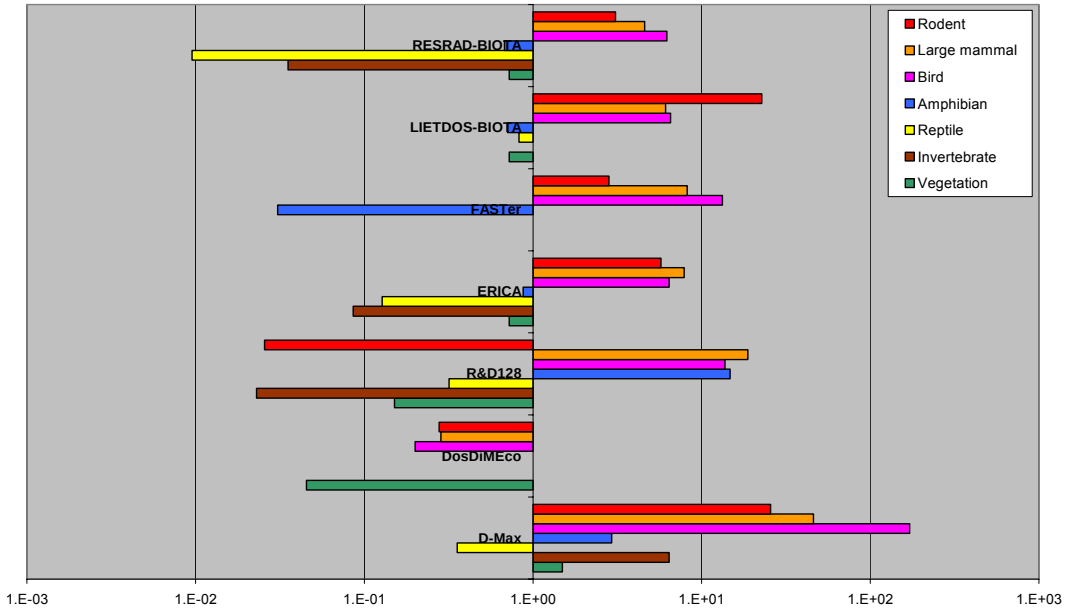

Fig. 6.17. A comparison of the mean normalised predicted ^{137}Cs activity concentrations for each model by organism type.

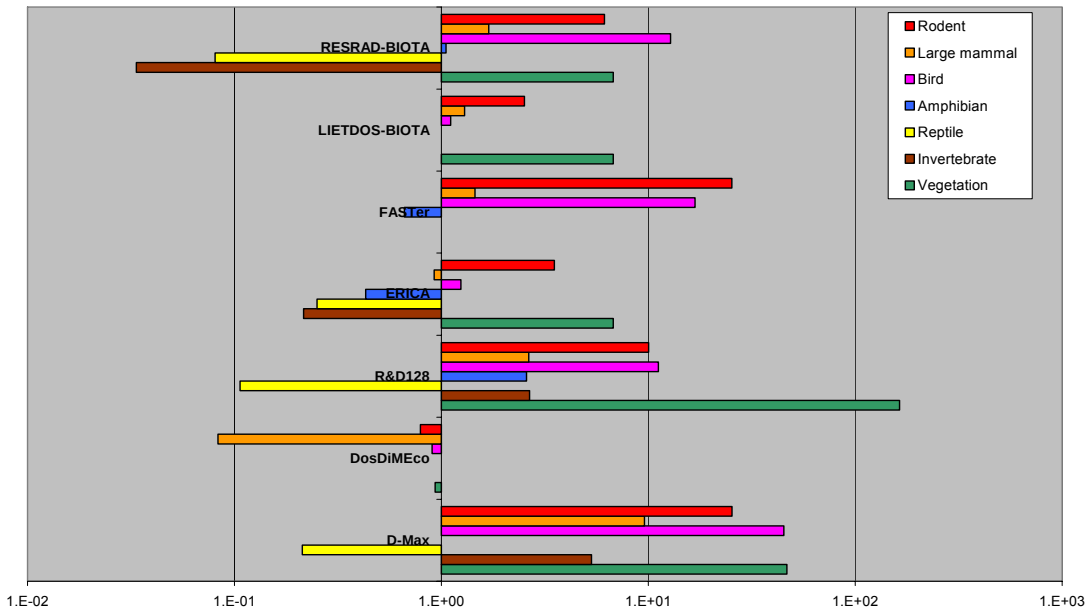

Fig. 6.18. A comparison of the mean normalised predicted ^{90}Sr activity concentrations for each model by organism type.

6.5.1.5. EA R&D128 – ^{90}Sr predictions

Strontium-90 activity concentrations in bird species, grassy vegetation and rodent species all tend to be over-predicted by the EA R&D128 approach (Z-scores ranging from 9 to 65). A CR value of 5 (derived for mice in a woodland close to the Sellafield reprocessing plant) is applied to **all** organisms within this approach to estimate ^{90}Sr activity concentrations.

6.5.1.6. ERICA Tool and RESRAD-BIOTA – Invertebrate predictions

RESRAD-BIOTA used CR values from the ERICA Tool to predict activity concentrations in the one sample of invertebrates ('beetle') included in the scenario. However, the ERICA Tool contains default CR values for a number of different terrestrial invertebrates and the values used by RESRAD-BIOTA were not the same as those used for the application of the ERICA Tool itself to this scenario (Tables 6.2 and 6.3). Both models under predicted ^{137}Cs and ^{90}Sr activity concentrations for the beetle sample. Although only one invertebrate sample was included in the scenario and hence we should not give undue weight to this observation, both RESRAD-BIOTA and the FASTer model used invertebrate CRs from the ERICA Tool to estimate radionuclide intake rates by some bird and mammal species. Therefore, if the ERICA Tool does under predict activity concentrations in invertebrate reference organisms it will impact on the predictions for higher organism by these two models.

6.5.1.7. LIETDOS-BIOTA – ^{137}Cs activity concentrations in rodent species

The LIETDOS-BIOTA model tends to over predict ^{137}Cs concentrations in rodent species (with an overall Z-score of >5). The CR values used by this model is 11.4 (Table 6.3) which was derived from stable element data for herbivorous mammals.

6.5.1.8. Predicted Pu activity concentrations

Predictions of Pu isotope concentrations in birds were relatively poorly predicted by a number of models. Over predictions of more than two orders of magnitude were made by the EA R&D128 approach, whilst RESRAD-BIOTA and LIETDOS-BIOTA under predicted by more than one and two orders of magnitude respectively. Due to the lack of specific data the EA R&D128 CR value of 0.70 was based upon the fresh weight activity concentration of soil (corrected from the models input of dry weight soil concentrations) consequently it is not surprising that this model overestimates. Predictions by RESRAD-BIOTA are discussed below.

The ERICA Tool over predicted Pu activity concentrations in three of the four rodent samples by more than an order of magnitude. The CR value used (2.34×10^{-2}) was derived from 18 entries (representing 123 measurements from 6 reference sources [23]) for a range of rodent species and large mammals. As a consequence of the poor predictions made here the ERICA Tool database has been reinvestigated and it was found that whilst data for reindeer (*Rangifer tarandus*) had been removed from the database two entries for (barren ground) caribou (*Rangifer arcticus*) had not. However, removing these data increases the CR value slightly to 2.47×10^{-2}. Furthermore, the ERICA Tool predicts the ^{241}Am activity concentrations in rodents relatively well (Figure 6.7) (N.B. the ERICA Tool ^{241}Am CR database was also found to contain caribou data, however, removing these only reduce the CR value used here by 30%).

Results in a CR value of 3.2×10^{-3}. However, the ERICA Tool CR value for Am is similar to the Pu default and predicts the observed ^{241}Am activity concentration in the limited data available for this scenario well.

The comparatively small standard deviation associated with the few available data (compare predictions to data and Z-scores for Pu to those for ^{241}Am in Figure 6.7 and Table 6.19) may contribute to the relatively poor Z-scores for predicted Pu activity concentrations in rodent species by the models in general (see Table 6.19).

6.5.1.9. RESRAD-BIOTA and FASTer predictions

The RESRAD-BIOTA and FASTer models differ from the 'equilibrium CR' approaches taken by the majority of the other models. The two models are comparable in concept and utilise similar allometric relationships to describe the biological half-life of radionuclides and also dietary intake rates.

The two models performed similarly to those using simple CR approaches. Both models had relatively poor Z-scores for the transfer for ^{90}Sr to some bird and rodent species, with the FASTer model especially tending to over predict by more than an order of magnitude. The FASTer model also underestimated ^{137}Cs activity concentrations in amphibians by more than an order of magnitude. The models also predicted considerably different activity concentrations in some instances. Below we consider the potential reasons for these observations.

Sensitivity analyses were conducted using the RESRAD-BIOTA code to find explanations for the differences in the RESRAD-BIOTA and FASTer predictions for some biota types. Parameters used in both models can be found in Sections 6.2.1 and 6.2.6 above (see also Chapter 2). In each sensitivity analysis, the value of one input parameter in RESRAD-BIOTA was changed to the FASTer input value, the resulting organism tissue concentration was recorded, and the fractional change in tissue concentration from the original RESRAD-BIOTA prediction was calculated. By comparing the fractional change in tissue concentration associated with each parameter, the sensitive parameters that account for most of the differences in RESRAD-BIOTA and FASTer predictions were identified. After each parameter was analysed, the values of all parameters were changed to FASTer input values to see how closely RESRAD-BIOTA reproduced FASTer predictions, thereby providing insights to the differences in RESRAD-BIOTA and FASTer calculation methodologies.

RESRAD-BIOTA was able to produce close results to FASTer predictions when FASTer input values were used. For ^{137}Cs activity concentrations in edible frogs, brown frogs, and root voles, the results produced by RESRAD-BIOTA and FASTer are within 3% of each other. Predictions of ^{90}Sr for bank voles, yellow necked mice, and wood mice, by the two models are within 12%, and those for Pu activity concentrations in great tit and long-tailed tit, the differences are within 13% of each other. Note because ^{240}Pu is not included in the RESRAD-BIOTA database, ^{239}Pu was used as a surrogate for ^{240}Pu. The following are the conclusions on the sensitive parameters drawn from the analyses:

(1) For ^{137}Cs for edible frogs and brown frogs, the most sensitive parameter is food ingestion rate. RESRAD-BIOTA predictions include consideration of radionuclide intake through inhalation and water ingestion, which are not included in FASTer predictions; however, both pathways have little contribution to the organism tissue concentration. In addition to inhalation and water ingestion, RESRAD-BIOTA predictions also included consideration of soil/sediment ingestion and ingestion of aquatic organisms (for edible frogs), which FASTer did not consider. Soil/sediment ingestion is the second most sensitive parameter that contributes to differences in

RESRAD-BIOTA and FASTer predictions. Biological loss rate and food sources are two other sensitive parameters.

(2) For ^{137}Cs in root voles, the most sensitive parameter that contributes to the differences in RESRAD-BIOTA and FASTer predictions is the food ingestion rate, followed by the biological loss rate parameter.

(3) For ^{90}Sr for bank voles, the differences in food source specifications contribute the most to the difference in tissue concentration predictions. FASTer considered ingestion of lichen and bryophytes for bank voles while RESRAD-BIOTA did not. Because the CR values for lichen and bryophytes are much larger (5.5 was assumed in the RESRAD-BIOTA sensitivity analyses; 8.68 was used in FASTer) than the CR for other food sources this, along with a greater food ingestion rate, results in FASTer giving a higher predicted whole-body ^{90}Sr activity concentrations for bank voles than RESRAD-BIOTA. In addition to food ingestion rate and food source, differences in life span and assimilation fraction also contribute to the difference in predictions for bank voles.

(4) For ^{90}Sr transfer to yellow necked mice and wood mice, differences in food source specifications contributed the most to the difference in tissue concentration predictions. Again, FASTer predicted higher tissue concentrations because it included lichen and bryophytes as a food source and assumed greater food ingestion rates. Differences in assumptions for mass and life span for wood mice also resulted in difference in predicted tissue concentrations.

(5) For the three rodent species (bank voles, yellow necked mice, and wood mice) for which sensitivity analyses was conducted, the ^{90}Sr biological loss rate is not a sensitive parameter, which is different to the case for ^{137}Cs. This is in part due to the fact that the biological loss rates assumed for ^{90}Sr by RESRAD-BIOTA and FASTer were similar, but is also due to the lower biological loss rates assumed for ^{90}Sr compared to ^{137}Cs. A smaller biological loss rate means longer biological decay half-life and less influence on tissue concentration.

(6) For Pu transfer to both tit species, the most sensitive parameter identified is biological loss rate, followed by food ingestion rate, soil ingestion rate, and assimilation fraction. Although the food source specifications for the RESRAD-BIOTA (which under predicted considerably) and FASTer predictions were different, they contributed little to the difference in tissue concentration predictions. This is because the CR's for the different food sources considered are not dramatically different from each other.

6.5.1.10. Scenario

The ability of the participating models to accurately predict whole-body activity concentrations in the species considered is dependent upon how appropriate the input soil data were (i.e. was it sufficiently replicated to encompass the likely heterogeneity, was it representative of the home range of the animals in question?). A similar caveat with respect to replication can be expressed about the available biota data (measured data were based upon samples sizes ranging from one to 65 animals).

All approaches assumed that the transfer of radionuclides to biota was consistent throughout the exclusion zone. In reality this will not be the case, some of the differing agreement between predictions and observations will be a consequence of site specific factors such as soil characteristics (see Sobotovich et al. [143]) and the contribution of 'hot particles' to the radionuclide deposit (potentially significant and variable within the exclusion zone).

6.5.2. Absorbed dose rates

Some of the variation between internal and total absorbed dose rates predicted by the different participating models can readily be explained by differences in the predicted whole-body activity concentrations. For instance, the comparatively high internal ^{90}Sr dose rate predicted for grassy vegetation by EA R&D128, the high internal Pu dose rate predicted for long-tailed tit by the ERICA Tool, the low internal dose rate predicted for the sand lizard by RESRAD-BIOTA, high internal dose rates predicted for rodent species by FASTer and the low ^{137}Cs internal dose rate predicted by EA R&D128.

Other aspects of the results can be related back to the findings of the comparison of DCC values (Chapter 2); for example the comparative lack of external dose from ^{90}Sr predicted by the ERCA Tool.

Given the relatively low contribution of external exposure to the total dose rate in this scenario, assumptions with regard to occupancy do not greatly influence comparisons between the predictions of the participating models.

There was reasonably good agreement between the TLD measurements of gamma dose rate and predicted external ^{137}Cs dose rates for five of the six possible comparisons. All of these data were from the study of Beresford et al. [140] conducted in 2005. This paper also provided the justification for (i) the value used to correct the TLD readings (to remove the beta dose contribution) and (ii) assuming that the external gamma dose rate could be equated the external ^{137}Cs dose rates predicted by the models. However, the other available dataset, which was not predicted as well, originated from a study conducted in the mid-1990's. It is therefore likely that the correction factor used and the assumed dominance of ^{137}Cs may not have been applicable to this dataset.

The most notable aspect of the predicted dose rates is that overall they appear to be less variable between models than may have been expected from the high variability observed in predictions of activity concentration. A good example of this is the total absorbed dose rate predictions of EA R&D128 for rodent species. These are comparable to those for the other models. However, this model under predict ^{137}Cs whole-body activity concentrations by more than one order of magnitude consistently predicting lower that all the other participating models by up to three orders of magnitude (see Figure 6.3). Conversely, the model tended to overestimate ^{90}Sr whole-body activity concentrations (see Figure 6.6). Therefore, the overestimation of ^{90}Sr whole-body activity concentrations and underestimation of ^{137}Cs appear to 'balance out' to produce a total dose rate estimate comparable to that of the other models.

CHAPTER 7. DISCUSSION

A total of 15 models have participated in the four exercises conducted by the BWG. These have included those available to any interested user (RESRAD-BIOTA, ERICA, EA R&D128 and FASSET[9]) and in-house models being used/developed by several of the BWG participants. Participants have included modellers, regulators, industry and researchers.

7.1. Dosimetry and transfer components of the models

The exercise to compare the calculation of unweighted whole-body absorbed dose rates (reported as DCCs) for a selection of the proposed ICRP Reference Animal geometries demonstrated that all the 11 participating approaches generally estimated comparable internal dose rates even though different assumptions (including the use of default geometry DCCs, rather than estimation of bespoke values for this exercise) were made (Chapter 3). The notable exception was as a consequence of different daughter products being included (e.g. one approach included ^{234}U in the estimation of the DCC for ^{238}U). Variation was greater for the estimation of external dose rates most notably for short-range α- or β-radiation (e.g. from ^{3}H, plutonium and some naturally-occurring radionuclides). However, it is generally accepted that external exposure of biota by such emitters is of little radiological significance, due to their low range in matter.

The comparison of predicted activity concentrations in a range of freshwater and terrestrial biota by eight of the participating models assuming 1 Bq per unit media demonstrated considerably more variability than the comparison of unweighted dose estimates (Chapter 4). For many radionuclide-reference organism combinations, variability in predictions was by three or more orders of magnitudes. Predictions were often most variable for poorly studied organisms such as carnivorous mammals, fish eggs, bird eggs, ducks, amphibians and aquatic mammals. Some of the more extreme variability could be explained by the use of 'guidance' methodology to provide values by a number of approaches when there were no available data from which to derive CR values. However, this guidance methodology, used within EA R&D128, FASSET and the ERICA Tool, is intended to be conservative if data are missing and in most instances, it resulted in comparatively high (and hence conservative) predictions.

7.2. Scenario applications

Two scenarios conducted have allowed model predictions to be compared to measured whole-body activity concentration data for a range of freshwater and terrestrial biota. In the case of both scenarios, the majority of the models predicted activity concentrations in most organism types to within an order of magnitude.

The results of the scenarios were largely in agreement with those of the earlier exercises to compare the dosimetry and transfer components of the models. The understanding of the different models gained in the early phases of the work aided interpretation of poor predictions and variability between the models.

The variability between the participating models in estimated dose rates could largely be explained by the variability in predicted whole-body activity concentrations. Surprisingly, for the terrestrial scenario, there was less variability observed in the estimated total dose rates (typically less than an order of magnitude) than may have been anticipated from observed

[9]All four of these models are available from the internet see http://www.ceh.ac.uk/protect/pages/env_protect_radio.html

variation in predicted activity concentrations (typically at least three orders of magnitude). This was the consequence of given models under-estimating for one radionuclide whilst over-predicting for another for the same organism and hence balancing out the overall prediction of dose. Total dose rate alone is therefore a poor prediction to compare in any model inter-comparison exercises.

External dose rate generally contributed little to the overall total dose in the two scenarios considered. Therefore, differences in assumptions with regard to occupancy contributed little to the overall variation in estimated dose rates. However, assumptions with regard to diet, and CR values used to predict the activity concentration in dietary components, were responsible for variation observed between those participating models which use food chain approaches rather than simple biota-media CR values.

The scenarios also allowed comparison of the predictions of simple CR-based approaches with more complex food-chain models under equilibrium conditions. Overall, when applied within their intended limitations, the two approaches compared favourably.

The freshwater scenario also demonstrated that models which accounted for water chemistry (i.e. relevant stable element concentrations) generally produced better predictions than those applying a generic CR value only. For Perch Lake, two of the models which take into account water chemistry better predicted the transfer of ^{90}Sr to fish. However, models parameterised using laboratory studies often under-predicted biota activity concentrations.

Being able to compare predictions to observed data ultimately leads to the question what is a good prediction? In the two scenarios, this was partially assessed statistically using Z-scores. Poor predictions were identified as those which deviated from the data mean by three or more times the standard deviation of the data. However, care was needed in the interpretation of these as for observed datasets with large standard deviations (>33% of the mean), the Z-score identifies over-predictions but not under-predictions. Hence, to identify outlying predictions the Z-scores were used in combination with an assessment of all predictions falling outside of an order of magnitude above or below the data mean. Whilst recognising that an order of magnitude variation may not be acceptable to regulators/industries who may use the models tested here, this level of agreement is pragmatic considering the inherent variability in the measured data and the values used to parameterise the models.

7.3. Recommendations for the future

The need for a system to protect the environment from ionising radiation is now generally recognised and environmental protection is referred to in the draft revision of International Atomic Energy Agency (IAEA) Basic Safety Standards and new recommendations of the ICRP [1]. However, many aspects including the discussion of protection goals, agreement of benchmark values and parameterisation of models applied in the work described here are still under development.

An aim of the BWG was to improve the models used by Member States. The collaborative exercises led to the sharing of parameters and re-parameterisation by some of the participating models (e.g. see discussion of the ERICA Tool in Chapter 4 and DosDiMEco in Chapter 6). However, the model-model inter-comparisons and the scenario applications only compared a limited number of radionuclides. Additionally, whilst the scenarios considered sites for which extensive databases were available, these may not be typical of situations needing to be assessed within regulatory frameworks (especially Chernobyl).

7.3.1. Recommendations

In the near future there are a number of international developments which will have an impact on the field of radiological protection:

- The ICRP will deliver its framework for assessing environmental protection;
- The UNSCEAR will report on its review of effects data;
- The EC EURATOM project PROTECT will make recommendations on protection goals and numeric benchmark values.

These are all likely to impact upon any future activities of the BWG. Bearing this in mind, and to aid the IAEA in the development of future guidance on the assessment of biota, a suggested future direction of the BWG is outlined below.

7.3.2. Transfer parameters

The work of the BWG has clearly demonstrated that the largest contribution to variability between model predictions, and comparison with available data, is the parameterisation of their transfer components. Other works are in agreement with this conclusion [29, 117]. Additionally, whilst some of the available models include probability distributions for their transfer parameters, there has been no consideration of how these are being used.

There is a clear need to better share knowledge on the transfer of radionuclides to biota and provide authoritative collations of those data which are available. It is suggested that a document for biota which is equivalent to the IAEA handbook on transfer parameters for human food chains [34][10] should be produced. This activity should attempt to bring in experts from outside the BWG who have useful data and are willing to collaborate. Furthermore, the output of this compilation will be of value to ICRP Committee 5 in their development of a framework for assessing the exposure of non-human biota to radiation.

7.3.3. ICRP framework

The outputs of the ICRP will clearly be something which the BWG should evaluate in any future scenario applications and model comparisons.

7.3.4. Future scenarios

The scenario applications described above have provided a useful first step in the evaluation of the participating models. However, future scenarios should be more focussed to consider situations which regulators/industry need to consider. Potential examples are assessment of waste repositories and TeNORM sites; the latter would benefit from collaboration with the TeNORM working group utilising scenarios which they are developing.

Such scenarios would also enable the comparison of the available approaches within a regulatory context evaluating the various tiers of assessment (from screening through to detailed) which the more comprehensive approaches contain. The PROTECT project has begun to address this by comparing screening level assessments of the three readily available models (RESRAD-BIOTA, ERICA Tool and EA R&D128). Assessing a freshwater scenario, the three models all predicted their initial screening criteria would be exceeded, but they each

[10] IAEA [34] is being updated by a separate working group within the EMRAS programme.

identified a different radionuclide and organism as being the dominant contributor to dose [144]. This clearly demonstrates that there is a requirement to compare the available approaches beyond simple comparisons of predicted activity concentrations.

The participation of users other than those involved in their development should be encouraged in these scenario applications. For instance, the application of RESRAD-BIOTA to the Perch Lake scenario by two groups new to the tool produced results, in some cases, which varied by up to four orders of magnitude.

7.3.5. Effects data

The models used by the BWG predict dose rates to biota, but there is also a need to be able to determine the potential consequences of predicted dose rates. A large amount of data on the effects of ionising radiation on biota has recently been collated into the FREDERICA data base [145]; www.frederica-online.org). This compilation can be used to aid decision-making on the potential impact of the predicted exposure to ionising radiation [146, 147].

However, the effects data available in the FREDERICA database covers only a proportion of the available scientific literature. Furthermore, to be of most use to decision makers, there is a need to better evaluate the quality of much of these data to ensure that they are applicable. It is suggested that this could be best achieved through a subgroup of the BWG. The outputs of this subgroup would be useful to the ICRP Committee 5 and UNSCEAR.

Whilst approaches from chemical assessments (such as species sensitivity distributions) are being adopted in trying to define dose rate benchmarks for biota, these do not really inform us of the actual potential impact on a given species or ecosystem. It is suggested that the BWG should consider how population modelling techniques (from other fields) might be applied to aid setting thresholds against which the degree of environmental protection can be determined.

REFERENCES

[1] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Recommendations of the International Commission on Radiological Protection, Annals of the ICRP 37, 2-3, Pergamon Press, Oxford (2007).

[2] UNITED STATES DEPARTMENT OF ENERGY, Radiation protection of the public and the environment, DOE order 5400.5, United States Department of Energy, Washington D.C., (1993), Available from: http:\\www.directives.doe.gov/pdfs/doe/doetext/oldord/5400/o54005c2.pdf

[3] UNITED STATES DEPARTMENT OF ENERGY, Environmental protection program, DOE Order 450.1, United States Department of Energy, Washington D.C. (2003), Available from: http://www.directives.doe.gov/pdfs/doe/doetext/neword/450/o4501.pdf

[4] BERESFORD, N.A., BROWN, J., COPPLESTONE, D., GARNIER-LAPLACE, J., HOWARD, B.J., LARSSON, C-M., OUGHTON, D., PRÖHL, G., ZINGER, I., D-ERICA: An Integrated approach to the assessment and management of environmental risks from ionising radiation, Deliverable of the ERICA project (FI6R-CT-2004-508847), Swedish Radiation Protection Authority, Stockholm, Sweden, (2007), Available from: http://www.ceh.ac.uk/PROTECT/ERICA deliverables.html

[5] COPPLESTONE, D., BIELBY, S., JONES, S.R., PATTON, D., DANIEL, P., GIZE, I., Impact assessment of ionising radiation on wildlife, R&D Publication 128, ISBN 1 85705590, Environment Agency, Bristol, (2001), Available from: http://www.coger.org.uk/RD128/R&DPublication128updated2002.PDF

[6] COPPLESTONE, D., WOOD, M.D., BIELBY, S., JONES, S.R., VIVES, J., BERESFORD, N.A., Habitat regulations for Stage 3 assessments: radioactive substances authorisations, EA R&D Technical Report P3-101/SP1a, Environment Agency, Bristol, (2003), Available from: HTTP://WWW.CEH.AC.UK/PROTECT/pages/documents/Habitatsregulationsforstage3assessment.pdf

[7] UNITED STATES DEPARTMENT OF ENERGY, A Graded approach for evaluating radiation doses to aquatic and terrestrail biota, Technical standard DOE-STD-1153-2002, , USDOE, Washington D.C., (2002).

[8] COPPLESTONE, D., WOOD, M.D., MERRILL, P.C., ALLOTT, R., JONES, S.R., VIVES, J., BERESFORD, N.A., ZINGER, I., Impact assessment of ionising radiation on wildlife: meeting the requirements of the EU birds and habitat directives, Radioprotection 40 (2005), S893-S898.

[9] SMITH, K., ROBINSON, C., IKONEN, A.T.K., Development and application of non-human biota assessment methodologies to determine potential environmental effects from a proposed nuclear waste repository, International conference on environmental radioactivity: From measurements and assessment to regulation, IAEA-CN-145, 23-27 April, 2007, Vienna, International Atomic Energy Authority, Vienna (2007) 349-350 pp.

[10] JONES, D.S., SCOFIELD, P.A., Implementation and validation of a DOE standardised screening method for evaluating radiation impacts to biota at long-term stewardship sites, ORNL/TM-2003/76, Oak ridge National Laboratory, Oak Ridge (2003).

[11] BERESFORD, N.A., HOWARD, B.J., BARNETT, C.L., (Eds.), Application of ERICA Integrated Approach at case study sites, Deliverable 10 of the ERICA project, Contract number FI6R-CT-2004-508847, Centre for Ecology and Hydrology, Lancaster (2007) Centre for Ecology and Hydrology, Lancaster. Available from: http://www.ceh.ac.uk/PROTECT/ERICAdeliverables.html

[12] VIVES I BATLLE, J., JONES, S.R., GOMEZ-ROS, J.M., A method for calculation of dose per unit concentration values for aquatic biota, Journal of Radiological Protection 24 (2004), A13-A34.
[13] ULANOVSKY, A., PROHL, G., A practical method for assessment of dose conversion coefficients for aquatic biota, Radiation and Environmental Biophysics 45 (2006), 203-214.
[14] YANKOVICH, T.L., BEATON, D., Concentration ratios of stable elements measured in organs of terrestrial, freshwater and marine non-human biota for input into internal dose assessment for PSL-2: A literature review (2000), AECL, Chalk River Laboratories, 120 pp.
[15] KRANE, K.S., Introductory Nuclear Physics, John Wiley and Sons Inc., New York (1988).
[16] INTERNATIONAL ATOMIC ENERGY AGENCY, Methods of dosimetry for aquatic organisms, Methodology for assessing impacts of radioactivity on aquatic ecosystems, Technical Reports Series No. 190, IAEA, Vienna (1979).
[17] BLAYLOCK, B.G., FRANK, M.L., O'NEAL, B.R., Methodology for estimating radiation dose rates to freshwater biota exposed to radionuclide in the environment, Report ES/ER/TM-78, Oak Ridge National Laboratory, Tennessee (1993).
[18] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, The concept and use of reference animals and plants for the purposes of environmental protection, Annals of the ICRP, Draft for discussion, (2005), (see http://www.icrp.org/remissvar/listcomments.asp?search=&how=0&recommendation=%7B19DE3262-2E66-4325-BF49-FF81A8F971CC%7D&Submit1=Search).
[19] ENVIRONMENT CANADA, Ecological risk assessment of priority substances under the Canadian Environmental Protection Act, Resource document, Draft 1.0, Environment Canada, (1996).
[20] ENVIRONMENT CANADA, Environmental assessment of priority substances under the Canadian Environmental Protection Act, Guidance manual Version 1.0, Environment Canada, (1997).
[21] HIGLEY, K.A., DOMOTOR, S.L., ANTONIO, E.J., A kinetic-allometric approach to predicting tissue radionuclide concentrations for biota, Journal of Environmental Radioactivity 66 (2003), 61-74.
[22] BROWN, J.E., ALFONSO, B., AVILA, R., BERESFORD, N.A., COPPLESTONE, D., PRÖHL, G., ULANOVSKY, A., The ERICA Tool, Journal of Environmental Radioactivity 99 (2008), 1371-1383.
[23] BERESFORD, N.A., BARNETT, C.L., HOWARD, B.J., SCOTT ,W.A., BROWN, J.E., COPPLESTONE, D., Derivation of transfer parameters for use within the ERICA Tool and the default concentration ratios for terrestrial biota, Journal of Environmental Radioactivity 99 (2008), 1393-1407.
[24] HOSSEINI, A., THØRRING, H., BROWN, J. E., SAXÉN, R., ILUS, E., Transfer of radionuclides in aquatic ecosystems – Default concentration ratios for aquatic biota in the Erica Tool, Journal of Environmental Radioactivity 99 (2008), 1408-1429.
[25] ULANOVSKY, A., PRÖHL, G., GÓMEZ-ROS, J.M., Methods for calculating dose conversion coefficients for terrestrial and aquatic biota, Journal of Environmental Radioactivitiy 99 (2008), 1440-1448.
[26] BROWN, J., STRAND, P., HOSSEINI, A., BØRRETZEN, P., (Eds.), FASSET: Handbook for assessment of the exposure of biota to ionising radiation from radionuclides in the environment, Deliverable 5 of the EC 5th Framework Programme FASSET, Contract FIGE-CT-2000-00102, Norwegian Radiation Protection Authority, Østerås, Norway (2003), Available from: http://www.erica-project.org/

[27] LARSSON, C.-M., JONES, S.R., GOMEZ- ROS, J.M., ZINGER, I., Framework for assessment of environmental impact of ionising radiation in major European ecosystems, Deliverable 6 for the FASSET project Contract No. FIGE-CT-2000-00102, Swedish Radiation Protection Authority, Stockholm (2004), Available from: http://www.erica-project.org/
[28] WILLIAMS C. (Ed.)., Framework for assessment of environmental impact (FASSET) of ionising radiation in European ecosystems, Journal of Radiological Protection 24 (2004).
[29] AVILA, R., BERESFORD, N.A., AGÜERO, A., BROED, R., BROWN, J., IOSPJE, M., ROBLES, B., SUAÑEZ, A., Study of the uncertainty in estimation of the exposure of non-human biota to ionizing radiation, Journal of Radiological Protection 24 (2004), A105–A122.
[30] NEDVECKAITE, T., FILISTOVIC, V., MARCIULIONIENE, D., KIPONAS, D., REMEIKIS, V., BERESFORD, N.A., Exposure of biota in the cooling pond of Ignalina NPP: hydrophytes, Journal of Environmental Radioactivity 97 (2007), 137-147.
[31] AMERICAN NUCLEAR SOCIETY, Gamma-ray attenuation coefficients and buildup factors for engineering materials, ANSI/ANS-6.4.3-1991 (1992).
[32] HUBBELL, H., SELTZER, S.M., Tables of X-Ray mass attenuation coefficients and mass energy-absorption coefficients, Ionizing Radiation Division, Physics Laboratory, National Institute of Standards and Technology Gaithersburg, MD 20899 (1996).
[33] GARTEN, C.T. (Jnr.), DAHLMAN, R.C., Plutonium in biota from an east Tennessee floodplain forest, Health Physics 34 (1978), 705-712
[34] INTERNATIONAL ATOMIC ENERGY AGENCY, Handbook of parameter values for the prediction of radionuclide transfer in temperate environments, IAEA Technical Reports Series No. 364, IAEA, Vienna (1994).
[35] LINSALATA, P., MORSE, R., FORD, H., EISENBUD, M., FRANCA, E.P., De CASTRO, M.B., LOBAO, N., SACHETT, I., CARLOS, M., Transport pathways of Th, U, Ra and La from soil to cattle tissues, Journal of Environmental Radioactivity 10 (1989), 115-140.
[36] MARTINEZ-AGUIRRE, A., GARCIAORELLANA, I., GARCIALEON, M., Transfer of natural radionuclides from soils to plants in a marsh enhanced by the operation of non-nuclear industries, Journal of Environmental Radioactivity 35 (1997), 149-171.
[37] RADHAKRISHNA, A.P., SOMASHEKARAPPA, H.M., NARAYANA, Y., SIDDAPPA, K., Distribution of some natural and artificial radionuclides in Mangalore environment of south-India, Journal of Environmental Radioactivity 30 (1996), 31-54.
[38] SAMPLE, B.E., APLIN, M.S., EFROYMSON, R.A., SUTER II, G.W., WELSH, C.J.E., Methods and tools for estimation of the exposure of terrestrial wildlife to contaminants, ORNL/TM-13391, Oak Ridge National Laboratory, Oak Ridge TN (1997).
[39] SANTCHI, P.H., HONEYMAN, B.D., Radionuclides in aquatic environments, Radiation physics and chemistry 34 (1989), 213-240.
[40] SWEECK, L., TH., Z., VOLCKAERT, G., VANDECASTEELE, C., Geologische berging van geconditioneerd langlevend hoog radioactief afval – Biosfeerparameters in performantie- en veiligheidsanalyse Deel II, SCK•CEN Report R-3194, SCK•CEN, Mol, (1998).
[41] NAGY, K.A., Field metabolic rate and food requirement scaling in mammals and birds, Ecological Monographs 57 (1987), 111-128.
[42] ARGONNE NATIONAL LABORATORY, Polonium human health fact sheet (2005a), Available from: http://www.evs.anl.gov/pub/doc/polonium.pdf

[43] ARGONNE NATIONAL LABORATORY, Technetium human health fact sheet, (2005b), Available from: http://www.evs.anl.gov/pub/doc/technetium.pdf
[44] ARGONNE NATIONAL LABORATORY, Radium human health fact sheet, (2005c), Available from: http://www.evs.anl.gov/pub/doc/radium.pdf
[45] BERESFORD, N.A., MAYES, R.W., COOKE, A.I., BARNETT, C.L., HOWARD, B.J., LAMB, C.S., NAYLOR, G.P.L., The importance of source dependent bioavailability in determining the transfer of ingested radionuclides to ruminant derived food products, Environmental Science and Technology 34 (2000), 4455-4462.
[46] COUGTHREY, P.J., THORNE, M.C., Radionuclide distribution in terrestrial and aquatic ecosystems – a critical review of data, Volume 1, A.A Balkema, Rotterdam (1983).
[47] COUGHTREY, P.J., JACKSON, D., THORNE, M.C., Radionuclide distribution and transport in terrestrial and aquatic ecosystems, Volume 3, A.A Balkema, Rotterdam (1983).
[48] COUGHTREY, P.J., JACKSON, D., JONES, C.H., KANE, P., THORNE, M.C., Radionuclide distribution and transport in terrestrial and aquatic ecosystems, Volume 4, A.A Balkema, Rotterdam (1984).
[50] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Limits for intakes of radionuclides by workers, ICRP Publication 30, Annals of the ICRP, 2, 3/4, Part 1, Pergamon Press, Oxford (1979).
[51] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Age-dependent doses to members of the public from intake of radionuclides: Part 3 – Ingestion dose coefficients, Annals of the ICRP, 25, Pergamon Press, Oxford (1995).
[52] TAYLOR, G.N., JONES, C.W., GARDNER, P.A., LLOYD, R.D., MAYS, C.W., CHARRIER, K.E., 2 new rodent models for actinide toxicity studies, Radiation Research 86 (1981), 115-122.
[53] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Limits for intakes of radionuclides by workers – part 3, ICRP Publication 30, Annals of the ICRP, 6, Pergamon Press, Oxford (1981).
[54] SAZYKINA, T.G., ECOMOD – An ecological approach to radioecological modelling, Journal of Environmental Radioactivity 50 (2000), 207-220.
[55] KRYSHEV, A.I., RYABOV, I.N., A dynamic model of Cs-137 accumulation by fish of different age classes, Journal of Environmental Radioactivity 50 (2000), 221-233.
[56] KRYSHEV, A.I., Modelling the accumilation of ^{137}Cs by age-structured fish population Radioprotection – Colloques 37, C1 (2002), 627-632.
[57] KRYSHEV, A.I., Model reconstruction of Sr-90 concentrations in fish from 16 Ural lakes contaminated by the Kyshtym accident of 1957, Journal of Environmental Radioactivity 64 (2003), 67-84.
[58] KRYSHEV, A.I., Evaluation of the biological transfer of ^{32}P, ^{137}Cs and ^{65}Zn by fish in the Yenisei River, Science of the Total Environment 322 (2004), 191-207.
[59] WINBERG, G.G., Metabolism rate and feeding needs of fish, Belarus State University, Minsk (1956) 254 pp. (in Russian).
[60] BEAUGELIN-SEILLER, K., JASSERAND, F., GARNIER-LAPLACE, J., GARIEL, J.C., EDEN: Software to calculate the dose rate of energy for the non-human biota, due to the presence of radionuclides in the environment, Environmental studies 11 (2004), 87-96.
[61] BEAUGELIN-SEILLER, K., GARNIER-LAPLACE, J., GARIEL, J.C., JASSERAND, F., E.D.E.N.: a tool for the estimation of dose coefficient equivalents for non-human biota, Radioprotection 40 (2005), S921-S926.

[62] BEAUGELIN-SEILLER, K., JASSERAND, F., GARNIER-LAPLACE, J., GARIEL, J.C., Modelling the radiological dose in non-human species: principles, computerization and application, Health Physics 90 (2006), 485-493.
[63] ORGANISATION FOR ECONOMIC CO-OPERATION AND DEVELOPMENT, JEF-PC version 2.0, OECD and Nuclear Energy Agency (NEA), (1997).
[64] BEAUGELIN-SEILLER.K, B.P., GARNIER-LAPLACE, J., ADAM.C, CASTEAUR: A simple tool to assess the transfer of radionuclides in waterways, Health Physics 83 (2002), 539-542.
[65] BOYER, P., BEAUGELIN-SEILLER, K., Casteaur: a tool for operational assessments of radionuclide transfers in river ecosystems Radioprotection – Colloques 37 (2002), 1127-1131.
[66] DUCHESNE, S., BOYER, P., BEAUGELIN-SEILLER, K., Sensitivity and uncertainty analysis of a model computing radionuclides transfers in fluvial ecosystems (CASTEAUR): Application to ^{137}Cs accumulation in chubs, Ecological Modelling 166 (2003), 257-276.
[67] BOYER, P., BEAUGELIN-SEILLER, K., TERNAT, F., ANSELMET, F., AMIELH, M.A., dynamic box model to predict the radionuclide behaviour in river for average and long-term periods, Radioprotection 40 (2005), S307-S313.
[68] MONTE, L., BOYER, P., BRITTAIN, J.E., HÅKANSON, L., LEPICARD, S., SMITH, J.S., Review and assessment of models for predicting the migration of radionuclides through rivers, Journal of Environmental Radioactivity 79 (2005), 273 - 296.
[69] GOLIKOV, V., BROWN, J.E., Internal and external dose models, Deliverable 4 for EC Inco-Copernicus project EPIC, Contract number CA2-CT-2000-10032, Norwegian Radiation Protection Authority, Østerås, Norway (2003) 52 pp.
[70] BROWN, J.E., THØRRING, H., HOSSEINI, A., The "EPIC" impact assessment framework: Towards the protection of the Arctic environment from the effects of ionising radiation, Deliverable 6 for EC Inco-Copernicus project EPIC, Contract number CA2-CT-2000-10032, Norwegian Radiation Protection Authority, Østerås (2003), Available from: http://www.ceh.ac.uk/PROTECt/ EPICdeliverables.html
[71] LOEVINGER, R., JAPHA, E.M., BROWNELL, G.L., Discrete radioisotope sources, Radiation Dosimetry, (HINE, G.J., BROWNELL, G.L., (Eds.), Academic Press, New York (1956) 693-799.
[72] WOODHEAD, D.S., Environmental dosimetry: The current position and the implications for developing a framework for environmental protection, Environment Agency, Bristol (2000).
[73] INTERNATIONAL ATOMIC ENERGY AGENCY, Methodology for assessing impacts of radioactivity on aquatic ecosystems, IAEA, Vienna (1979).
[74] GOLIKOV, V, BARKOVSKI, A, KULIKOV, V, BALONOV, M, RANTAVAARA, A, VETIKKO, V., Gamma ray exposure due to sources in the contaminated forest, Contaminated Forests, (LINKOV, I., SHELL, W.R., (Eds.)), Kluwer Academic Publishers, The Netherlands (1999) 333-341.
[75] BERESFORD, N.A., WRIGHT, S.M., BROWN, J.E., SAZYKINA, T., Review of approaches for the estimation of radionuclide transfer to reference Arctic biota: Transfer and uptake models for reference Arctic organisms, Centre for Ecology and Hydrology – Lancaster, (2003) 75 pp.
[76] AMIRO, B.D., Radiological dose conversion factors for generic non-human biota used for screening potential ecological impacts, Journal of Environmental Radioactivity 35 (1997), 37-51.
[77] SMITH, J., Effects of ionising radiation on biota: do we need more regulation?, Journal of Environmental Radioactivity 82 (2005), 105-122.

[78] LOWE, V.P.W., HORRILL, A.D., Caesium concentration factors in wild herbivores and the Fox (*Vulpes-Vulpes*), Environmental Pollution 70 (1991), 93-107.
[79] SMITH, J.T., BERESFORD, N.A., Chernobyl – catastrophe and consequences, Praxis Publishing/Springer, Chichester (2005).
[80] AGENCY FOR TOXIC SUBSTANCES AND DISEASE REGISTRY, Toxicological profiles – cobalt, ATSDR, (2006), Available from: http://www.atsdr.cdc.gov/toxprofiles/tp33-c6.pdf
[81] BLAYLOCK, B.G., Radionuclide data bases available for bioaccumulation factors for freshwater biota, Nuclear Safety, 23, (1982), 427-438.
[82] SMITH, J.T., Modelling the dispersion of radionuclides following short duration releases to rivers – Part 2. Uptake by fish, Science of the Total Environment 368 (2006), 502-518.
[83] ROWAN, D.J., RASMUSSEN, J.B., Bioaccumulation of radiocaesium by fish: the influence of physicochemical factors and trophic structure, Canadian Journal of Fish Aquatic Science 51 (1994), 2388-2410.
[84] SMITH, J.T., KUDELSKY, A.V., RYABOV, I.N., HADDERINGH, R.H., Radiocaesium concentration factors of Chernobyl-contaminated fish: a study of the influence of potassium, and "blind" testing of a previously developed model, Journal of Environmental Radioactivitiy 48 (2000), 359-369.
[85] NICHOLSON, S., MACKENZIE, J., The remobilisation of radionuclides from marine sediments; implication for the collective dose assessment, SRD Report R 453, Safety and Reliability Directorate, UKAEA (1988).
[86] DE VRIES, W., Accumulation of heavy metals in organisms, bioaccumulation in pikeperch, data analysis for Lake IJsselmeer, Ketelmeer and Markermeer, Delft Hydraulics, (1989), (In Dutch).
[87] [INTERNATIONAL ATOMIC ENERGY AGENCY, Modelling of radiocaesium from deposition to lake ecosystems, IAEA Tech. Doc. 1143, IAEA, Vienna (2000).
[88] POPOV, A.G., HELING, R., Aquatic transfer models to predict wash-off from watersheds and the migration of radionculides in lakes for implementation, in RODOS, Proceedings of the First International Conference of the European Commission, 18-22 March 1996, Russian Federation and Ukraine on the radiological consequences of the Chernobyl accident, Minsk, Belarus (1996).
[89] HELING, R., Off site emergency planning support, The validation of the lake ecosystem model LAKECO, report Nr. 40901-NUC 95-9113, KEMA, Arnhem (1996).
[90] HELING, R., LAKECO: Modelling the transfer of radionuclides in a lake ecosystem, Radiation Protection Dosimetry 73 (1997), 191-194.
[91] ZHELEZNYAK, M., HELING, R., RASKOB, W., POPOV, A., BORODIN, R., GOFMAN, D., LYASHENKO, G., MARINETS, A., POKHIL, A., SHEPELEVA, T., TKALICH, P., Modelling the hydrological pathways in RODOS, Proceedings of the first International conference of the European commission, Belarus, Russian Federation and Ukraine on the radiological consequences of the Chernobyl accident, Minsk, 18-22 March, (1996).
[92] KRYSHEV, I.I., SAZYKINA, T.G., HOFFMAN, O., THIESSEN, K.M., BLAYLOCK, B.G., FENG, Y., GALERIU, D., HELING, R., KRYSHEV, A.I., KONONOVICH, A.L., WATKINS, B., Assessment of the consequences of the radioactive contamination of aquatic media and biota for the Chernobyl NPP cooling pond: model testing using Chernobyl data, Journal of the Environmental Radioactivity 42 (1999), 143-156.
[93] HELING, R., NICULAE, C., The MOIRA project: The radiostrontium uptake model, testing Burn98 on freshwater ecosystems, NRG report P20071/99.55597/P, NRG, Arnhem (1999).

[94] SCOTT, E.M., GURBUTT, P., HARMS, I., HELING, R., NIELSEN, S.P., OSVATH, I., PRELLER, R., SAZYKINA, T., WADA, A., SJOEBLOM, K.L., Benchmarking of numerical models describing the dispersion of radionuclides in the Arctic Seas, Science of the Total Environment 202 (1997), 123-134.
[95] LEPICARD, S., HELING, R., MADERICH, V., POSEIDON/RODOS for radiological assessment of marine environment after accidental releases: application to coastal areas of the Baltic, Black and North Seas, IRE Conference, Proceedings of International Symposium, In situ measurements as a tool for radioecology, 10-12 June 2002, Fleurus, Belgium, (2002).
[96] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Report of the task group on reference man, ICRP (1975).
[97] INTERNATIONAL ATOMIC ENERGY AGENCY, Effects of ionizing radiation on plants and animals at levels implies by current radiation protection standards, IAEA, Vienna (1992).
[98] VIVES I BATLLE, J., BALONOV, M., BEAUGELIN-SEILLER, K., BERESFORD, N.A., BROWN, J., CHENG, J-J., COPPLESTONE, D., DOI, M., FILISTOVIC, V., GOLIKOV, S., HORYNA, J., HOSSEINI, A., HOWARD, B.J., JONES, S.R., KAMBOJ, S., KRYSHEV, A., NEDVECKAITE, T., OLYSLAEGERS, G., PRÖHL, G., SAZYKINA, T., ULANOVSKY, A., VIVES-LYNCH, S., YANKOVICH, T., YU, C.I., Inter-comparison of unweighted absorbed dose rates for non-human biota, Radiation Environmental Biophysics 46 (2007) 349-373.
[99] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Radionuclide transformations – energy and Intensity of transmissions, Annals of the ICRP, 11, Pergamon Press, Oxford (1983).
[100] R DEVELOPMENT TEAM, R: A language and environment for statistical computing, ISBN 3-900051-07-0, R Development Core Team, R Foundation for Statistical Computing, Vienna (2006), Available from: http://www.R-project.org
[101] KOMSTA, L., Moments: Moments, skewness, kurtosis and related tests, R package version 0.1, (2005), Available from: http://www.r-project.org and http://www.komsta.net/
[102] PROPHET, PROPHET StatGuide: Possible alternatives if your data violate normality test assumptions, (1997), Available from: http://www.basic.northwestern.edu/statguidefiles/n-dist_alts.html
[103] THOMPSON, M., WOOD, R., International harmonized protocol for proficiency testing of (chemical) analytical laboratories, Journal of Pure and Applied Chemistry 65 (1993), 2123-2144.
[104] LAWN, R.E., THOMPSON, M., WALKER, R.F., Proficiency testing in analytical chemistry, The Royal Society of Chemistry, London (1997) 110 pp.
[105] INTERNATIONAL ORGANISATION FOR STANDARDISATION, Proficiency testing by inter-laboratory comparisons Part 1: Development and operation of proficiency testing schemes, ISO/IEC Guide 43-1, ISO, Geneva (1997).
[106] POVINEC, P.P., Developments in analytical technologies for marine radionuclide studies, Marine Radioactivity, Radioactivity in the Environment No. 6, (LIVINGSTON, H.D., (Ed.)), Pergamon Press, Oxford (2004) 237-294.
[107] KOCHER, D.C., SJOREEN, A.L., Dose-Rate conversion factors for external exposure to photon emitters in soil, Health Physics 48 (1985), 193-205.
[108] ECKERMAN, K.F., RYMAN, J.C., External exposure to radionuclides in air, water, and soil, Federal Guidance Report No. 12, EPA-402-R-93-081, Oak Ridge National Laboratory, Oak Ridge (1993).
[109] KAMBOJ, S., LEPOIRE, D., YU, C., External exposure model in the RESRAD computer code, Health Physics 82 (2002), 831-839.

[110] BERESFORD, N.A., BARNETT, C.L., BROWN, J., CHENG, J-J., COPPLESTONE, D., FILISTOVIC, V., HOSSEINI, A., HOWARD, BJ., JONES, S.R., KAMBOJ, S., KRYSHEV, A., NEDVECKAITE, T., OLYSLAEGERS, G., SAXÉN, R., SAZYKINA, T., VIVES I BATLLE, J., VIVES-LYNCH, S., YANKOVICH, T., YU, C., Inter-comparison of models to estimate radionuclide activity concentrations in non-human biota, Radiation Environmental Biophysics (2008), DOI 10.1007/s00411-008-0186-8.
[111] SHEPPARD, S.C., Application of the international Union of radioecolgists soil-to-plant database to Canadian settings, AECL-11474, AECL, Pinawa (1995).
[112] BROWN, J., STRAND, P., ALI HOSSEINI, A., BØRRETZEN, P. (Eds.), Handbook for Assessment of the Exposure of Biota to Ionising Radiation from Radionuclides in the Environment. Deliverable 5: Appendix 2, Underpinning scientific information (Life history sheets, empirical data and models) of the FASSET project, EC Contract No. FIGE-CT-2000-00102, (2003) Available from: http://www.ceh.ac.uk/PROTECt/FASSETdeliverables.html
[113] COPPLESTONE, D., JOHNSON, M.S., JONES, S.R., TOAL, M.E., JACKSON, D., Radionuclide behaviour and transport in a coniferous woodland ecosystem: vegetation, invertebrates and wood mice, *Apodemus sylvaticus*, The Science of the Total Environment 239 (1999), 96-109.
[114] LIDE D.R. (Ed.), Handbook of Chemistry and Physics, CRC Press, Cleveland, Ohio (2003) pp.
[115] BERESFORD, N.A., HOWARD, B.J. (Eds.), Application of FASSET framework at case study sites, Deliverable 9 for the ERICA project, EU Contract No. FI6R-CT-2003-508847, Centre for Ecology and Hydrology – Lancaster, (2005).
[116] COPPLESTONE, D., The food chain transfer of radionuclides through semi natural habitats. PhD thesis, Liverpool, (1996).
[117] HIGLEY, K.A., DOMOTOR, S.L., ANTONIO, E.J., A probabilistic approach to obtaining limiting estimates of radionuclide concentration in biota, Journal of Environmental Radioactivity 66 (2003b), 75-87.
[118] BERESFORD, N.A., BROADLEY, M.R., HOWARD, B.J., BARNETT, C.L., WHITE, P.J., Estimating radionuclide transfer to wild species-data requirements and availability for terresrial ecosystems, Journal of Radiological Protection 24 (2004), A89-A103.
[119] YANKOVICH T., 2005 The Perch Lake Freshwater Scenario Description. (Revision 2). EMRAS Biota Working Group 2005. Available from: http://www-ns.iaea.org/projects/emras/emras-biota-wg.htm.
[120] VANDERPLOEG, H.A., PARZYCK, D.C., WILCOX, W.H., KERCHER, J.R., KAYE, S.V., Bioaccumulation factors for radionuclides in freshwater biota, (1975) pp.
[121] KOULIKOV N.V., CHEBOTINA, M.Y., Radioecology of freshwater biosystems, Ural Branch of the USSR Academy of Sciences, Sverdlovsk (1988) 129 pp (in Russian).
[122] VADZIS, D.R., LEINERTE, M.N., SEISUMA, Z.K., SLOKA, Y.Y., Strontium and calcium in freshwater ecosystems, Zinatne, Riga (1979) 196 pp. (in Russian).
[123] KOULIKOV, N.V., CHEBOTINA, M.Y., Radioecology of freshwater biosystems, Ural Branch of the USSR Academy of Sciences, Sverdlovsk (1988) 129 pp. (in Russian).
[124] KRYSHEV, II, ROMANOV, G.N., ISAEVA, L.N., KRYSHEV, A.I., KHOLINA, Y.B., Radioecological state of lakes in the territory of the Eastern-Ural radioactive trace, TRAPEZNIKOV, A.V., VOVK, S.M., (Eds.), Problems of radioecology and boundary disciplines, vol. 4, Zarechny: Technocentre, (2001), 107–122 (in Russian).

[125] GUDKOV, D.I., DEREVETS, V.V., ZUB, L.N., KAGLYAN, A.E., KIREEV S.I., KLENUS, V.G., Distribution of radionuclides by major components of lake ecosystems in the exclusion zone around the Chernobyl NPP, Radiation Biology, Radioecology **45** (2005), 271–280 (in Russian).
[126] KRYSHEV, A.I., ^{90}Sr in fish: A review of data and possible model approach, Science of the Total Environment 370 (2006), 182-189.
[127] OPHEL, I.M., FRASER, C.D., The fate of ^{60}Co in a natural freshwater ecosystem, Radionuclides In Ecosystems, Proceedings of the Third National Symposium on Radioecology, US AEC Tech. Inf. Cent., (NELSON, D.J., (Ed.)) Oak Ridge National Laboratory, Oak Ridge, US AEC Tech. Inf. Cent. (1973), 323-327.
[128] COUGHTREY, P.J., THORNE, M.C., Radionuclide distribution and transport in terrestrial and aquatic ecosystems, Volume 3, A.A Balkema, Rotterdam (1983).
[129] LÜTTGE, U., Mikroautoradiographische Untersuchungen uber die Funktion der hydropoten von Nymphaea, Protoplasma 59, (1964), 157-162.
[130] OPHEL, I.M., The fate of radiostrontium in a freshwater community, Radioecology, Reinhold Publishing Corporation, New York, (1976), 213-216.
[131] OPHEL, I.M., JUDD, J.M., Sr–Ca relationships in aquatic food chains, Proceedings of the Second National Symposium on Radioecology, US AEC Tech. Inf. Cent., CONF-670503, Oak Ridge National Laboratory, Oak Ridge, TN., (1967), 221–225.
[132] INTERNATIONAL ATOMIC ENERGY AGENCY, Quantification of Radionuclide Transfer in terrestrial and Freshwater Environments for Radiological Assessments, IAEA-TECDOC, Vienna, (to be published).
[133] BIRD, G.A., SCHWARTZ, W.J., MOTYCKA, M., Fate of Co-60 and Cs-134 added to the hypolimnion of a Canadian Shield Lake: Accumulation in biota, Canadian Journal of Fish Aquatic Science 55 (1998), 987-998.
[134] YANKOVICH, T.L., SCHWEITZER, D., RYAN. R., Preliminary screening of aquatic macrophyte sensitivities as biomonitors for environmental risk assessment of nuclear facilities: An ecosystem approach, Proceedings of the Second International Symposium on Ionizing Radiation, Environmental Protection Approaches for Nuclear Facilities, Ottawa, Ontario, (1999), 10 pp.
[135] CHESSER, R.K., SUGG, D.W., LOMAKIN, M.D., VAN DEN BUSSCHE, R.A., DEWOODY, J.A., JAGOE, C.H., DALLAS, C.E., WHICKER, F.W., SMITH, M.H., GASCHAK, S.P., CHIZHEVSKY, I.V., LYABIK, V.V., BUNTOVA, E.G., HOLLOMAN, K., BAKER, R.J., Concentrations and dose rate estimates of (134,137)cesium and (90)strontium in small mammals at Chornobyl, Ukraine, Environmental Toxicology and Chemistry 19 (2000), 305-312.
[136] GASCHAK, S., CHIZHEVSKY, I., ARKHIPOV, A., BERESFORD, N.A., BARNETT, C.L., The transfer of Cs-137 and Sr-90 to wild animals within the Chernobyl exclusion zone, International conference on the protection of the environment from the effects of ionizing radiation, IAEA, Stockholm, (2003), 200-202.
[137] JAGOE, C.H., MAJESKE, A.J., OLEKSYK, T.K., GLENN, T.C. AND SMITH, M.H., Radiocesium concentrations and DNA strand breakage in two species of amphibians from the chernobyl exclusion zone, Radioprotection-Colloques 37 (2002), 873-878.
[138] RYABOKON, N.I., SMOLICH, I.I., KUDRYASHOV, V.P., GONCHAROVA, R.I., Long-term development of the radionuclide exposure of murine rodent populations in Belarus after the Chernobyl accident, Radiation Environmental Biophysics 44 (2005), 169–181.

[139] BERESFORD, N.A., WRIGHT, S.M., BARNETT, C.L., WOOD, M.D., GASCHAK, S., ARKHIPOV, A., SAZYKINA, T.G., HOWARD, B.J., Predicting radionuclide transfer to wild animals: an application of a proposed environmental impact assessment framework to the Chernobyl exclusion zone, Radiation and Environmental Biophysics 44 (2005), 161-168.
[140] BERESFORD, N.A., GASCHAK, S., BARNETT, C.L., HOWARD, B.J., CHIZHEVSKY, I., STRØMMAN, G., OUGHTON, D.H., WRIGHT, S.M., MAKSIMENKO, A., COPPLESTONE, D., Estimating the exposure of small mammals at three sites within the Chernobyl exclusion zone – a test application of the ERICA Tool, Journal of Environmental Radioactivity 99 (2008), 1496-1502.
[141] MCGEE, E.J., JOHANSON, K.J., KEATINGE, M.J., SYNNOTT, H.J., COLGAN, P.A., An evaluation of ratio systems in radioecological studies, Health Physics 70 (1996), 215-221.
[142] NEDVECKAITE, T., Radiation protection in Lithuania, Vilnius, Kriventa (2004) 240 pp. (in Lithuanian).
[143] SOBOTOVICH, E.M., BONDARENKO, G.N., DOLIN, V.V., Biogenic and abiogenic migration of ^{90}Sr and ^{137}Cs of Chernobyl origin in terrestrial and aqueous ecosystems, Environmental Science and Pollution Research 1 (2003), 31-38.
[144] BERESFORD, N.A., ANDERSON P., BEAUGELIN-SEILLER, K., BROWN J., COPPLESTONE D., HOSSEINI, A., HOWARD B.J., Approaches to demonstrate protection of the environment from ionising radiation, Workshop report for the PROTECT project, EC Contract Number: 036425 (FI6R), Centre for Ecology and Hydrology – Lancaster, (2007).
[145] COPPLESTONE, D., HINGSTON, J., REAL, A., The development and purpose of the FREDERICA radiation effects database, Journal of Environmental Radioactivity 99 (2008), 1456-1463.
[146] GARNIER-LAPLACE, J., DELLA-VEDOVA, C., GILBIN, R., COPPLESTONE, D., HINGSTON, J., CIFFROY, P., First derivation of predicted-no-effect values for freshwater and terrestrial ecosystems exposed to radioactive substances, Environmental Science and Technology 40 (2006), 6498-6505.
[147] GARNIER-LAPLACE, J., COPPLESTONE, D., GILBIN, R., ALONZO, F., CIFFROY, P., GILEK, M., AGÜERO, A., BJÖRK, M., OUGHTON, D.H., JAWORSKA, A., LARSSON, C.M., HINGSTON, J.L., Issues and practices in the use of effects data from FREDERICA in the ERICA Integrated Approach, Journal of Environmental Radioactivity 99 (2008), 1474-1483.
[148] COUGHTREY, P.J., JACKSON, D., THORNE, M., Radionuclide Distribution and Transport in Terrestrial and Aquatic Ecosystems: A Compendium of Data, 6, Balkema, AA, Rotterdam (1985).

APPENDIX I. OVERVIEW OF MODELS/APPROACHES USED WITHIN THE BWG EXERCISES

Model	Short description	Documentation
Tools/approaches enabling all aspects of exposure assessment to be conducted and which are freely available to all users		
EA R&D 128	The approach and associated spreadsheet tools have been developed primarily to assess compliance with the EC Habitats Directive in England and Wales. The tools cover three ecosystem types: coastal, freshwater and terrestrial. The approach uses 'reference organisms' to represent biota and covers 16 and 18 radionuclides in aquatic/terrestrial ecosystems respectively. The tool uses an equilibrium based approach and default databases contain parameters for concentration ratios for each reference organism geometry/radionuclide (obtained using guidance where there are gaps in the literature), weighting factors, occupancy factors and dose conversion coefficients (DCCs). DCCs are estimated using energy absorbed fraction functions calculated separately for photons and electrons. Organisms are defined as three-axis ellipsoids, assuming uniform distribution of internally incorporated radionuclides. The tools and guidance have been, and continue to be used by the Environment Agency to assess the impact of authorised discharges of radioactive substances to Natura 2000 sites in England and Wales in a regulatory context (ie. if/when predicted doses exceed certain screening levels, regulatory action is required).	Copplestone D, Bielby S, Jones SR, Patton D, Daniel CP, Gize I (2001) Impact assessment of ionising radiation on wildlife. R&D Publication 128, Environment Agency and English Nature, Bristol. Copplestone D, Wood MD, Bielby S, Jones SR, Vives i Batlle J, Beresford NA (2003) Habitat regulations for stage 3 assessments: Radioactive substances authorisations. R&D Technical Report P3-101/SP1a. Environment Agency, Bristol. Vives i Batlle J, Jones SR, Gomez-Ros JM (2004) A method for calculation of dose per unit concentration values for aquatic biota. Journal of Radiological Protection 24: A13-A34. Latest version of the report is available from the publications section of the Environment Agency's website (www.environment-agency.gov.uk) but this does not include the spreadsheet tools for those please visit http://www.coger.org.uk/R&D128index.html).
ERICA	Tiered approach considering exposure of biota in freshwater, terrestrial and marine ecosystems. In Tier 1 input media activity concentrations are compared to environmental media concentration limits. Tiers 2 and 3 include default CR and DCC databases for radionuclides of 31 elements and 38 reference organisms. Further organism and radionuclides can be define by the user. Tier 3 has probabilistic ability. The tool contains outputs from/links to an on-line radiation effects database.	The ERICA Tool is freely available from: http://www.project.facilia.se/erica/download.html. The tool contains extensive on-line help and associated documentation for the ERICA Integrated Approach is available from: http://www.ceh.ac.uk/PROTECT/ERICAdeliverables.html. The approach will be further described in a forthcoming special issue of *J. Environ. Radioact.*.
FASSET	Documentation for the environmental assessment framework includes tabulated CR and DCC values for marine, freshwater and terrestrial reference organisms. NOTE – the FASSET framework has been superseded by the ERICA Tool.	All documentation available from: http://www.ceh.ac.uk/PROTECT/FASSETdeliverables.html. Elements of the framework were described within a special issue of J. Radiol. Prot. (2004; volume 24, 4A).

Model	Short description	Documentation
RESRAD-BIOTA	A computer code that implements the U.S. Department of Energy's (DOE's) graded approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota. Its database contains 46 radionuclides, four organism types (terrestrial animals, terrestrial plants, aquatic animals, and riparian animals), and eight default geometries. "New organism wizard" provides step by step instructions on creating new organisms for consideration, which can be linked to organisms of lower trophic levels as food sources, thereby enabling the establishment of food web relationships. Text reports and graphic charts are generated and can be exported. Sensitivity analyses on input parameters can also be automatically conducted.	RESRAD-BIOTA is freely available and can be downloaded from the RESRAD Web site (http://www.evs.anl.gov/resrad) or the U.S. Department of Energy Biota Dose Assessment Committee Web site (http://homer.ornl.gov/nuclearsafety/nsea/oepa/bdac/resrad.html). Related documents on the methodology and operation of the code (user's guide) are also available from the web sites.
Tools/approaches enabling all aspects of exposure assessment to be conducted (not freely available)		
AECL	AECL has typically adopted a multi-tiered approach ranging from very conservative Tier 1 to more realistic Tier 3 (based on Environment Canada, 1996 and 1997). Site-specific transfer parameters are preferred, with values from the scientific literature being taken when site-specific data are not available. To determine dose, DCC values and methods to estimate them are taken from various published sources.	The approach is described in Appendix III.
D-Max	Screening model for assessing exposure of biota in freshwater, terrestrial and marine ecosystems. Calculates maximum possible dose to any organism or tissue in the given ecosystem. No assumptions concerning species of organism, geometry, or behaviour are required for this screening approach.	Approach is suggested in: Smith, J.T. (2005) Effects of ionising radiation on biota: do we need more regulation? *Journal of Environmental Radioactivity* 82, 105-122.
DosDiMEco	In this model CRs are used for soil-plant (dry/dry basis) transfer and the calculation of the concentration in invertebrates, fish and plankton. For terrestrial mammal and bird species concentrations are calculated from the intake rate (using an allometric relation between body mass and intake rate), fractional gastrointestinal radionuclide absorption and retention inside the animal body. By using a DCC, derived by using a build-up factor corrected point Kernel technique (γ) and the Beth-Bloch equation (α and β), the internal and external dose can be calculated.	The model is being developed through interaction in the EMRAS Biota WG and it is described in Appendix IV. Consequently no published documentation on this model is yet available. For specific information contact golyslae@sckcen.be.
LIETDOS-BIO	LIETDOS-BIO for Environmental Protection is being developed to address contamination issues associated with nuclear power production in Lithuania. The code is designed to be consistent with	The LIETDOS-BIO approach is still under development. Calibration has been performed by participating in the IAEA EMRAS project. The LIETDOS-BIO dose rate evaluation model was used for the Ignalina NPP cooling pond Druksiai

Model	Short description	Documentation
	MCNPX, a commonly used general purpose Monte-Carlo radiation transport model. An in-built method for describing phantoms allows exposure to be calculated for organisms of any size or form. The uncertainty in model parameter values is determined by a statistical approach.	Lake hydrophytes exposure evaluation. Some results of these investigations are described in: Nedveckaite, T., Filistovic, V., Marciulioniene, D., Remeikis, V., Beresford, N. A. 2007 Exposure of biota in the cooling pond of Ignalina NPP: hydrophytes. *Journal of Environmental Radioactivity*, 97, 137-147.
Transfer tools/approaches		
CASTEAUR	Calculation tool for a dynamic assessment of spatio-temporal distribution of the radionuclide concentrations in the main abiotic and biotic components of the rivers, taking into account hydrography, hydraulic, sedimentary aspects, ecological functioning (trophic chain) and radioecology. Used for both routine and accidental discharges, with default parameterization for ^{110m}Ag, ^{241}Am, ^{58}Co, ^{60}Co, ^{134}Cs, ^{137}Cs, ^{54}Mn, ^{103}Ru and ^{106}Ru.	Information: casteaur@irsn.fr Boyer P., Beaugelin-Seiller K., Ternat F., Anselmet F. and Amielh M., (2005) A dynamic box model to predict the radionuclide behaviour in rivers for medium and long-term periods. *Radioprotection, Suppl. 1, vol. 40 (2005) S307-S313.* Duchesne, S., Boyer, P. and Beaugelin-Seiller K. (2003) Sensitivity and uncertainty analysis of a model computing radionuclides transfers in fluvial ecosystems (CASTEAUR): application to ^{137}Cs accumulation in chubs. *Ecological modelling, vol. 166, 257-276.* Beaugelin-Seiller K., Boyer, P., Garnier-Laplace J. and Adam C. (2002). CASTEAUR: a simple tool to assess the transfer of radionuclides in waterways. *Health Physics, 84(3), 539-542*.
ECOMOD	A freshwater transfer model which uses stable element concentrations in water for some radionuclides. Within BWG, have applied DCCs derived from literature.	Elements of the model are described in: Kryshev, A.I. (2002). Modelling the accumulation of ^{137}Cs by age –structured fish population. *Radioprotection – Colloques,* 37, 627-632. Kryshev, A.I. (2002). Model reconstruction of ^{90}Sr concentrations in fish from 16 Ural lakes contaminated by the Kyshtym accident of 1957. *Journal of Environmental Radioactivity,* 64, 67-64. Kryshev, A.I. and Ryabov, I.N. (2000). A dynamic model of ^{137}Cs accumulation by fish of different age classes. *Journal of Environmental Radioactivity,* 50, 221-233. Sazykina, T.G. (2000). ECOMOD – An ecological approach to radioecological modelling. *Journal of Environmental Radioactivity*, 50, 207-220.
FASTer (lite)	FASTer (lite) is a multi-compartmental model that can be used to simulate transfer through a simple terrestrial food-chain. The rate of change of the radionuclide inventory in compartments is described with a linear differential equation. The activity concentrations of dietary components are characterized using ERICA concentration ratios. Intakes of radionuclides are simulated using (i) allometrically derived	The model is not openly available as a completed software code but its configuration within appropriate simulation software (e.g. Model Maker, ECOLEGO) is a straight-forward process. The original model description can be found in: Brown, J.E., Strand, P., Hosseini, A. and Børretzen, P. (Eds.) (2003). Handbook for assessment of the exposure of biota to ionising radiation from radionuclides in the environment.

Model	Short description	Documentation
	ingestion rates, (ii) radionuclide-dependent assimilation efficiencies and (iii) assumptions concerning dietary composition. Biological half-lives are defined using allometric relationships. An earlier version of the model was used to derive numerous CR values in the FASSET project and a few values used as default in ERICA.	Deliverable Report for the EC Project FASSET (Contract No. FIGE-CT-2000-00102). Norwegian Radiation Protection Authority, Østerås, Norway, pp.101. http://www.ceh.ac.uk/protect/FASSETdeliverables.html
LAKECO-B	LAKECO-B is a dynamic uptake model, developed by NRG Arnhem, the Netherlands from 1992-1999. It takes into account the propagation of radionuclides throughout the entire food web. It was validated within the IAEA coordinated VAMP project (1992-1995), and tested within BIOMOVS II (1995). Its aim is to have a generally applicable ecological model for lakes ecosystems with a minimum amount of input parameters. The predictive power is high due to the use of subroutines which uses environmental parameters as input, so calibration is therefore not needed. For the uptake of radionuclides it uses the target-tissue approach instead of nuclide bases approach, limiting the amount of input parameters. LAKECO-B is part of the DSS system RODOS, and MOIRA. Variants are also applied for the marine environment (BURN98, POSEIDON - R). A release exists to calculate the tritium uptake in biota as well including OBT.	Popov and Heling, 1996; Heling, 1996, Heling, 1997, Zheleznyak et al; 1996, Kryshev et al, 1999; IAEA, 2000. LAKECO-B is available as stand-alone package, and as part of the hydrological module of RODOS, RODOS-HDM (www.rodos.fzk.de/).
Dosimetry tools/approaches		
EDEN	Calculation tool based on an intermediate solution between full Monte Carlo calculation and analytical empirical equations, to evaluate the energy dose rate (expressed as a Dose Conversion Coefficient, DCC) delivered to non-human species exposed to any radionuclide present in the environment or internalised, for numerous user-defined configurations (any organism, any radionuclide (alpha, beta and gamma radiation) and from internal or external exposure).	Free access on request at eden@irsn.fr – user license issued for traceability Beaugelin-Seiller K., Jasserand F., Garnier-Laplace J and Gariel J.C. (2006). Modelling the radiological dose in non-human species: principles, computerization and application. *Health Physics, 90 (5): 485-493.* Beaugelin-Seiller K. (2006). EDEN version 2 – User's Manual. IRSN/DEI, report SECRE/06-29, 38p. Beaugelin-Seiller K (2006). EDEN version 2 – Theoretical note. DCC calculation formalism. IRSN/DEI, rapport SECRE/06-28, 55p.
EPIC DOSES-3D	Research tool that allows doses from external (β particles, photons) and internal exposure (α, β particles, photons) in biological objects of any (user-defined) size and form to be calculated. Doses can be calculated for any radionuclide, although in the present version of the program an initial data set for 42 radionuclides is used. The software has been used to derive dose	A trial version of the tool is freely available on request from the developer – The Institute of Radiation Hygiene, Russia. Golikov, V. and Brown, J.E. (Eds.) . (2003). Internal and External Dose Models. Deliverable report 4 for EPIC. Deliverable Report to EC Inco-Copernicus project ICA2-CT-2000-10032. Norwegian Radiation Protection Authority,

Model	Short description	Documentation
	conversion coefficients in the EPIC project (Golikov and Brown, 2003) and is under further development.	Østerås, Norway, pp. 52. Other EPIC project reports are available from: http://www.ceh.ac.uk/PROTECT/EPICdeliverables.html
SÚJB approach	The model is used to carry out environmental impact assessments of nuclear facilities. The approach for estimating absorbed DCCs uses derived dose rate formulas as published elsewhere. Selected categories of organisms are represented by ellipsoid geometries of stated dimensions.	IAEA Technical Report Series No. 190 and No. 332. Kimmel L. P., Maschkovich V. P.: Radiation Protection Handbook, Moscow 1972 (in Russian) The approach is further described in BWG documents.

APPENDIX II. REVIEW OF THE SELECTION CRITERIA USED BY DIFFERENT BIOTA DOSE ASSESSMENT MODELS IN THE SELECTION OF REFERENCE ORGANISMS

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The terminology and definitions used by the different approaches (in terms of 'reference organisms', 'selected species' etc) are given below along with an overview of the selection criteria.

II.1. Terminology and definitions used in the different approaches

Different terms are used in the different approaches: "reference organism", "reference animal and plant", "representative species", "feature species" and receptor. The definitions of these in the various approaches considered within the BWG exercises are given below.

II.1.1. FASSET, EPIC and EA R&D 128

Reference Organisms: *"A series of imaginary entities that provides a basis for the estimation of the radiation dose rate to a range of organisms that are typical, or representative, of a contaminated environment. These estimates, in turn, would provide a basis for assessing the likelihood and degree of radiation effects. It is important to recognise that they are not a direct representation of any identifiable animal or plant species".*

II.1.2. R&D 128

Feature species (or habitat): these are the species of interest (usually in the context of conservation) that require some form of specific assessment to demonstrate that there are no significant impacts.

II.1.3. EPIC

Reference organism is *"Analogous to reference man"*

II.1.4. ICRP

Reference Animals and Plants (RAP): "A Reference Animal or Plant is a hypothetical entity, with the assumed basic biological characteristics of a particular type of animal or plant, as described to the generality of the taxonomic level of Family, with defined anatomical, physiological, and life-history properties, that can be used for the purposes of relating exposure to radiation dose, and relating dose to different categories of effect, for that type of living organism."

II.1.5. RESRAD-BIOTA (and AECL)

Reference organism: "is intended to represent typical characteristics within a particular population group." There are eight predefined geometries for reference organisms. These reference organisms are used to evaluate radiological doses to ecological receptors. A receptor is the species of interest that is exposed at the site under evaluation.

II.1.6. AECL

Valued ecosystem component (VEC) is defined as a species or habitat of significance (equivalent to feature species).

II.2. Overview of selection criteria for approaches used in BWG exercises

The selection criteria used for each of the different approaches can be summarised into four main criteria categories:

II.2.1. Ecological status/niche

Appropriate reference organisms for assessment are considered to be dominant representatives of the basic trophic levels in their ecosystem. These species are instrumental in the major energy/material flows in an ecosystem and as a result their protection is integral to the well-being of the whole ecosystem. As a general rule, one reference organism per trophic level may be selected. Whilst the organism should have ecological relevance, it has been noted that this may be difficult to assess in reality.

II.2.2. Radioecological sensitivity

Consideration for consideration as a reference organisms on the basis of the likelihood to be amongst the most exposed organisms (which may vary between radionuclides). Factors considered include:

- Exposure pathways;
- Distribution of species within ecosystem;
- Likelihood of exposure;
- Duration of exposure;
- Importance of species in energy or nutrient flow (similar to ecological status);
- Uptake and accumulation of radionuclides within species.

II.2.3. Radiobiological sensitivity

Biological species within an ecosystem vary considerably in respect to their sensitivity to ionising radiation. For example it is well known that many lower organisms are resistant to radiation. For example, bacteria and planktonic algae can be several orders of magnitude less sensitive to radiation exposure compared with fish or mammals. As a result the most radioresistant organisms are often not selected under this criteria but they are often considered under the previous two (ecological status or radioecological sensitivity). If the likely effects of radiation are to be assessed then some radiosensitive species should be included in the assessment. Information to aid selection is usually obtained from radiation effects databases/compilations.

II.2.4. Amenability for sampling and monitoring

In terms of this criteria category the following is considered:

- Typical, numerous and widespread species in the investigated area;
- Species, which can be collected easily (e.g. microscopic-size organisms are not suitable);

— Species, which can be easily identified;
— Species of commercial importance, which are monitored because of importance to man.

II.2.5. Protected status

Recognising that protected species will often be the object of conservation driven assessments the ERICA approach selected a reference organism list to encompass all European protected species.

II.3. ICRP RAPs

The ICRP concept of Representative Animals and Plants (RAPs) is envisaged to allow '*points of reference*' that are analogous to reference man to be determined. The concept is similar to that for human radiological protection, thereby providing consistency within the field of radiation protection. The points of reference are not necessarily benchmark values etc. but they are numbers to which an assessor can compare their results.

The key criteria put forward by the ICRP in the selection of RAPs are:

— Legislation relating to wildlife protection;
— Use in toxicity testing;
— Human resource;
— Data on radionuclide accumulation;
— Data on radiation effects;
— Amenable to further study;
— Public resonance.

Each of these points is expanded within the ICRP documents and attempts to draw together similarities etc with other systems of environmental or radiation protection.

APPENDIX III. AECL APPROACH

T. Yankovich[11], AECL, Chalk River Laboratories, Canada

For the purposes of some of the BWG exercises the opportunity was taken to apply the FASSET transfer parameters (Section 2.5), the RESRAD-BIOTA tool (Section 2.3), as well as the methodology to estimate DCCs that has been developed by Blaylock et al. [III.1] as tools for transfer and dose assessment to gain more familiarity with available international approaches; however, for the purposes of ecological risk assessment in Canada, AECL has typically adopted a multi-tiered approach ranging from very conservative Tier 1 to more realistic Tier 3 (based on Environment Canada [III.2; III.3], as described in the sections that follow.

Total dose to biota (D_{TOT}), also called the estimated exposure value (EEV), consists of internal plus external doses, as described by the equation:

$$D_{TOT} \text{ or } EEV = DE_R + DI_R \quad \text{(III.1)}$$

where:

D_{TOT} or EEV represents the total radionuclide dose received by biota ($Gy \cdot a^{-1}$);
DE_R is the external radionuclide (R) dose received by biota ($Gy \cdot a^{-1}$);
DI_R is the internal radionuclide dose received by biota ($Gy \cdot a^{-1}$).

Doses are weighted to account for factors, such as habitat-use, habitat quality, home range, period of residence and in some cases, trophic position, which are known to influence the probability of biota exposure to radionuclides, as well as the magnitude and duration of exposure.

III.1. Estimation of Dose to Aquatic Receptor Species

III.1.1. Estimation of External Dose to Aquatic Biota

Aquatic biota are assumed to receive external dose through immersion in water, sediments (including the solid particulates and the surrounding porewater) and aquatic vegetation [III.4–III.7] as described by the general equation:

$$DE_R = [CO_{R,W} \cdot UW \cdot DCFEW_R] + [CO_{R,S} \cdot US \cdot DCFES_R] + [CO_{R,V} \cdot UV \cdot DCFEV_R] \quad \text{(III.2)}$$

where:

$CO_{R,(W,S,V)}$ is the concentration of radionuclide (R) in water (W), sediments (S) or vegetation (V) ($Bq\ l^{-1}$ or $Bq\ kg^{-1}$);
UW, US and UV are the habitat-use factors of biota for water, sediments and vegetation, respectively (dimensionless);
$DCFEW_R$, $DCFES_R$ and $DCFEV_R$ are the external dose conversion factors for water, sediments, and vegetation, respectively.

It is assumed that the relative external dose received by biota through immersion in each type of environmental medium or compartment is dictated by the proportion of time an organism

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spends in contact with a given environmental medium (as reflected by species-specific and medium-specific habitat-use factors), the concentration of radionuclides in each medium, and the relationship between exposure concentrations and dose (i.e. the DCF or DCC) [III.4; III.5]. This parameter reflects the 'life-style', behaviour and the types of habitats occupied by the organism under consideration.

In estimating external exposure from tritium in vegetation, it is often necessary to estimate tritium concentrations due to the lack of measured data. This is accomplished using a ^{3}H specific activity model to account for 'dilution' of ^{3}H by stable H isotopes [III.8; III.9]. Using this approach, ^{3}H levels in a given compartment are considered relative to the total percent weight of hydrogen in the compartment and total hydrogen contents are chosen based on the chemical composition of the compartment, as described by the following equation:

$$SA_{HTO} = \frac{CW_T}{CW_H} \cdot CV_H \qquad \text{(III.3)}$$

where:

SA_{HTO} is the specific activity of tritium in the vegetation (Bq kg^{-1} (fw));
CW_T is the concentration of tritium in the surface water (Bq l^{-1});
CW_H is the concentration of hydrogen atoms in the surface water (g hydrogen l^{-1});
CV_H is the mean hydrogen concentration in vegetation (g hydrogen kg^{-1} fresh biomass).

It is assumed that the weight percent of hydrogen in lake water is 11.1%

(or 111 g H l^{-1}), based on the hydrogen content of water [9; 10]. CV_H is also set to a constant value of 120 g H kg^{-1} fresh biomass for plant tissues. Therefore, for aquatic plants, Equation 3 is re-written as:

$$SA_{HTO} = \frac{CW_T}{111} \cdot 120 \qquad \text{(III.4)}$$

By comparison, a mean hydrogen concentration of 130 g H/kg fresh biomass can be assumed for animal tissues. This approach is conservative, typically over-estimating tritium concentrations.

Once concentrations of radionuclides in environmental media have been quantified, they are then related to the external radionuclide dose received by biota by applying an environmental medium-specific DCF (or DCC) for each radionuclide, which is weighted for the proportion of time an organism spends in contact with a given medium. External DCFs are taken from Amiro [III.11], which have been developed using information provided in Barnard and D'Arcy [III.12], Holford [III.13]) and Holford [III.14]. The $DCFE_R$ values include contributions from all the radioactive progeny (e.g. ^{137m}Ba for ^{137}Cs). Additional dose conversion factors not provided in Amiro [III.11] are tabulated for aquatic biota using the same approach.

III.1.2. Estimation of Internal Dose to Aquatic Biota

The internal whole body dose (DI) received by aquatic biota through exposure to radionuclides (R) that have been incorporated in body tissues are represented by the equation [III.5]:

$$DI_R = CO_R \cdot DF_R \cdot DCFI_R \cdot AF \quad \text{(III.5)}$$

where:

DI_R is the internal dose received from each radionuclide, R ($Gy \cdot a^{-1}$);
CO_R is concentration of a given radionuclide in biota tissues (Bq kg^{-1} (fw));
DF_R is the radiation distribution factor in the body (dimensionless);
$DCFI_R$ is the internal dose conversion factor for the radionuclides of interest ($Gy \cdot a^{-1}$ per Bq kg^{-1} (fw));
AF is the absorption factor (dimensionless).

The internal dose received by biota is dictated by the concentration of radionuclides in environmental media and the dietary items which are ingested by an organism, the net radionuclide accumulation into biota tissues, the internal distribution of radionuclides once they enter the body and the energy deposition patterns in biota tissues as radionuclides in tissues undergo radiological decay.

The total internal dose received by an organism represents the sum of doses received from all radioisotopes present in the body. To estimate internal dose to biota the radionuclide concentration in an organism (CO_R) are either directly measured, or indirectly estimated based on transfer coefficients or bioaccumulation factors (also called CRs). However, such factors can vary considerably between different sites and species. Therefore, where possible, measured radionuclide concentrations are used to estimate internal doses to aquatic biota from radionuclides. When site-specific data are not available for a given aquatic ecosystem, water-to-biota CRs from the Canadian literature and/or international reviews (e.g. [III.15] are applied to estimate radionuclide concentrations in aquatic plants, invertebrates, fishes and amphibians. Allometric approaches are applied to estimate transfer to aquatic mammals and waterfowl (as described below).

Again, tritium concentrations in aquatic vegetation are approximated using a specific activity approach, as described by Equation 4. As for plants, CV_H is also constant for animal tissues, with a value of 130 g H kg^{-1} (fw) in comparison to the value of 120 g H kg^{-1} (fw) for plant tissues [III.5]. The animal value is, therefore, plugged into Equation 4 to estimate the HTO specific activity in aquatic animal tissues, where necessary. Similarly, a specific activity approach is also used for C-14 (e.g. [III.16; III.17].

The CO_R is then converted to an internal dose by multiplying CO_R by an internal dose conversion factor, $DCFI_R$, which is specific to the radioisotope under consideration [III.11]. As for the external dose conversion factors, Amiro [III.11] compiled $DCFI_R$ values using techniques and assumptions taken from Barnard and D'Arcy [III.12], Holford [III.13] and Holford [III.14]. Estimates rely on radiation emission data published by the ICRP [III.18] and include contributions from all the radioactive progeny (e.g. ^{137m}Ba for ^{137}Cs). However, Amiro's $DCFI_R$ values can be conservative depending upon their application, for example, assuming that all radiation emitted internally is self-absorbed by the organism regardless of energy of the radionuclide or body size. Blaylock et al. [III.1] developed a methodology to account for the fraction of energy absorbed internally by aquatic organisms for a number of radionuclides and body sizes (by applying an absorption factor, AF), which produces more realistic internal dose conversion factors for whole organisms [III.11; III.19]. This second approach, along with others that account for organism size and shape (e.g. FASSET as described in Section 2.5 and RESRAD-BIOTA as discussed in Section 2.3) have been used for more realistic biota dose assessments, whereas the more conservative approach that was

developed by Amiro [III.11] (which assumes that the fraction of energy absorbed is equal to one; i.e. AF = 1) is used for screening purposes.

A Distribution Factor ($DF_{R,T}$) is applied to account for the uneven distribution of a radionuclide in the body of the organisms, which can lead to higher doses than for those received from uniformly distributed radionuclides. The DF_R represents the concentration of a radionuclide (R) in the tissue where it tends to accumulate relative to the concentration of that radionuclide in the whole body [III.20], and can be described by the following equation:

$$DF_R = \frac{CO_{PeakTissue}}{CO_{WholeBody}} \quad \text{(III.6)}$$

In Equation 6, $CO_{Peak\ Tissue}$ represents the radionuclide concentration in the tissue that shows the highest concentration in the body, and $CO_{Whole\ Body}$ represents the radionuclide concentration in the whole body.

In cases where biota are receiving exposure to α particles, a quality factor is applied to account for the greater biological effectiveness of α particles, relative to β and γ emissions. Values of 10, 1 and 1 are typically applied for α, β and γ radiation, respectively [III.21].

III.2. Estimation of Dose to Terrestrial Receptor Species

In terrestrial systems, biota are expected to come into contact with air, soil and terrestrial vegetation to varying extents depending upon the species and its lifestyle. It is assumed that the soil includes solid soil particulates, in addition to porewater. Terrestrial biota can, therefore, receive external dose through immersion in soil, air and vegetation, although the dose received from immersion in air is likely relatively small. External doses to terrestrial non-human biota are further sub-divided into aboveground and belowground doses, depending upon habitat-use patterns of a given species or type of organism [III.21]. For riparian species that utilize both aquatic and terrestrial environments, occupancy or habitat-use factors are weighted representatively between these two habitat types.

III.2.1. Estimation of Aboveground External Dose to Terrestrial Biota

Aboveground external doses are defined as those received by terrestrial organisms through direct radiation or shine from contaminated soil or vegetation, as described by the equation reported by Sample et al. [III.21] as follows:

$$D_{\substack{\text{above}\\\text{ground}}} = F_{\substack{\text{above}\\\text{ground}}} \cdot F_{\substack{\text{roughness}\\\text{factor}}} \sum_R C_{\text{soil,R}} \cdot DC_{\text{ground,R}} \cdot CF_b \cdot ECF \quad \text{(III.7)}$$

where:

$D_{\substack{above\\ground}}$ represents the external dose rate from aboveground exposure to contaminated soil (Gy/a);

$F_{\substack{above\\ground}}$ is the proportion of time spent above ground (dimensionless);

$F_{\substack{roughness\\factor}}$ is the dose rate reduction factor accounting for surface roughness of soil (dimensionless);

$C_{soil,i}$ represents activity of radionuclide, *R*, in surface soil (Bq kg^{-1} (dw));

$DC_{ground,R}$ is the dose coefficient factor for radionuclide, *R*, in soil contaminated to a given depth (Sv/s per Bq m^3) (from Eckerman and Rymann [III.22];

CF_b is the conversion factor to change Sv/s per Bq m^3 to Gy a^{-1} per Bq kg^{-1} (is equal to 5.05×10^{10}); and

ECF is the elevation correction factor to adjust dose coefficients to value representative of effective height of animal aboveground.

Above ground dose values are based on dose coefficients reported by Eckerman and Ryman [III.22] for terrestrial non-human biota exposed to soils contaminated to 15 cm depths, a default value typically applied in terrestrial biota dose estimates [III.21]. A depth of 15 cm is selected from possible depths of 1 cm, 5 cm, 15 cm or an infinite depth, as recommended by Sample et al. [III.21], since a 15 cm depth is thought to represent the depth range at which biological activity primarily occurs in many cases. In doing so, it is assumed that external α-doses are negligible due to the poor penetration power of α–particles. As a result, the formulae of Eckerman and Ryman [III.22] deal with high-energy β- and γ-rays. A dose rate reduction factor $F_{\substack{above\\ground}}$ accounts for the fraction of time spent aboveground by biota, as opposed to belowground (which is covered separately by a different model, as described below) [III.21].

Surface roughness is accounted for by $F_{\substack{roughness\\factor}}$, which can vary between 0.5 for a deeply plowed field, to unity for a perfect plane, with 0.7 representing a reasonable default value for most natural surfaces. For small mammals that are closer to the soil surface than 1 m, an elevation correction factor (*ECF*) of 2 is applied, which represents a measure of proximity to the source. For larger mammals and humans, which are taller and therefore, further from the source (i.e. being exposed to decreased groundshine), an ECF value of unity is used.

III.2.2. Estimation of Belowground External Dose to Terrestrial Biota

Belowground external doses are received when organisms spend time underground, for example in burrows or dens, or during activities, such as hibernation, and are estimated using the equation provided by Sample et al. [III.21], as follows:

$$D_{\substack{below\\ground}} = 1.05 \cdot F_{\substack{below\\ground}} \sum_R C_{soil,R} \cdot E_R \cdot CF_a \qquad \text{(III.8)}$$

where:

$D_{\substack{below\\ground}}$ represents the external dose rate to non-human biota from contaminated soil (Gy/a);

$F_{\substack{below\\ground}}$ is the proportion of time spent below ground (dimensionless);

E_R is the energy for □ emissions by nuclide R (MeV/nuclear transition);

$C_{soil,R}$ is the activity of radionuclide, *R*, in surface soil (Bq kg^{-1} (dw));

1.05 is a conversion factor to account for differences between DCFs for immersion of small volumes of tissue in soil versus DCFs for immersion of small volumes of tissue in water;

CF_a is the conversion factor to convert from MeV/nuclear transition to Gy·kg/Bq·a (5.05×10^{-6}).

The belowground exposure values are estimated based on the assumptions provided in Sample et al. [III.21], whereby it is assumed that the receptor organism is immersed in a continuous soil medium and that the organism itself represents a small volume of tissue. A 1.05 conversation factor is applied to account for the difference between the immersion of a small volume of tissue in soil relative to water.

External DCFs for soil are taken from USDOE [III.6], whereas external DCFs reported by Amiro [III.11] are applied for vegetation. The energies for γ emissions are taken from Blaylock et al. [III.1].

III.2.3. Estimation of Internal Dose to Terrestrial Biota

In terrestrial ecosystems, soil-to-plant and soil-to-animal transfer factors are applied to estimate radionuclide partitioning into plants and terrestrial invertebrates (from site-specific measurements and from the literature; e.g. [III.15; III.21; III.23– III.25], in combination with allometric approaches, which are applied to assess transfer to vertebrates.

Radionuclides can be transferred internally to terrestrial vertebrates through inhalation, ingestion of food, ingestion of soil and water containing radionuclides and dermal absorption pathways. The total amount of radionuclide that becomes incorporated into vertebrate tissues is dependent upon the concentration of the radionuclide in the atmosphere, food, soil and water, the rate of exposure (as reflected by the breathing rate, food ingestion rate, soil ingestion rate and water ingestion rate, respectively), and the efficiency of incorporation into biota tissues relative to losses. In most cases, it is expected that ingestion pathways contribute relatively more to the doses received by terrestrial biota [III.26] than do the inhalation or dermal contact pathways. In fact, in most cases, exposure via inhalation and dermal absorption are both considered negligible (e.g. for birds and mammals [III.21].

As for the dermal pathway, radionuclide uptake via inhalation is also generally assumed to be negligible [III.21] because most contaminated sites tend to be either capped or covered with vegetation, which minimizes aerial suspension of contaminated dust particulates. In addition, volatile contaminants are assumed to rapidly volatilize from soil and surface water to air, and are then diluted and dispersed. That said, for screening purposes, radionuclide uptake through inhalation is conservatively estimated to represent the product of the inhalation rate and the concentration of a given radionuclide in air, assuming that 100% of the inhaled radionuclide is incorporated into the organism tissues.

In addition, uptake of radionuclides through the ingestion of food and soil by animals and the relative contributions of each pathway to internal dose are estimated using the approach developed by the USEPA [III.27], where dose estimates are typically normalized relative to body weight as follows:

$$DI_R = \sum_{m=1}^{n} I_m \cdot C_{m,R} \cdot BTF_R \cdot f_t \cdot f_d \cdot AF \cdot DCFI_R \qquad \text{(III.9)}$$

where:

DI_R is the total internal dose from radionuclide *R* (Gy a^{-1}) through ingestion of contaminated media *n* is the total number of ingested media (e.g., food, water soil, sediment, etc.);
I_m is the species-specific intake rate for medium *m* (kg day^{-1} or l day^{-1});

$C_{m,R}$ is the concentration of radionuclide *R* in medium *m* (Bq kg^{-1} (fw)) or Bq l^{-1});
BTF_R is the biotransfer factor of radionuclide *R* (d kg-$^{1)}$;
f_t is the fraction of time an animal spends at the site of interest;
f_d is the fraction of the diet of an animal that consists of food from the site of interest;
AF is the absorption factor (dimensionless);
$DCFI_R$ is the internal dose conversion factor for radionuclide *R* (Gy a^{-1} per Bq kg^{-1} (fw)).

As shown in Equation 9 above, food, soil, water and air concentrations ($C_{m,R}$) are multiplied by their respective intake rates (I_m) and by a contaminant-specific biotransfer factor (food-to-tissue) (BTF_R), then summed to obtain the concentration of individual radionuclides in biota tissues [28]. Again, it may be necessary to apply a quality factor (QF) to the dose estimate to account for differences in the biological effectiveness of α particles, where QF values of 10, 1 and 1 are typically used for α, β and γ radiation, respectively.

Animal diet is accounted for assuming a representative diet, based on data from several literature sources, as follows:

$$E_j = \sum_{i=1}^{m} \sum_{k=1}^{n} p_{ik} I_i \cdot C_{ijk} \qquad \text{(III.10)}$$

where:

E_j is the total ingestion exposure to contaminant *j* (Bq kg^{-1} medium ingested);
n is the number of types of medium *i* consumed (dimensionless);
p_{ik} is the proportion of type *k* of medium *i* consumed (dimensionless);
C_{ijk} is the concentration of contaminant *j* in type *k* of medium *i* (Bq kg^{-1} medium or Bq l^{-1}medium).

In addition, on average, a soil ingestion rate of approximately 10% of the food intake rate [III.7].

Inhalation and ingestion rates are assumed to change as a function of body size, as described by allometric relationships [III.29] using the approach described by the USEPA [III.27], as represented by the following general equation for power functions:

$$y = aWt^b \qquad \text{(III.11)}$$

where:

y is the predicted biological function (e.g. inhalation rate);
Wt the fresh weight of the animal (in kg or g);
a and b are fitted empirical coefficients that were quantified based on data representing many observations of one broad characteristic group (e.g. inhalation rates measured for many mammals.

In cases where a large range of body masses are reported in the literature, it is conservatively assumed that typical body masses fall at the upper limit of the literature range. In addition, estimated food ingestion rates are multiplied by 3 for all receptor species, since actual values may be 2- to 3-fold higher than estimated values in nature [III.27]. Exposure through dermal absorption is not represented through use of allometric equations, but is accounted for, to

some extent, through application of external dose conversion factors that include dermal absorption of radionuclides in amphibians and invertebrates (which have living integuments) [III.5; III.21].

Radionuclide intake rates for each radionuclide and medium are then used to estimate radionuclide concentration in biota tissues by applying a biotransfer factor (BTF). The BTF represents the ratio of contaminant concentration in animal tissue (Bq contaminant/kg tissue, (fw)) to daily intake (Bq contaminant/d) and is a measure of how much of what an animal ingests is actually transferred to tissue. The BTF (d/kg) is multiplied by a species-specific food ingestion rate (kg/d) and by contaminant concentration in food (Bq kg^{-1}) to obtain an estimate of the concentration in biota tissues. BTFs are both chemical- and species-specific and can be influenced by site-specific conditions, so site-specific values are preferred when available. However, food-to-animal tissue BTFs are often the only BTF values available (e.g. [III.15; III.30–III.33]. When values are unavailable for soil-to-tissue and water-to-tissue, it is assumed that radionuclide transfer from soil- and water-to-tissue is similar to that for food-to-tissue and the same BTF is used for each exposure route.

Once radionuclide concentrations have been estimated for biota tissues, concentrations are then weighted for the proportion of time an organism spends feeding and living in an area where they receive exposure to radionuclides (f_t), as well as the proportion of contaminated media they ingest or intake (f_d). For example, the period of residence of a given species will affect their total potential exposure to radionuclides in the environment. Period of residence is defined as the proportion of a year that a given species spends in a local area. Non-migratory species are assumed to spend 100% of their time in the local area, and therefore, have a period of residence that is equal to 1 ($f_t = 1$). However, migratory species that leave the local area for part of the year, would have a period of residence that is less than 1. For screening purposes, it is conservatively assumed that the receptor species under consideration (which are non-migratory) spend 100% of their time in the area of interest (i.e. $f_t = 1$) and that 100% of the media they ingest contains radionuclides (i.e. $f_d = 1$). In situations where more realistic assessments would be needed (at the higher assessment tiers), more realistic assumptions are made and factors, such as home range relative to the spatial extent of contamination would be taken into account.

REFERENCES

[III.1] BLAYLOCK, B.G., FRANK, M.L., O'NEAL, B.R., Methodology for estimating radiation dose rates to freshwater biota exposed to radionuclide in the environment. Report ES/ER/TM-78, Oak Ridge National Laboratory, Tennessee (1993).

[III.2] ENVIRONMENT CANADA, Ecological risk assessment of priority substances under the Canadian Environmental Protection Act. Resource document. Draft 1.0 (1996).

[III.3] ENVIRONMENT CANADA, Environmental assessment of priority substances under the Canadian Environmental Protection Act. Guidance manual Version 1.0. EPS/2/CC/3E (1997).

[III.4] AMIRO, B.D., ZACH, R., A method to assess environmental acceptability of releases of radionuclides from nuclear facilities, Environ. Internat., **19** (1993) 341–358.

[III.5] ZACH, R., ROWAT, J.H., DOLINAR, G.M. SHEPPARD, S.C., KILLEY, R.W.D., Ecological risk assessment for the proposed IRUS low level waste disposal facility at AECL's Chalk River Laboratories, TR-791 (1998).

[III.6] UNITED STATES DEPARTMENT OF ENERGY, DOE Technical Standard, A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota, US DOE, Washington D.C. (2000).

[III.7] UNITED STATES DEPARTMENT OF ENERGY, A graded approach for evaluating radiation doses to aquatic and terrestrial biota, Technical Standard DOE-STD-1153-2002, US DOE, Washington D.C. (2002).

[III.8] MASON, A.S. ÖSTLUND, H.G., Atmospheric HT and HTO: V. Distribution and large-scale circulation. In: Behaviour of Tritium in the Environment. IAEA-SM-232/62 (1979) pp. 3–16.

[III.9] ZACH, R., SHEPPARD, S.C., Food-chain and dose model, CALDOS, for assessing Canada's nuclear fuel waste management concept. Health Physics, **60** (1991) 643–656.

[III.10] ADAMS, L.W., PETERLE, T.J. WHITE, G.C., Tritium behaviour in aquatic plants and animals in a freshwater marsh ecosystem. In: Behaviour of Tritium in the Environment. IAEA-SM-232/74 (1979) pp. 231–245.

[III.11] AMIRO, B.D., Radiological dose conversion factors for generic non-human biota used for screening potential ecological impacts, J. Environ. Radioact., **35** (1997) 37–51.

[III.12] BARNARD, J.W., D'ARCY, D., EDEFIS, a code for calculating effective dose equivalent for immersion in contaminated media, TR-244, (1986) 152 pp.

[III.13] HOLFORD, R.M., Dose conversion factors for air, water, soil and building materials, AECL-09825 (1988) 449 pp.

[III.14] HOLFORD, R.M., Supplement to dose conversion factors for air, water, soil and building materials. Atomic Energy of Canada Limited Report, AECL-9825-1 (1989).

[III.15] INTERNATIONAL ATOMIC ENERGY AGENCY, Handbook of parameter values for the prediction of radionuclide transfer in temperate environments, Technical Reports Series No. 364, IAEA, Vienna (1994).

[III.16] YANKOVICH, T.L., SHARP, K.J., BENZ, M.L., CARR, J., KILLEY, R.W.D., Validation of the Carbon-14 specific activity model in a wetland environment for application in biota dose assessment. Proceedings of the American Nuclear Society (ANS) Topical Meeting on Decommissioning, Decontamination and Reutilization (DD&R), Chattanooga, Tennessee, 16–19 September 2007 (2007a).

[III.17] YANKOVICH, T.L., KUPFERSCHMIDT, D.A., SHARP, K.J., BENZ, M.L., KIM, S.B., SHULTZ, C., AUDETTE-STUART, M., CARR, J., Application of plants as biomarkers to assess the health of wetland ecosystems receiving ^{14}C through groundwater influx. Proceedings of the American Nuclear Society (ANS) Topical Meeting on Decommissioning, Decontamination and Reutilization (DD&R), Chattanooga, Tennessee, 16–19 September 2007 (2007).

[III.18] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Radionuclide transformations – energy and Intensity of transmissions, Annals of the ICRP, 11, Pergamon Press, Oxford (1983).

[III.19] BECHTEL JACOBS COMPANY LLC (for the United States Dept. of Energy), Radiological benchmarks for screening contaminants of potential concern for effects on aquatic biota at Oak Ridge National Laboratory, Oak Ridge, Tennessee. BJC/OR-80 (1998).

[III.20] YANKOVICH, T.L., BEATON, D., Concentration ratios of stable elements measured in organs of terrestrial, freshwater and marine non-human biota for input into internal dose assessment for PSL-2: A literature review, COG-99-106-I, (2000) 120 pp.

[III.21] SAMPLE, B., APLIN, M.S., EFROYMSON, R.A., SUTER II, G.W., WELSH, C.J.E., Methods and tools for estimation of the exposure of terrestrial wildlife to contaminants, Oak Ridge National Laboratory, ORNL/TM-13391 (1997) 148 pp.

[III.22] ECKERMAN, K.F., RYMAN, J.C., External Exposure To Radionuclides In Air, Water, And Soil, Federal Guidance Report No. 12, EPA-402-R-93-081 (1993).

[III.23] SHEPPARD, S.C., EVENDEN, W.G., Critical compilation of plant/soil concentration ratios for uranium, thorium, and lead, J. Environ. Radioact. **8** (1988) 255–285.

[III.24] SHEPPARD, S.C., EVENDEN, W.G., Bioavailability indices for uranium: Effect of concentration in eleven soils, Arch. Environ. Chem. **51** (1992) 844–851.

[III.25] SHEPPARD, S.C., EVENDEN, W.G., POLLOCK, R.J., Uptake of radionuclides by field and garden crops, Can. J. Soil Sci. **69** (1989) 751–767.

[III.26] NUCLEAR REGULATORY COMMISSION, Regulatory Guide 1.109: Calculation of annual doses to man from routine releases of reactor effluents for the purpose of evaluating compliance with 10 CFR Part 50, Appendix I. NRC, Washington, D.C. (1977).

[III.27] UNITED STATES ENVIRONMENTAL PROTECTION AGENCY, Wildlife Exposure Factors Handbook, Vol. I, EPA/600/R-93/187a, US EPA Washington, D.C. (1993).

[III.28] UNITED STATES ENVIRONMENTAL PROTECTION AGENCY, Exposure Assessment Methods Handbook, EPA/600, Exposure Assessment Group, Office of Health and Environmental Assessment, US EPA Washington, D.C. (1989).

[III.29] NAGY, K.A., Field metabolic rate and food requirement scaling in mammals and birds, Ecological Monographs, **57(2)** (1987) 111–128.

[III.30] NG, Y.C., COLSHER, C.S., THOMPSON, S.E., Transfer coefficients of the dose-to-man via the forage-cow-milk pathway from radionuclides released to the biosphere, Rep. No. UCRL-51939, Lawrence Livermore National Laboratory, Livermore, CA (1977).

[III.31] NG, Y.C., COLSHER, C.S., THOMPSON, S.E., Transfer factors for assessing the dose from radionuclides in agricultural products, p. 295 in Biological Implications of Radionuclides Released from Nuclear Industries, Rep. No. IAEA-STI/PUB/522, IAEA, Vienna (1979).

[III.32] NG, Y.C., COLSHER, C.S., THOMPSON, S.E., Soil-to-plant concentration factors for radiological assessments, NUREG/CR-2975, Environmental Sciences Division, Lawrence Livermore National Laboratory, Livermore, CA (1982).

[III.33] NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS, Screening techniques for determining compliance with environmental standards: releases of radionuclides to the atmosphere, NCRP Commentary No. 3, Bethesda, MD (1989).

APPENDIX IV. DOSDIMECO

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IV.1. Dosimetric model: Approach and description

The aim of the model is to calculate the energy delivery to reference organisms according to different exposure conditions (scenarios). We will speak of Dose Conversion Coefficient (DCC) as a measure of the absorbed dose an organism receives during its residence in a certain contaminated environment (e.g. on soil, in soil, in air, on water, on benthic sediment) or the dose due to internal contamination. The program is written in MathCad 2001i professional and can be subdivided in 3 parts, each linked to the calculation of the energy absorption of the 3 irradiation types considered (i.e. gamma, beta and alpha irradiation) coming from a volumetric source. The DCC depends on the type and energy level of the particle, the scenario and the reference organism (RO) selected.

To perform the calculations some assumption had to be made. For practical purposes, all RO are represented as ellipsoids. Furthermore we presume homogeneity in chemical composition, density and radionuclide concentration for the source. Because the attenuation coefficient of soil was unknown, we use the attenuation coefficient of concrete. Because of the close chemical and physical similarity (density) of both soil and concrete, and tissue and water, the Taylor exposure build-up factor coefficients (A_1, α_1 and α_2) for concrete and water were used, in stead of the ones for soil and tissue, respectively to calculate the build-up factor.

IV.1.1. Point-Kernel technique for calculating gamma DCC

A well-known method for the assessment of gamma-ray and neutrons shielding is the Point-Kernel technique. It is designed for estimating the effects of radiation (e.g. γ and n) that originate in a volume-distributed source. The program calculates gamma-ray fluxes and normalized dose rates at discrete locations within a source-geometry configuration by representing a volume-distributed source by a number of point isotropic sources and computing the distances through all regions traversed by the line-of-sight from the source points to a desired receiver point. In our cast this is the centre of the reference organism.

From these distances and the characteristics of the materials within them, energy-dependent attenuation factors and build-up factors for gamma-rays are applied to calculate the direct gamma-ray dose with build-up. These build-up factors take into account the effect of scattering.

The mass attenuation coefficient (μ/ρ) and mass energy-absorption coefficient (μ_{en}/ρ) for different energy levels of gamma radiations were published by [IV.1].

The build-up factor is, in the passage of radiation through a medium, the ratio of the total value of a specified radiation quantity at any point to the contribution to that value from radiation reaching the point without having undergone a collision. In our model the build-up factor was calculated using the Taylor development [IV.2]. The equation for calculation the build-up factor is a function of the distance from the source and has following form:

$$B'_x(E,r) = A.\exp\left[-\alpha_1(E).\mu_x(E).r\right] + (1-A).\exp\left[-\alpha_2(E).\mu_x(E).r\right] \qquad \text{(IV.1)}$$

where:

$B'_x(E,r)$ is the build-up factor for a material x, with a attenuation coefficient μ_x (also energy dependent!), for a photon with energy level E traveling a distance r. A, α_1 and α_2 represent the Taylor parameters. $\mu_x(E).r$ is also called the main free path of a particle traversing material x. To avoid overestimations of the absorbed dose the build-up factor is kept constant when the mean free path exceeds 50.

As an example the energy deposition due to the photon flux in the biota hosted on a contaminated soil is calculated. The mathematical formulation is written as follows:

$$\phi_{photon,s_1} = \int_0^{\frac{\pi}{2}} \int_D \int_0^{h} \int_0^{\infty} \frac{S.R}{\pi\left(h^2 + R^2\right)}.B_{tiss}(E,k_d).e^{(-\mu_{soil}.d_g)}.e^{(-\mu_{air}.k_{air})}.e^{(-\mu_{tiss}.k_d)}.dR.dh.d\phi \quad \text{(IV.2)}$$

where:

E energy of the photon (1 J = 6.24151 × 10^{12} MeV);
Φ angle, projected in the horizontal plane, between the ray and the z-axis of the coordinate system;
$\Phi_{photon,s1}$ photon flux (m^{-2} s^{-1}) received by the reference organism living on a contaminated soil (scenario 1);
S level of contamination (Bq cm^{-3}).

$$D_{gamma,s_1} = E.\phi_{photon,s_1}.\mu_{en} \quad \text{(IV.3)}$$

where:

μ_{en} energy absorption coefficient (cm^2 g^{-1});
$D_{gamma,s1}$ Dose Conversion Coefficient (μGy hr^{-1} per Bq kg^{-1} (dw)) for the reference organism living on a contaminated soil (scenario 1).

IV.1.2. Application of the Bethe-Bloch equation for calculating the beta DCC

Unlike the neutral radiations (e.g. neutrons and gamma/X rays), the charged particles (e.g. electrons, protons and alphas) are subjected to the coulomb forces from electrons within the material through which they pass. The loss of energy by charged particles travelling through a material is constituted into two components: The electronic energy loss due to Coulomb interactions (i.e. the ionization and excitation; $(dE/dx)_{col}$), and the nuclear energy loss (e.g. due to emission of *Bremsstrahlung* or Cerenkov radiation, and nuclear interactions; $(dE/dx)_{rad}$). Both losses combined gives the total stopping power:

$$\frac{dE}{dx} = \left(\frac{dE}{dx}\right)_{col} + \left(\frac{dE}{dx}\right)_{rad} \quad \text{(IV.4)}$$

Excitation raises an electron to a higher energy shell, whereas ionization completely removes the electron from the atomic energy shell. Ionization creates an ion pair, which consists of the (now) free electron and the positively charged atom from which the electron was removed. The freed electron may possess sufficient kinetic energy to cause further ionization events (such energetic electrons are sometimes called delta rays).

Betas are easily scattered due to their small mass and charge. The electrons travel a non-linear path. Their range in air is on the order of meters. The collision energy loss for fast electrons may be computed from the Beth-Bloch equation [IV.3; IV.4]:

$$-\left(\frac{dE}{dx}\right)_{col,x} = \frac{2\pi.e^4.z^2}{m_e.v^2} N_x.Z_{eff,x}.B \qquad \text{(IV.5)}$$

where:

z atomic number of the ionizing particle (z=1 for β, p);
e unit electrical charge = $(r_o.m_e.c^2)^{0.5}$;
r_o Bohr electron radius = 2.818×10^{-13} cm;
m_e rest mass of an electron = 9.1085×10^{-31} kg;
c speed of light in a vacuum = 299792458 m/s;
v velocity of the ionizing particle;
N_x number of absorber atoms per cm³ of medium x;
$Z_{eff,x}$ effective atomic number of the absorber x.

B consists of different terms:

$$B = \left[\ln\left(\frac{m_e.v^2.E_{kin}^{\max}}{2.W_x^2.(1-\beta^2)}\right) - \ln 2.\left(2\sqrt{1-\beta^2} - 1 + \beta^2\right) + \left(1-\beta^2\right) + \frac{(1-\sqrt{1-\beta^2})}{8} \right] \qquad \text{(IV.6)}$$

in addition to the prior nomenclature:

β fraction of the speed of light particle is travelling = v/c;
W_x mean excitation and ionization potential of absorber atoms;
E_{kin}^{max} maximum kinetic energy of the electron.

Besides ionisation and excitation, energy loss is also possible due to *Bremsstrahlung*. This is electromagnetic radiation produced by the acceleration of a charged particle (e.g. electron), when deflected by another charged particle, such as an atomic nucleus. The energy loss, which accompanies this electromagnetic radiation, can be calculated using following equation:

$$-\left(\frac{dE}{dx}\right)_{rad,x} = \frac{N_x.E_{kin}^{\max}.Z_{eff,x}.(Z_{eff,x}+1.).e^4}{137.m_e.c^2}\left[4.\ln\left(\frac{2E_{kin}^{\max}}{m_e.c^2}\right) - \frac{4}{3}\right] \qquad \text{(IV.7)}$$

IV.1.3. Dose due to beta irradiation for an organism on a contaminated soil

The energy of a beta particle, that was emitted from a contaminated soil, lost energy during its flight and reached the surface of the reference organism, was calculated using following equation:

$$E'_{beta}(x_{soil}, x_{air}) = E_{kin}^{\max} + \left[\left(\frac{dE}{dx}\right)_{soil}.x_{soil}\right] + \left[\left(\frac{dE}{dx}\right)_{air}.x_{air}\right] \qquad \text{(IV.8)}$$

The DCC for a reference organism living on a contaminated soil is obtained after integration:

$$D_{beta,s_1} = \int_0^{\frac{\pi}{2}} \int_D^h \int_0^{\infty} \frac{S.R}{\pi\left(h^2 + R^2\right)} . E_{beta}(x_{soil}, x_{air}) \frac{Area}{2.Wgt} dR.dh.d\phi \quad \text{(IV.9)}$$

where:

S level of contamination (Bq cm^{-3});
$E_{beta}(x_{soil},x_{air})$ energy of the beta particle after passing the soil and the air;
D distance from centre reference organism (RO) to top soil;
h distance from centre RO to bottom soil;
R distance from RO to origin rays (in horizontal plane);
a,b,c axis of the ellipsoid representing the RO;
$D_{beta,s1}$ Dose Conversion Coefficient (μGy hr^{-1} per Bq kg^{-1} (dw)) for a reference organism living on contaminated soil;
$Area$ surface area of the RO;
Φ angle, projected in the horizontal plane, between the ray and the z-axis of the coordinate system;
Wgt calculated weight of the RO ($4/3.\pi.a.b.c.\rho_{tiss}$).

IV.1.4. Application of the Bethe-Bloch equation for calculation the alpha DCC

As for beta particles, the loss of energy by alpha particles travelling through a material can also be calculated using the Bethe-Bloch equations [IV.3– IV.5]. Because these particles have a higher charge, are larger and don't need a relativistic considerations, a slightly altered form of the Bethe-Bloch equation is used. Also no *Bremsstrahlung* needs to be taken in to account. The stopping power is calculated using following equation:

$$-\left(\frac{dE}{dx}\right) = \frac{4.\pi.e^4.z^2}{m_e.v^2} N_x.Z_{eff,x}.\left[\ln\left(\frac{2.m_e.v^2}{W_x}\right) - \ln\left(1 - \frac{c^2}{v^2}\right) - \frac{c^2}{v^2}\right] \quad \text{(IV.10)}$$

where:

z atomic number of the ionizing particle (z=2 for α);
e unit electrical charge = $(r_o.m_e.c^2)^{0.5}$;
r_o Bohr electron radius = 2.818×10^{-13} cm;
m_e rest mass of an electron = 9.1085×10^{-31} kg;
c speed of light in a vacuum = 299792458 m/s;
v velocity of the ionizing particle;
N_x number of absorber atoms per cm³ of medium x;
$Z_{eff,x}$ effective atomic number of the absorber;
W_x mean excitation and ionization potential of absorber atoms.

IV.1.5. Dose due to alpha irradiation for an organism on a contaminated soil

The energy of an alpha particle, that was emitted from a contaminated soil, lost energy during its flight and reached the surface of the reference organism, was calculated using following equation:

$$E'_{alpha}(x_{soil}, x_{air}) = E_{kin}^{ini} + \left[\left(\frac{dE}{dx}\right)_{soil} .x_{soil}\right] + \left[\left(\frac{dE}{dx}\right)_{air} .x_{air}\right] \quad \text{(IV.11)}$$

The DCC is obtained after integration:

$$D_{alpha,s_1} = \int_0^{\frac{\pi}{2}} \int_D^h \int_0^{\infty} \frac{S.R}{\pi\left(h^2 + R^2\right)} . E_{alpha}(x_{soil}, x_{air}) \frac{Area}{2.Wgt} dR.dh.d\phi \qquad \text{(IV.12)}$$

where:

$D_{alpha,s1}$ Dose Conversion Coefficient (μGy/hr per Bq kg^{-1} (dw)) of a reference organism on a contaminated soil.

IV.1.6. Calculation of the Dose Conversion Coefficient for radionuclides

All calculations were made for **mono-energetic photons, electrons and alpha particles** for different energy ranges. Radionuclide specific dose conversion coefficients are determined by interpolation, taken into account the nuclide-specific energies emitted and their emission probabilities. The values of the absorbed dose rate normalized per starting photon/electron/alpha particle and volume unit give a dose conversion coefficient with can be defines as:

$$DCC_{RN,RO,S} = \sum_i \sum_j y_{RN,i} \times D_{E,RO,S} \qquad \text{(IV.13)}$$

where:

$DCC_{RN,RO,S}$ Dose Conversion Coefficient for radionuclide RN, reference organism RO and scenario S;
$D_{E,RO,S}$ Energy dependent Dose Conversion Coefficient for reference organism RO and scenario S;
$y_{RN,I}$ radionuclide dependent yield;
j index for the radiation type (α, β or γ radiation);
i index for the energy.

IV.2. Bioaccumulation model

Soil-plant transfer factors and CR values for invertebrates, fish, zooplankton and phytoplankton are predominantly derived from review and other publications [IV.6– IV.13]. The equation can be written as:

$C_{plant} = C_{soil} \cdot TF_{soil-plant}$ for terrestrial species and
$C_{invertebra\ te} = C_{soil} \cdot BAF_{invertebra\ te}$ and $C_{fish} = C_{water} \cdot CF_{waterorganism}$ for aquatic species.

where:

C_{plant} Concentration in plant (Bq kg^{-1} (fw));
C_{soil} Concentration in soil (Bq kg^{-1} (dw));
$TF_{soil\text{-}plant}$ Soil plant transfer factor (dw/fw);
$C_{invertebrate}$ Concentration in invetebrate (Bq kg^{-1} (fw));
$BAF_{invertebrate}$ Bioaccumulation factor (dw/fw);
C_{fish} Concentration in fish (Bq kg^{-1} (fw));
C_{water} Concentration in water (Bq l^{-1});
$CF_{waterorganism}$ Concentration factor (l kg^{-1} (fw)).

Table IV.1. Allometric constants used to calculate the ingestion rate [IV.26].

Organism	**A (d^{-1})**	**b(-)**
Herbivorous Mammals	0.0875	0.727
Carnivorous Mammals	0.0687	0.822
Rodent	0.0306	0.564
Duck	0.0582	0.651
Bird Egg	0.0141	0.85

In general, transfer factors for transfer specifically to animals (or animal products) are referred to as TF , F_m and/or F_f (the latter two symbols mostly refer to transfer to milk and meat respectively). For dynamic modelling it is necessary to know the fractional uptake from the gastrointestinal tract to the systemic circulation and the retention into the animal body. This is described through the use of an f_1 value where retention in organs and tissues are characterised as $R_{organ}(t)$. The fraction of the initial amount of activity entering the systemic circulation from a time zero (t_0) can be represented as:

$$TF_{organism}(t_1) = f_1 \int_{t_0}^{t_1} \frac{\sum R_{organ}(t)dt}{M_{organism}(t)} \qquad \text{(IV.14)}$$

where f_1 is the fractional gastrointestinal absorption, $M_{organism}$ is the mass of the organism (kg fw) and t_1 is the time (e.g. year) until where the concentration needs to be calculated (most of the time this is equal to the lifetime of the animal). Elementarily the values used for fractional gastrointestinal absorption and retention into the organism are species specific.

The fractional absorption of the radionuclide from the gastrointestinal tract and the ability of the radionuclide to be retained by the animal body was obtained from [IV.14– IV.25].

For deriving the radionuclide concentration in the organism it is necessary to know daily food intake rates. These are linked to body mass and physiological state. In our approach we use the allometric relation between body mass and intake rate derived by [IV.26] to describe the relation between body weight and intake rate of terrestrial mammal and birds:

$$IR_{organism}(t) = \frac{a \cdot \left[BW_{organism}(t)\right]^b}{DW_{food}} \qquad \text{(IV.15)}$$

where:

$IR_{organims}$ Ingestion rate of the organism (kg (fw) day^{-1});
$BW_{organism}$ Body weight (kg (fw));
DW_{food} Dry weight fraction of the food (-);
b Allometic constant used to calculate the ingestion rate (-);
a Allometic constant used to calculate the ingestion rate (d^{-1}).

The values used for a and b are shown in Table IV.1.

Table IV.2. Birth, maximum weight and allometric constants used to calculate the body weight.

Organism	Maximum weight (kg)	Birth weight (kg)	Lifespan (year)	n (-)	k (-)
Herbivorous Mammals	35	1.6	8	-1.2	0.009
Carnivorous Mammals	80	0.45	10	-2.5	0.02
Rodent	0.027	0.00219	1.67	-0.5	0.05
Duck	1	0.051	21	-0.5	0.05
Bird Egg	0.03	0.003	19	-0.5	0.05

The variation in body weight of the organism due to growth is taken into account by using the Richards equation [IV.27]:

$$BW_{organism}(t) = \left[A^n - \left(A^n - Y^n\right) \cdot e^{-k \cdot t}\right]^{\frac{1}{n}} \quad \text{(IV.16)}$$

where:

A Maximum weight of the organism (kg (fw))
Y Birth weight of the organism (kg (fw))
t Time at which the body weight needs to be derived (e.g. days, i.e. Lifespan of the animal)
n Empirical constant used to calculate the body weight at time t (-)
k Empirical constant used to calculate the body weight at time t (-)

Table IV.2 presents the values used for these parameters.

For some species also the ingestion of soil was taken into account. The reference species used in the latter approach are *Capreolus capreolus* for Herbivorous Mammals, *Canis lupus* for Carnivorous Mammals, *Apodemus sylvaticus*, *Anas platyrhynchos* for Duck and Passer *domesticus* for Bird Egg.

REFERENCES

[IV.1] HUBBELL, H., SELTZER, S.M., Tables of X-Ray Mass Attenuation Coefficients and Mass Energy-Absorption Coefficients - Ionizing Radiation Division, Physics Laboratory National Institute of Standards and Technology Gaithersburg, MD 20899 (1996), (last update July 2004). http://physics.nist.gov/PhysRefData/XrayMassCoef/cover.html

[IV.2] AMERICAN NUCLEAR SOCIETY, Gamma-ray attenuation coefficients and buildup factors for engineering materials, ANSI/ANS-6.4.3-1991 (1992).

[IV.3] FITZGERALD, J.J., BROWNELL, G.L., MAHONEY, F.J., Mathematical Theory of Radiation Dosimetry, Gordon and Breach Science Publishers, Inc (1967).

[IV.4] ATTIX, F.H., Introduction to radiological physics and radiation dosimetry, A Wiley-Interscience Publication John Wiley& Sons (1986).

[IV.5] ZIEGLER, J.F., The Stopping of Energetic Light Ions in Elemental Matter, J. Appl. Phys / Rev. Appl. Phys. **85** (1999) 1249-1272.

[IV.6] INTERNATIONAL ATOMIC ENERGY AGENCY, Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments. Technical Report Series 364, IAEA, Vienna (1994).

[IV.7] SWEECK, L., ZEEVAERT. TH., VOLCKAERT, G., VANDECASTEELE, C., Geologische berging van geconditioneerd langlevend hoog radioactief afval – Biosfeerparameters in performantie- en veiligheidsanalyse Deel II, SCK•CEN Report R-3194 (1998).

[IV.8] RADHAKRISHNA, A.P., SOMASEKHARAPPA, H.M., NARAYANA, Y., SIDDAPPA, K., Distribution of Some Natural and Artificial Radionuclides in Mangalore Environment of South India - Journal of Environmental Radioactivity **30** 1 (1996) 31–54.

[IV.9] LINSALATA, P., MORSE, R., FORD, H., EISENBUD, M. FRANCA, E.P., DE CASTRO, M.B., LOBAO, N., SACHETT, I., CARLOS, M.,Transport Pathways of Th, U, Ra and La from soil to Cattle Tissue - Journal of Environmental Radioactivity **10** (1989) 115-140.

[IV.10] MARTÍNEZ-AGUIRRE, A., GARCÍA-ORELLANA, I., GRACIA-LEÓN, M., Transfer of natural radionuclides from soils to plant in a marsh enhanced by the operation of non-nuclear industries - Journal of Environmental Radioactivity **35** (1997) 149–171.

[IV.11] SANTCHI, P.H., HONEYMAN, B.D., Radionuclides in aquatic environments – Radiation physics and chemistry **34** (1989) 213–240.

[IV.12] SAMPLE, B.E., M.S., APLIN, R.A., EFROYMSON, G.W. SUTER, II., WELSH, C.J.E., Methods and tools for estimation of the exposure of terrestrial wildlife to contaminants. Oak Ridge National Laboratory, Oak Ridge TN. ORNL/TM-13391 (1997).

[IV.13] GARTEN, C.T., JR., DAHLMAN R.C., Plutonium in biota from an east Tennessee floodplain forest, Health Physics, **34** (1978) 705–712.

[IV.14] COUGHTREY, P.J., THORNE, M.C., Radionuclide Distribution and Transport in Terrestrial and Aquatic Ecosystems, Volume 1, AA Balkema, Rotterdam (1983).

[IV.15] BERESFORD, N.A., MAYES, R.W., COOKE, A.I., BARNETT, C.L., HOWARD, B.J., LAMB, C.S., NAYLOR, G.P.L., The importance of source dependent bioavailability in determining the transfer of ingested radionuclides to ruminant derived food products. Environ Sci and Tech, **34** (2000) 4455–4462.

[IV.16] COUGHTREY, P.J., JACKSON, D., JONES, C.H., KANE, P., THORNE, M.C., Radionuclide Distribution and Transport in Terrestrial and Aquatic Ecosystems, Volume 4. AA Balkema, Rotterdam (1984).

[IV.17] TAYLOR, D.M., The absorption of plutonium and related elements from the gastrointestinal tract: a re-appraisal. Institute for Genetics and Toxicology, W Germany (1981).
[IV.18] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION , Age-dependent Doses to Members of the Public from Intake of Radionuclides: Part 3 – Ingestion Dose Coefficients. ICRP Publication 69, Ann ICRP 25 Pergamon Press, Oxford (1995).
[IV.19] ARGONNE NATIONAL LABORATORY, Polonium Human Health Fact Sheet (2005a) http://www.evs.anl.gov/pub/doc/polonium.pdf
[IV.20] ARGONNE NATIONAL LABORATORY, Technetium Human Health Fact Sheet (2005b) http://www.evs.anl.gov/pub/doc/technetium.pdf
[IV.21] ARGONNE NATIONAL LABORATORY, Radium Human Health Fact Sheet (2005c) http://www.ead.anl.gov/pub/doc/Radium.pdf
[IV.22] COUGHTREY, P.J., JACKSON, D., THORNE, M.C., Radionuclide Distribution and Transport in Terrestrial and Aquatic Ecosystems, Volume 3, AA Balkema, Rotterdam (1983).
[IV.23] COUGHTREY, P.J., JACKSON, D., THORNE, M.C., Radionuclide Distribution and Transport in Terrestrial and Aquatic Ecosystems, Volume 6, AA Balkema, Rotterdam (1985).
[IV.24] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Limits for Intakes of Radionuclides by Workers – Part 3, ICRP Publication 30, Ann ICRP 6 Pergamon Press, Oxford (1981).
[IV.25] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Limits for Intakes of Radionuclides by Workers, ICRP Publication 30, Part 1, Annals of the ICRP, **2** 3/4 (1979).
[IV.26] NAGY, K.A., Field metabolic rate and food requirement scaling in mammals and birds. Ecol. Monogr. **57** (1987) 111-128.
[IV.27] LAWRENCE, T.L.J., FOWLER, V.R., Growth of Farm Animals, CAB International, Wallingford, Oxon., OX10 8DE (1997).

APPENDIX V. PERCH LAKE SCENARIO INSTRUCTIONS

The Perch Lake Freshwater Scenario Description

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V.1. Background information

Past routine environmental monitoring programs at nuclear facilities have traditionally focused on the protection of humans with the implicit assumption that non-human biota would also be protected. However, recent paradigm shifts in national and international views on environmental protection are leading to the development of more detailed evaluations to also assess potential risks to non-human biota. Part of the reasoning behind these changes in international ideology pertains to ensuring the protection of the environment by its own right, as opposed to relying on the traditional belief that if humans are protected, then the environment will also be adequately protected following exposure to radionuclides.

Although human dosimetric models are well-developed, much less work has been done to develop approaches to evaluate radiological doses received by non-human biota (particularly for non-mammalian species) and their potential impacts on receptor species. As a result, extensive efforts are underway in many countries to develop methodologies to estimate radionuclide levels in the environment, their corresponding doses to non-human biota, as well as their potential effects in natural ecosystems for various types of habitats and receptor species (with varying sizes, shapes, radiosensitivities, habitat-use patterns and exposure pathways) (e.g. FASSET; RESRAD-BIOTA and others).

In order to work towards ensuring that standardized and comparable approaches are being applied internationally to protect the environment from the potential impacts of ionizing radiation, the IAEA has formed the Biota Working Group (BWG) under its EMRAS (Environmental Modeling for Radiation Safety) program. The primary objective of the EMRAS BWG (which was developed and agreed upon at the EMRAS plenary meeting that was held in Vienna, Austria in November 2004) is *to compare and validate models being used and developed by Member States for biota dose assessment (that may be used) as part of regulatory process of licensing and compliance monitoring of authorized releases of radionuclides, in order to improve Member State's capabilities for protection of the environment*.

The focus of the EMRAS BWG will initially be placed on comparison of international approaches that are currently being applied to estimate radionuclide exposure and dose to non-human biota, with future plans to then link exposure to effects. To accomplish these objectives, a number of aquatic and terrestrial scenarios are being developed for testing by the EMRAS BWG.

V.1.1. Objective of Scenario

The objective of the *Perch Lake Freshwater Scenario* is to compare model predictions of ^{90}Sr, ^{137}Cs, ^{60}Co and tritium activity concentrations in a range of freshwater non-human biota receptor species with measured data to evaluate various modeling approaches being applied internationally. Predicted activity concentrations will subsequently be used to derive unweighted absorbed dose rates.

V.1.2. Proposed Participants

Proposed participants in the *Perch Lake Freshwater Scenario* include:

- RESRAD-BIOTA (United States);
- EA R&D 128 (UK);
- ERICA/FASSET (European Union);
- AECL (Canada);
- ECOMOD (Russia);
- SCK-CEN (Belgium);
- LIETDOS-BIO (Lithuania).

Potentially for dosimetry only:

- EDEN (France)
- EPIC-DOSES3D (European Union INCO-Copernicus)

V.2. Description of study site

Perch Lake (Figures V.1 to V.3), a lake on Atomic Energy of Canada Limited (AECL)'s Chalk River Laboratories site on which the *Perch Lake Freshwater Scenario* is based, has received chronic, low-level inputs of ^{90}Sr, tritium, ^{60}Co and ^{137}Cs over a period of approximately 50 years (Figures V.4 to V.6). As a result, Perch Lake surface waters are routinely monitored as part of AECL's routine environmental monitoring program and the lake has been extensively studied historically, as well as in the recent past.

A detailed description of the physico-chemical attributes, as well as the species composition and ecology, of the Perch Lake ecosystem is provided in Annex I to provide context for consideration in scenario testing.

V.3. Summary of available data for Perch Lake receptor species

Perch Lake represents a relatively diverse and productive wetland habitat that is likely comparable to wetland ecosystems in other parts of the world. As a result, it is possible to estimate doses to a number of receptor biota, as part of the *Perch Lake Freshwater Scenario*. A brief overview of the resident biota species, as well as the radiological data available for Perch Lake biota is provided in the sections that follow and is summarized in Tables V.1 to V.6. Additional information on the ecological attributes of receptor species can be found by accessing the web-sites provided in Table V.7. A list of the predictions to be made for specific receptor species and radionuclides as part of this scenario is provided in Table V.8.

V.3.1. Primary Producers

Radiological data have been collected intermittently for a number of broad categories of primary producers in Perch Lake, including free-floating (unrooted) species; rooted, submergent macrophytes; rooted, floating-leafed macrophytes; and emergent species.

A list of available radiological data for each vegetation species and category is provided in Table V.1.

V.3.2. Plankton and Invertebrate Communities

In general, freshwater invertebrate receptor species have been sub-divided into four cateogories, which include zooplankton, macroinvertebrates (as listed in Section V.2 above), snails and mussels.

A list of available radiological data for each category has been given in Table V.2.

V.3.3. Perch Lake Herpetofauna

Limited radiological data have been collected for a number of amphibian and reptilian species inhabiting Perch Lake, as summarized in Table V.3. Key herpetofauna for which radiological data exist include tadpoles, adult frogs and turtles.

V.3.4. Avian Species

Although a number of avian species make use of habitat in the Perch Lake watershed, as well as in the lake itself (as listed in Table V.4), it was out of the scope of this scenario to provide radiological data for birds.

V.3.5. Mammalian Receptors

In addition to the radionuclide data available for primary producers, aquatic invertebrates and herpetofauna, limited data are also available for freshwater mammalian species, including the star-nose mole and the American water shrew, both representing carnivorous small mammals (Table V.5).

V.3.6. Perch Lake Fish Species

As discussed in Section I-6 of Annex I, the species composition and abundance of the Perch Lake fish community changed significantly following the introduction of northern pike, a top predatory fish species, into the lake in the mid- to late-1980s (Table V.6). Despite these changes, for the most part (i.e. with the exception of northern pike and yellow perch), it remains possible to estimate radionuclide transfer to Perch Lake fishes based on historical and more recent radionuclide levels in Perch Lake surface waters.

V.4. Input data

A summary of the time-points for which radiological data are available for sediments and/or receptor biota has been compiled in Table I-1 (of Annex I). Based on this summary, three time periods have been selected by the EMRAS BWG for inclusion in the *Perch Lake Freshwater Scenario*. These include the period between 1968 and 1971 (for doses from ^{90}Sr, ^{60}Co and/or ^{137}Cs), the period between 1994 and 1998 (for doses from ^{90}Sr, ^{60}Co and/or ^{137}Cs), as well as the period between 2003 and 2004 (for doses from tritium).

Measured ^{90}Sr, ^{60}Co, ^{137}Cs and ^{3}H concentrations in Perch Lake surface waters have been provided in Table V.9, whereas radionuclide concentrations in sediments have been provided in Table V.10 for these time periods. In addition, measured data for key physicochemical attributes of Perch Lake have been provided in Table V.11 to serve as context regarding lake conditions for models that require such information.

V.5. Model outputs

As discussed in Section V.3 above, Table V.8 provides a list of the predictions to be made for specific receptor species, radionuclides and time points as part of the *Perch Lake Freshwater Scenario*.

Based on the information provided in Tables V.9 to V.11, Participants will be expected to perform calculations applying methodologies and assumptions typically applied when conducting an assessment. Such assumptions should be clearly specified as part of the scenario output, in addition to any difficulties that arose during the course of the calculations.

Model outputs will include the following for each species and each radionuclide:

(1) Whole-body activity concentrations for key receptor species (in Bq/kg fresh weight);
(2) Internal unweighted dose rates; and
(3) External dose rates.

Results should then be entered into a Microsoft Excel spreadsheet (entitled EMRAS_BWG_Perch Lake Freshwater Scenario results sheet (participant name).xls) using the structure outlined in Tables V.12 and V.13. Participants should also provide values for each input parameter used in the model, as appropriate (some models may not use all parameters).

It is expected that one Excel worksheet will be filled in for each receptor and time-point (as listed in Table V.8) to facilitate compilation. This represents a total of 67 Excel sheets (i.e. PL1 to PL67 in Table V.8), if all predictions are made.

— Worksheets cells are colour coded
— yellow – entry required
— grey – no entry required [e.g. if a given nuclide is not included in a specific calculation]
— black – no entry required [no reportable parameter]

Note black and grey cells are locked.

Additional measurements can be taken in the lake as follow-up to scenario completion.

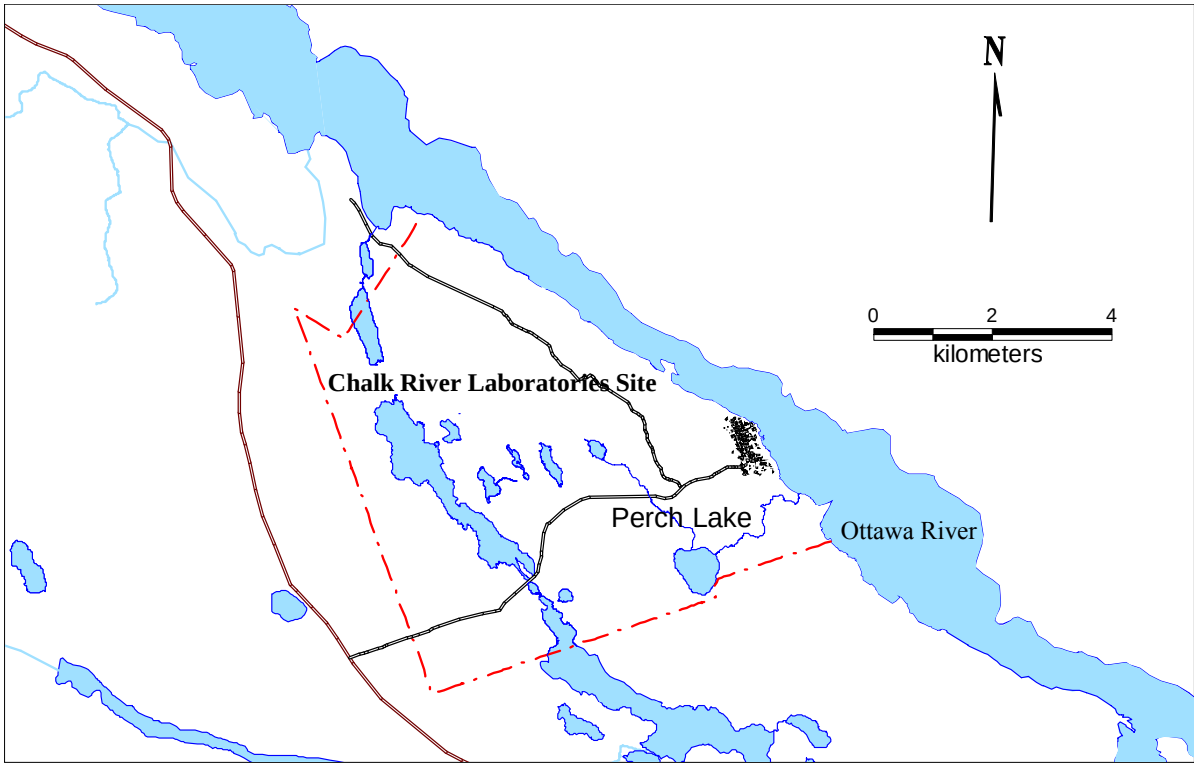

Fig. V.1. Map depicting the location of Perch Lake.

Fig. V.2. Aerial photograph depicting shallow littoral zone of Perch Lake.

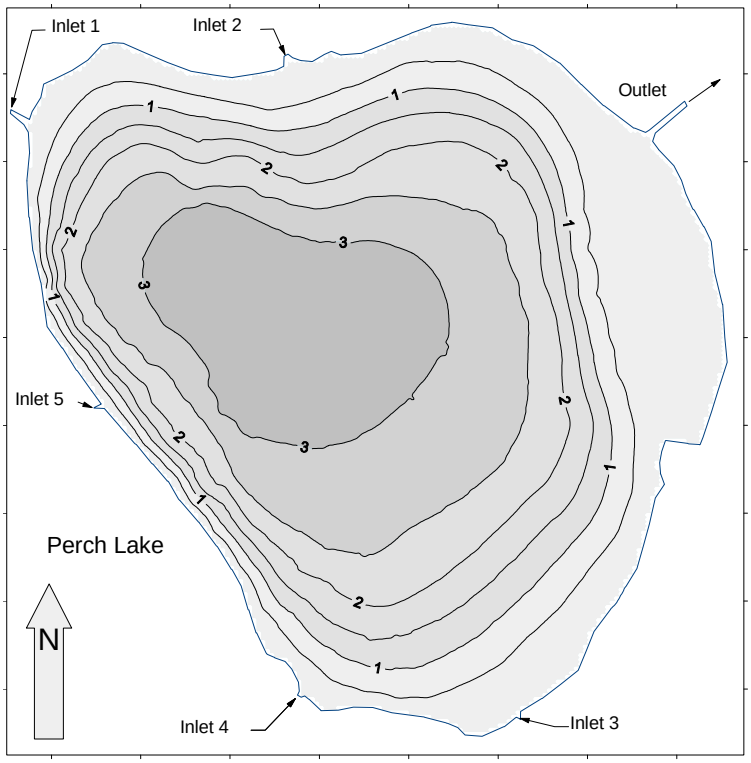

Fig. V.3. Depth contour map of Perch Lake water depth (in metres) to the gyttja or sediment surface (after Jay, 1975). Littoral zone represents areas with water depths of less than 1.5 m.

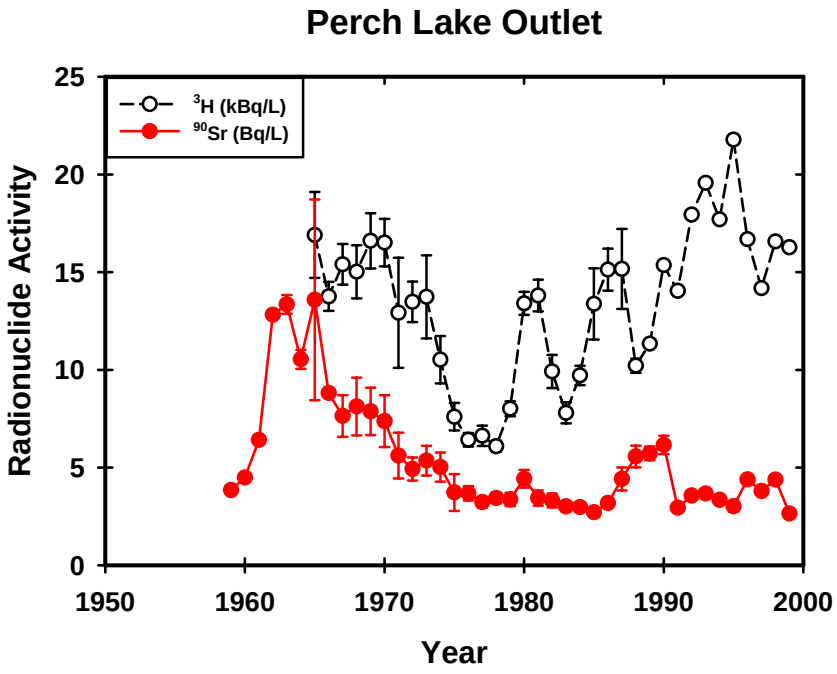

Fig. V.4. Mean ^{3}H and ^{90}Sr concentrations (± standard error) in surface waters depicting temporal trends at Perch Lake outlet (from Yankovich et al.*, 2000).*

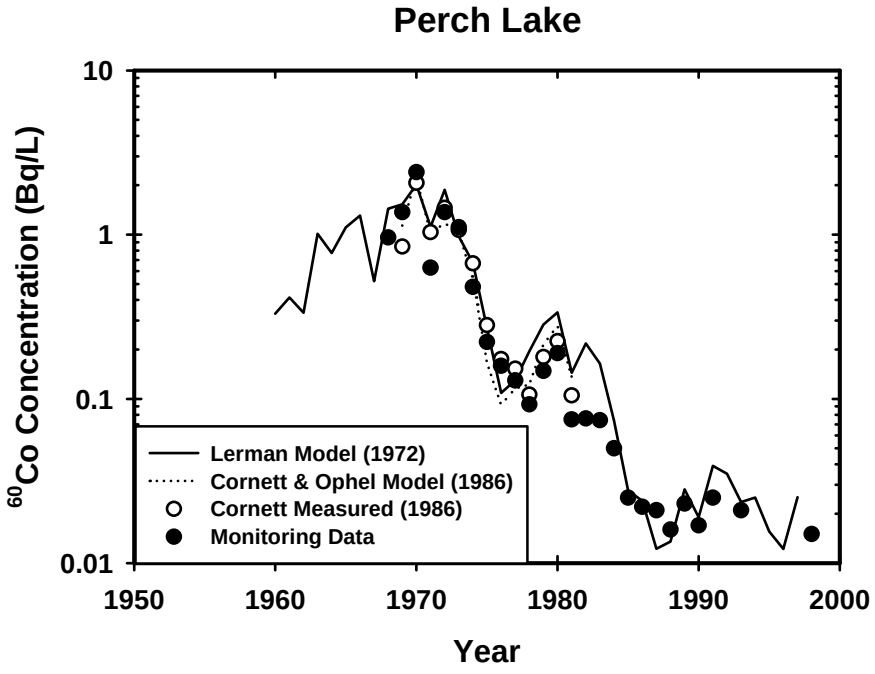

Fig. V.5. Comparison of measured and modeled data depicting temporal trends in ^{60}Co concentrations in Perch Lake surface waters (from Yankovich et al.*, 2000).*

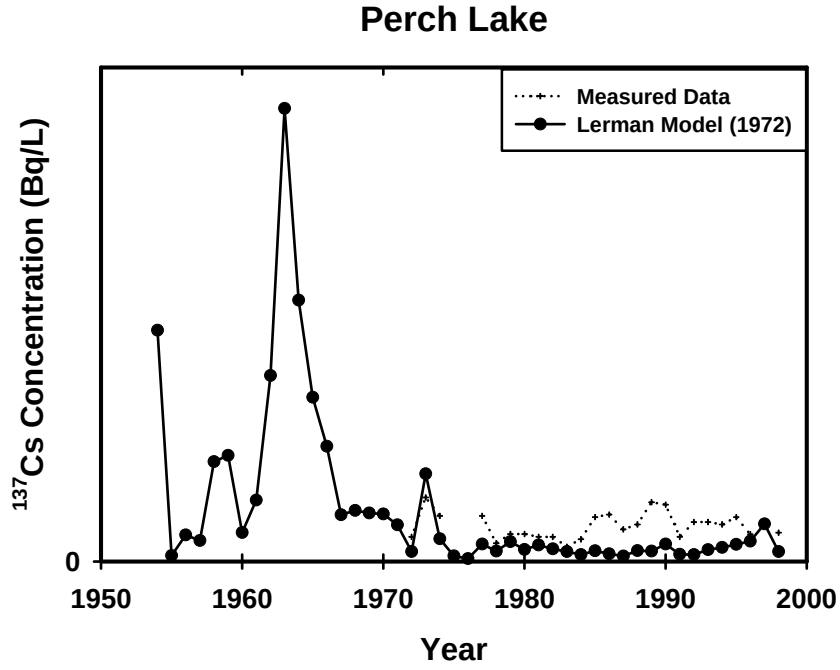

Fig. V.6. Comparison of measured and modeled data depicting temporal trends in ^{137}Cs concentrations in Perch Lake surface waters (from Yankovich et al.*, 2000).*

Table V.1. Inventory of key Perch Lake emergent and aquatic macrophyte species for which radiological data are available.

Common Name	Scientific Name	^{90}Sr	^{60}Cs	^{137}Cs	Tritium
Emergent Macrophytes:					
Common cattail	*Typha latifolia*	×	×	×	×
Bur-reed spp.	*Sparganium americanum* *Sparganium emersum* *Sparganium fluctuans*	×	×	×	
Sedge spp.	*Carex* spp.	×			
Bulrush spp.	*Scirpus acutus* *Scirpus americanus*	×	×	×	
Pickerelweed	*Pontederia cordata*	×	×	×	
Aquatic Macrophytes:					
Free-floating (unrooted), Submergent Species:					
Phytoplankton	Various species	×	×	×	
Epiphyton	*Spirogyra* spp.	×	×	×	×
Bladderwort spp.	*Utricularia cornuta* *Utricularia gibba* *Utricularia intermedia* *Utricularia minor* *Utricularia purpurea* *Utricularia vulgaris*	×	×	×	×
Coontail spp.	*Ceratophyllum demersum* *Ceratophyllum echinatum*	×	×	×	
Stonewort (also called Muskgrass)	*Chara* sp.	×	×	×	
Water nymph	*Najas flexilis*	×	×	×	
Rooted, Submergent Species:					
Pipewort	*Eriocaulon septangulare*	×	×	×	
Pondweed spp.	*Potamogeton amplifolius* *Potamogeton epihydrus* *Potamogeton foliosus* *Potamogeton gramineus* *Potamogeton natans* *Potamogeton pusillus*	×	×	×	
Rooted, Floating-Leafed Species:					
Watershield	*Brasenia schreberi*	×	×	×	
Fragrant white water lily	*Nymphaea odorata*	×	×	×	
Yellow pond lily	*Nuphar variegatum*	×	×	×	

Table V.2. Inventory of invertebrate species that utilize the Perch Lake watershed.

Common Name	Scientific Name	^{90}Sr	^{60}Cs	^{137}Cs	Tritium
Zooplankton	Various species	×	×	×	×
Snail spp.	*Amnicola* spp. *Heliosoma* spp.	×	×	×	
Macroinvertebrates	Various species	×	×	×	
Barnes Mussel	*Elliptio complanata*	×	×	×	×

Table V.3. Inventory of Perch Lake herpetofauna species for which radiological data are available.

Common Name	Scientific Name	^{90}Sr	^{60}Cs	^{137}Cs	Tritium
Amphibians:					
Bullfrog	*Rana catesbeiana*	×	×	×	
Green frog	*Rana clamitans*	×	×	×	
Aquatic Reptiles:					
Painted turtle	*Chrysemys picta*	×	×	×	
Snapping turtle	*Chelydra serpentina*	×	×	×	
Northern water snake	*Nerodia sipedon*				

Table V.4. Inventory of bird species that utilize Perch Lake.

Common Name	Scientific Name
Great blue heron	*Ardea herodias*
American black duck	*Anas rubripes*
Mallard duck	*Anas platyrhynchos*
Wood duck	*Aix sponsa*
Blue-winged teal	*Anas discors*
Green-winged teal	*Anas crecca*
Canada goose	*Branta canadensis*
[a] Common loon	*Gavia immer*
[b] Double-crested cormorant	*Phalacrocorax auritus*
Belted kingfisher	*Ceryle alcyon*
Red-winged blackbird	*Agelaius phoeniceus*

[a] Only makes transient use of the lake in early spring while waiting for larger lakes in the area to thaw.
[b] Only documented once in the lake.

Table V.5. Inventory of Perch Lake mammalian species for which radiological data are available.

Common Name	Scientific Name	^{90}Sr	^{60}Cs	^{137}Cs	Tritium
[a] American beaver	*Castor canadensis*				
American mink	*Mustela vison*				
American water shrew	*Sorex palustris*	×	×	×	
Moose	*Alces alces*				
Muskrat	*Ondatra zibethicus*				
River otter	*Lontra canadensis*				
Star-nose mole	*Condylura cristata*	×	×	×	

[a] Beaver lodges are often built in inflowing and outflowing streams near the lake. A young beaver was observed in Perch Lake in the early spring for 2 to 3 weeks, but no beaver lodges have been observed in the lake itself.

Table V.6. Temporal changes in presence-absence data for the Perch Lake fishes.

Common Name	Scientific Name	Type of Fish Species	1968–1971	1973	1975	1980	1994–2003
Northern pike	*Esox lucius*	Piscivore	[1]Absent	Absent	Unknown	Absent	Present
Yellow perch	*Perca flavescens*	Piscivore	Present	Present	Present	Present	Present (n = 8)
Brown bullheads	*Ameirus nebulosis*	Benthivore	Present	Present	Present	Present	Present
Pumpkinseeds	*Lepomis gibbosus*	Forage species	Present	Present	Present	Present	Present
Lake chub	*Couesius plumbeus*	Forage species (cyprinid)	[2]Unknown	Present	Present	Absent	Absent
Creek chub	*Semotilus atromaculatus*	Forage species (cyprinid)	Unknown	Absent	Unknown	Present	Absent
Pearl dace	*Margariscus margarita*	Forage species (cyprinid)	Unknown	Present	Present	Present	Absent
Blacknose shiners	*Notropis heterolepis*	Forage species (cyprinid)	Unknown	Absent	Unknown	Absent	Present
Bluntnose minnows	*Pimephales notatus*	Forage species (cyprinid)	Unknown	Absent	Unknown	Present	Absent
Fathead minnows	*Pimephales promelas*	Forage species (cyprinid)	Unknown	Present	Unknown	Present	Absent

[1] *Absent* means that the fish species was absent from the lake catch records for a given sampling time point.
[2] *Unknown* means that the fish species was not documented, but does not necessarily mean that the species was absent from the lake, since a detailed fish survey of the lake had not been carried out at the time.

Table V.7. Relevant internet web-sites describing key receptor species considering in the Perch Lake Freshwater Scenario being tested by the EMRAS Biota Working Group (BWG).

Receptor	Scientific Name	Relevant Internet Web-Site
Aquatic Primary Producers	General	http://www.ecy.wa.gov/programs/wq/plants/plantid2/categories.html
Barnes Mussel	*Elliptio complanata*	http://research.amnh.org/biodiversity/mussel/elliptiogenustext.html
Bullfrog	*Rana catesbeiana*	http://www.fcps.k12.va.us/StratfordLandingES/Ecology/mpages/bullfrog.htm
Green frog	*Rana clamitans*	http://museum.nhm.uga.edu/gawildlife/amphibians/anura/ranidae/rclamitans.html
Painted turtle	*Chrysemys picta*	http://www.fcps.k12.va.us/StratfordLandingES/Ecology/mpages/eastern_painted_turtle.htm
Snapping turtle	*Chelydra serpentina*	http://www.fcps.k12.va.us/StratfordLandingES/Ecology/mpages/common_snapping_turtle.htm
Northern water snake	*Nerodia sipedon*	http://www.fcps.k12.va.us/StratfordLandingES/Ecology/mpages/northern_water_snake.htm
American water shrew	*Sorex palustris*	http://www.nhest.org/penquis/penquismammals.html#shrews
Star-nose mole	*Condylura cristata*	http://www.nhest.org/penquis/penquismammals.html#shrews
Northern pike	*Esox lucius*	http://www.pikezander.co.uk/pike.htm
Yellow perch	*Perca flavescens*	http://www.fcps.k12.va.us/StratfordLandingES/Ecology/mpages/yellow_perch.htm
Brown bullheads	*Ameiurus nebulosis*	http://www.issg.org/database/species/ecology.asp?si=612&fr=1&sts=
Pumpkinseeds	*Lepomis gibbosus*	http://www.combat-fishing.com/fishencyclo1/sunfish/pumpkinseed.htm
Lake chub	*Couesius plumbeus*	http://fish.dnr.cornell.edu/nyfish/Cyprinidae/lakechub.html
Creek chub	*Semotilus atromaculatus*	http://www.iowadnr.com/fish/iafish/crc-card.html
Pearl dace	*Margariscus margarita*	http://www.iowadnr.com/fish/iafish/ped-card.html
Blacknose shiners	*Notropis heterolepis*	http://www.iowadnr.com/fish/iafish/bks-card.html
Bluntnose minnows	*Pimephales notatus*	http://www.iowadnr.com/fish/iafish/bnm-card.html
Fathead minnows	*Pimephales promelas*	http://www.iowadnr.com/fish/iafish/fhm-card.html
General	General	http://animaldiversity.ummz.umich.edu/site/index.html
General	General	http://www.bbc.co.uk/nature/wildfacts/animals_a_z.shtml

Table V.8. Summary of receptor species and radionuclides for which predictions to be made as part of the Perch Lake Freshwater Scenario. Each prediction to be made for a given time point, receptor and radionuclide has been given a number (i.e. PL#). X's have been placed in cells where radiological data are available.

Medium or Species	Radionuclide	1968	1969	1970	1971	1994	1995	1996	1997	1998	2003	2004
Abiotic Media												
Surface Water	^{90}Sr	×	×	×	×	×	×	×	×			
	^{60}Co	×	×	×	×	×	×	×	×			
	^{137}Cs	×	×	×	×	×	×	×	×			
	Tritium	×	×	×	×	×	×	×	×		×	×
Sediments	^{90}Sr	×	×	×	×	×	×	×	×	×		
	^{60}Co	×	×	×	×	×	×	×	×	×		
	^{137}Cs	×	×	×	×	×	×	×	×	×		
	Tritium										×	×
Non-Human Biota:												
Aquatic Primary Producers:												
Free-floating (unrooted) Submergents	^{90}Sr	PL1			PL24	PL31	PL39	PL45	PL51			
	^{60}Co	PL1	PL6			PL31	PL39					
	^{137}Cs					PL31	PL39					
	Tritium										PL60	PL65
Rooted, Submergent Macrophytes	^{90}Sr				PL25		PL40	PL46				
	^{60}Co	PL2	PL7	PL13			PL40	PL46				
	^{137}Cs						PL40	PL46				
	Tritium											
Rooted, Floating-Leafed Macrophytes	^{90}Sr	PL3	PL8	PL14	PL26	PL32	PL41	PL47				
	^{60}Co	PL3	PL8	PL14	PL26	PL32	PL41	PL47				
	^{137}Cs	PL3	PL8	PL14	PL26	PL32	PL41	PL47				
	Tritium											
Emergent Macrophytes	^{90}Sr			PL15	PL27	PL33	PL42	PL48				
	^{60}Co				PL27	PL33	PL42	PL48				
	^{137}Cs					PL33	PL42	PL48				
	Tritium										PL61	
Aquatic Invertebrates:												
Zooplankton	^{90}Sr					PL34						
	^{60}Co					PL34						
	^{137}Cs					PL34						
	Tritium											PL66
Macroinvertebrates	^{90}Sr			PL16		PL35						
	^{60}Co					PL35						
	^{137}Cs					PL35						
	Tritium											
Snails	^{90}Sr											
	^{60}Co			PL17								
	^{137}Cs											
	Tritium											
Freshwater Mussels	^{90}Sr		PL9									
	^{60}Co											
	^{137}Cs											
	Tritium										PL62	PL67
Fishes:												
Forage Fishes:												
Cyprinid Species	^{90}Sr			PL18	PL28	PL36			PL52			
	^{60}Co			PL18		PL36	PL43					
	^{137}Cs											
	Tritium											
Pumpkinseeds	^{90}Sr				PL29	PL37		PL49	PL53			
	^{60}Co		PL10	PL19	PL29	PL37			PL53			
	^{137}Cs											
	Tritium											
Benthivorous Species:												
Brown Bullhead	^{90}Sr					PL38	PL44					
	^{60}Co	PL4	PL11	PL20		PL38	PL44	PL50				
	^{137}Cs											
	Tritium										PL63	

Medium or Species	Radionuclide	1968	1969	1970	1971	1994	1995	1996	1997	1998	2003	2004
Piscivorous Species												
Yellow Perch	^{90}Sr ^{60}Co ^{137}Cs Tritium	PL5	PL12	PL21								
Northern Pike	^{90}Sr ^{60}Co ^{137}Cs Tritium										PL64	
Amphibians:												
Green Frogs	^{90}Sr ^{60}Co ^{137}Cs Tritium									PL58 PL58 PL58		
Bullfrogs	^{90}Sr ^{60}Co ^{137}Cs Tritium			PL22						PL59 PL59 PL59		
Reptiles:												
Painted Turtle	^{90}Sr ^{60}Co ^{137}Cs Tritium				PL30				PL54 PL54 PL54			
Common Snapping Turtle	^{90}Sr ^{60}Co ^{137}Cs Tritium			PL23					PL55 PL55 PL55			
Aquatic Mammals:												
Star-nose Mole	^{90}Sr ^{60}Co ^{137}Cs Tritium								PL56 PL56 PL56			
American Water Shrew	^{90}Sr ^{60}Co ^{137}Cs Tritium								PL57 PL57 PL57			

Table V.9. Mean annual concentrations of key radionuclides in Perch Lake surface waters during years for which sediment and/or biota radiological data are available (n.a. – data not available).

Year	Radionuclide Activity Concentration in Water (Bq/L)											
	Strontium-90 (^{90}Sr)			Cesium-137 (^{137}Cs)			Cobalt-60 (^{60}Co)			Tritium (^{3}H)		
	Mean	Minimum	Maximum	Mean	Minimum	Maximum	Mean	Minimum	Maximum	Mean	Minimum	Maximum
1968	8.13	5.79	13.3	8.03E-02	n.a.	n.a.	0.521	n.a.	n.a.	15,017	7,300	20,800
1969	7.88	5.82	12.3	7.64E-02	n.a.	n.a.	1.44	n.a.	n.a.	16,650	9,000	27,000
1970	7.38	4.29	11.0	7.47E-02	n.a.	n.a.	1.53	n.a.	n.a.	16,511	9,600	24,700
1971	5.61	3.07	9.0	5.73E-02	n.a.	n.a.	2.01	n.a.	n.a.	13,988	2,900	31,300
1994	3.35	2.12	4.87	1.10E-02	9.22E-08	2.20E-02	0.0235	n.a.	n.a.	12,152	7,667	24,470
1995	3.02	0.84	5.76	1.34E-02	9.22E-08	2.68E-02	0.0251	n.a.	n.a.	14,955	15,196	42,082
1996	4.39	3.22	5.32	1.59E-02	1.92E-15	3.19E-02	0.0156	n.a.	n.a.	11,457	11,640	22,268
1997	3.80	1.67	5.53	5.86E-02	5.80E-02	5.93E-02	0.0122	n.a.	n.a.	9,737	10,122	19,611
1998	4.38	2.59	5.77	8.39E-03	1.30E-03	1.55E-02	0.0252	n.a.	n.a.	11,380	10,693	22,256
[a] 2003	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	4,723	2,030	9,350
[a] 2004	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	4,905	n.a.	n.a.

[a] Based on water data collected between May and October.

Table V.10. Concentrations of key radionuclides in Perch Lake sediments during years for which biota radiological data are available.

Year	^{137}Cs (Bq/kg dry)		^{60}Co (Bq/kg dry)		^{90}Sr (Bq/kg dry)		^{3}H (Bq/L)	
	Gyttja	Sand	Gyttja	Sand	Gyttja	Sand	[a] HTO	[b] OBT
1968	103	16.6	1,219	197	71.3	11.5	n.a.	n.a.
1969	104	16.8	1,219	197	3.47	0.561	n.a.	n.a.
1970	105	16.9	1,483	240	3.03	0.490	n.a.	n.a.
1971	105	17.0	2,031	328	4.39	0.709	n.a.	n.a.
1994	76.0	12.3	455	73.4	2,231	347	n.a.	n.a.
1995	74.7	12.1	405	65.5	2,352	366	n.a.	n.a.
1996	73.6	11.9	363	58.6	2,575	401	n.a.	n.a.
1997	72.5	11.7	323	52.1	2,723	424	n.a.	n.a.
1998	72.0	11.6	286	46.3	2,839	442	n.a.	n.a.
2003	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	3,885	1,236
2004	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	3,905	1,161

n.a. – not applicable

[a] HTO represents tritiated water in sediment porewaters.

[b] OBT represents organically-bound tritium in sediment particles.

Table V.11. Summary of parameter values describing key Perch Lake physico-chemical attributes.

Parameter	Units	n	Mean	Standard Error	Minimum	Maximum
pH	Not applicable	9	6.73	0.175	5.5	7.64
Conductivity (Cond.)	$\mu S \cdot cm^{-1}$	6	82.3	14.1	42.3	125
Dissolved Oxygen (D.O.)	ppm	4	8.50	0.500	8.00	10.0
Dissolved Organic Carbon (DOC)	$mg \cdot L^{-1}$	9	10.8	1.50	5.50	15.5
Dissolved Inorganic Carbon (DIC)	$mg \cdot L^{-1}$	9	10.4	2.74	2.99	26.8
Total Dissolved Solids (TDS)	$meq \cdot L^{-1}$	2	1.96	0.135	1.82	2.09
Calcium (Ca)	$mg \cdot L^{-1}$	9	6.90	0.219	6.00	7.50
Magnesium (Mg)	$mg \cdot L^{-1}$	9	2.40	0.133	1.68	2.74
Sodium (Na)	$mg \cdot L^{-1}$	9	9.25	1.31	2.64	12.1
Potassium (K)	$mg \cdot L^{-1}$	9	0.912	2.95E-02	0.730	1.00
Strontium (Sr)	$mg \cdot L^{-1}$	7	4.58E-02	2.99E-03	3.40E-02	5.40E-02
Cesium (Cs)	$mg \cdot L^{-1}$	9	< 9.00E-05	n.a.	n.a.	n.a.
Cobalt (Co)	$mg \cdot L^{-1}$	9	< 2.00E-04	n.a.	n.a.	n.a.
Alkalinity	$mg\ HCO_3^- \cdot L^{-1}$	9	17.7	0.897	13.2	23.2
Nitrate (NO_3^-)	$mg \cdot L^{-1}$	9	0.307	0.108	$< 5 \times 10^{-2}$	0.870
Phosphate (PO_4^{3-})	$mg \cdot L^{-1}$	9	< 0.2	n.a.	n.a.	n.a.
Sulphate (SO_4^{2-})	$mg \cdot L^{-1}$	9	5.15	0.854	< 9.00E-02	8.83
Chloride (Cl^-)	$mg \cdot L^{-1}$	9	15.4	2.73	< 0.02	22.9
Bromide (Br^-)	$mg \cdot L^{-1}$	4	< 0.07	n.a.	n.a.	n.a.
Fluoride (F^-)	$mg \cdot L^{-1}$	9	< 0.1	n.a.	n.a.	n.a.

Note: Although a detailed evaluation of Perch Lake water chemistry was conducted in both 2001 and 2003, temporal trends in these parameters are not available. However, sporadic measurements of key parameters (e.g. pH, Ca, Mg, K, Na and others) in the past relative to current measurements are comparable, suggesting that it is likely reasonable to assume that water quality conditions have been similar over time in Perch Lake.

Table V.12. List of output parameters, parameter codes and units for the Perch Lake Freshwater Scenario.

Model Parameter	Model Parameter Code	Units
Internal dose conversion coefficient	DCCI	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (fresh weight)
External dose conversion coefficient from water	DCCEW	$\mu Gy \cdot h^{-1}$ per BqL^{-1}
External dose conversion coefficient from sediments	DCCES	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (dry weight)
External dose conversion coefficient from vegetation	DCCEV	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (fresh weight)
Activity concentration in whole biota	Cb	$Bq \cdot kg^{-1}$ (fresh weight)
Concentration ratio	CR	$L \cdot kg^{-1}$ (fresh weight)
Internal dose	DI	$\mu Gy \cdot h^{-1}$
External dose from water	DEW	$\mu Gy \cdot h^{-1}$
External dose from sediments	DES	$\mu Gy \cdot h^{-1}$
External dose from vegetation	DEV	$\mu Gy \cdot h^{-1}$
Total dose	DTot	$\mu Gy \cdot h^{-1}$

Table V.13. Parameter output sheet for the Perch Lake Freshwater Scenario.

Model Used:
Name of Participant:
Receptor Species:
Sediment Type:
Reference Organism Used:
Organism Geometry Used:
Assumed Occupancy Factors:
Media Geometry Assumptions:
Year of Exposure:

Radionuclide	Model Parameter	Parameter Units	Parameter Value	Scenario Outputs	Description of How Parameter/Output was Derived	Difficulties Encountered
Strontium-90 (^{90}Sr)	Internal dose conversion coefficient	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (fresh weight)				
	External dose conversion coefficient from water	$\mu Gy \cdot h^{-1}$ per $Bq \cdot L^{-1}$				
	External dose conversion coefficient from sediments	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (dry weight)				
	External dose conversion coefficient from vegetation	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (fresh weight)				
	Concentration ratio	$L \cdot kg^{-1}$ (fresh weight)				
	Activity concentration in biota	$Bq \cdot kg^{-1}$ (fresh weight)				
	Internal dose	$\mu Gy \cdot h^{-1}$				
	External dose from water	$\mu Gy \cdot h^{-1}$				
	External dose from sediments	$\mu Gy \cdot h^{-1}$				
	External dose from vegetation	$\mu Gy \cdot h^{-1}$				
	Total dose	$\mu Gy \cdot h^{-1}$				
Cesium-137 (^{137}Cs)	Internal dose conversion coefficient	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (fresh weight)				
	External dose conversion coefficient from water	$\mu Gy \cdot h^{-1}$ per $Bq \cdot L^{-1}$				
	External dose conversion coefficient from sediments	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (dry weight)				
	External dose conversion coefficient from vegetation	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (fresh weight)				
	Concentration ratio	$L \cdot kg^{-1}$ (fresh weight)				
	Activity concentration in biota	$Bq \cdot kg^{-1}$ (fresh weight)				
	Internal dose	$\mu Gy \cdot h^{-1}$				
	External dose from water	$\mu Gy \cdot h^{-1}$				
	External dose from sediments	$\mu Gy \cdot h^{-1}$				

Radionuclide	Model Parameter	Parameter Units	Parameter Value	Scenario Outputs	Description of How Parameter/Output was Derived	Difficulties Encountered
	External dose from vegetation	$\mu Gy \cdot h^{-1}$				
	Total dose	$\mu Gy \cdot h^{-1}$				
Cobalt-60 (^{60}Co)	Internal dose conversion coefficient	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (fresh weight)				
	External dose conversion coefficient from water	$\mu Gy \cdot h^{-1}$ per $Bq \cdot L^{-1}$				
	External dose conversion coefficient from sediments	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (dry weight)				
	External dose conversion coefficient from vegetation	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (fresh weight)				
	Concentration ratio	$L \cdot kg^{-1}$ (fresh weight)				
	Activity concentration in biota	$Bq \cdot kg^{-1}$ (fresh weight)				
	Internal dose	$\mu Gy \cdot h^{-1}$				
	External dose from water	$\mu Gy \cdot h^{-1}$				
	External dose from sediments	$\mu Gy \cdot h^{-1}$				
	External dose from vegetation	$\mu Gy \cdot h^{-1}$				
	Total dose	$\mu Gy \cdot h^{-1}$				
Tritium (^{3}H)	Internal dose conversion coefficient	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (fresh weight)				
	External dose conversion coefficient from water	$\mu Gy \cdot h^{-1}$ per $Bq \cdot L^{-1}$				
	External dose conversion coefficient from sediments	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (dry weight)				
	External dose conversion coefficient from vegetation	$\mu Gy \cdot h^{-1}$ per $Bq \cdot kg^{-1}$ (fresh weight)				
	Concentration ratio	$L \cdot kg^{-1}$ (fresh weight)				
	Activity concentration in biota	$Bq \cdot kg^{-1}$ (fresh weight)				
	Internal dose	$\mu Gy \cdot h^{-1}$				
	External dose from water	$\mu Gy \cdot h^{-1}$				
	External dose from sediments	$\mu Gy \cdot h^{-1}$				
	External dose from vegetation	$\mu Gy \cdot h^{-1}$				
	Total dose	$\mu Gy \cdot h^{-1}$				

[a] Note that an EMRAS scenario has been completed for Perch Lake to estimate free-water tritium (HTO) and organically-bound tritium (OBT) concentrations in Perch Lake biota by the Tritium/^{14}C Working Group. The objective of including tritium in the *Perch Lake Freshwater Scenario* for the Biota Working Group (BWG) is to test how other/biota model results compare.

DETAILED DESCRIPTION OF THE PERCH LAKE STUDY SYSTEM

V.6. Background information

Perch Lake (the lake on which this scenario is based) is an aquatic system that was studied by the EMRAS Tritium/C14 Working Group in an earlier EMRAS scenario that was designed for model-data validation of steady state free-water tritium (FWT) and organically-bound tritium (OBT) models for a range of freshwater receptor species. Perch Lake is also currently being studied by the EMRAS Tritium/C14 Working Group as part of a second scenario, to assess tritium (FWT and OBT) uptake by transplanted freshwater mussels under dynamic exposure conditions.

Perch Lake represents an ideal system for such scenarios, since Perch Lake has received chronic, low-level inputs of a number of radionuclides (including ^{90}Sr, HTO, ^{60}Co, ^{137}Cs) over a period of approximately 50 years; Perch Lake surface waters are routinely monitored as part of AECL's routine environmental monitoring program; the lake has been extensively studied historically, as well as in the recent past; additional measurements can be taken in the lake to fill in any gaps that have been identified during scenario development or as follow-up to scenario completion; and Perch Lake represents a relatively diverse and productive wetland habitat that is likely comparable to wetland ecosystems in other parts of the world. A detailed description of Perch Lake is provided in the sections that follow to provide context for consideration in scenario testing.

V.7. Physical Attributes and Limnology of Perch Lake

Perch Lake is a littorally-dominated Canadian Shield lake located on Atomic Energy of Canada Limited (AECL)'s Chalk River Laboratories (CRL) site (Figure V.1). The lake is small and shallow, with a mean depth of 2 m, a maximum depth of 4.1 m, a surface area of 450,000 m^2 and a volume of 910,000 m^3 (Figure V.2). The lake fetch is approximately 880 m and the surface area of the Perch Lake watershed is 5.65×10^6 m^2.

Perch Lake typically freezes sometime between mid-November to early-December, and thaws in mid-April. The lake water warms rapidly following ice-melt due to its small volume and shallow depth. Summer water temperatures range from approximately 18 to 34 ^{o}C, often exceeding 30^{o}C [V.1]. Most of the lake remains unstratified due to its shallow depth, although weak stratification does occur in the deeper areas of the lake during the summer, with surface water temperatures of approximately 5 ^{o}C higher than those at the lake bottom. Based on historical measurements, mean monthly water temperatures are 13, 19, 24, 23, 19 and 11^o C for the months of May through October, respectively. Oxygen levels at the bottom of the lake can become depleted during the summer and winter, but do not tend to reach full anaerobic conditions [V.2].

Perch Lake can be described as a dystrophic-eutrophic lake due to its high humic content and the clear signs of nutrient enrichment in the lake, corresponding to its maturity. Increasing numbers of floating hummocks consisting of decomposing plant materials from the sediment surface have been observed in the lake over the past 5 to 10 years, particularly during the autumn, and existing floating hummocks have become more extensive in terms of their size and the plant growth they support. The pH of the lake water ranges from 5.5 to 7.6 and the surface water in Perch Lake is highly coloured due to the presence of humic and fulvic acids, which originate from decomposing plant materials flowing in through surface streams, as well

as the decay of vegetation in the lake itself. Perch Lake contains a low total dissolved solid content, with a value of 1.96 meq/L.

Perch Lake sustains a relatively large biomass of aquatic plants, which dieback each year and contribute to the accumulation of organic sediments, or gyttja, in the lake bottom. Organic sediment deposits in Perch Lake are 5 to 6 m thick, on average, and are as thick as 8 m in the deepest part of the lake [V.3]. Perch Lake sediments primarily consist of fine, semi-liquid, highly organic mud composed of decomposing plant debris, known as gyttja, with sandy areas located on the west side of the lake. The percent water content of the Perch Lake gyttja is approximately 93%, whereas that of the sandy sediments is approximately 50% (by weight). The mean dry bulk density is approximately 185 kg m^{-3} for Perch Lake sediments, but values vary substantially across the lake depending on the local composition of the sediments. In addition, lake gyttja becomes denser with increasing depth.

Below the organic sediments, is a layer of clay and possibly gravel [V.4], whereas the upper layers of the sediments primarily consist of plant material and detritus in varying stages of decomposition. The rate of sediment deposition in the lake is approximately 0.06 $cm \cdot a^{-1}$ or 0.16 $kg \cdot m^{-2} \cdot a^{-1}$, with a constant sedimentation rate being observed over time [V.5].

Lake water levels are maintained due to the presence of a series of dykes surrounding the lake, which were constructed between 1966 and 1967 to control lake levels and to separate inflowing from outflowing water in the basin for the purposes of hydrological assessment and routine environmental monitoring [V.6]. Culverts with gauged weirs have been built through the dykes at points of major stream flow [V.6], so that the rates of stream flow entering and leaving the lake can be monitored. Despite the dykes, however, Perch Lake water levels fluctuate seasonally. During the spring, lake levels rise due to inputs of meltwater to the lake, and during the late summer to early autumn, the lake level can drop by as much as 0.25 m. Precipitation events in the mid- to late-autumn restore lake levels to typical values.

V.8. Hydrology of the Perch Lake Watershed

The Perch Lake catchment consists of 7 sub-basins, five of which contribute water to the lake through inflowing surface streams, which enter the lake at Inlets 1 to 5, and one of which drains into the lake through sub-surface groundwater inflow [V.7]. The seventh sub-basin, known as the Creek sub-basin, is situated between Perch Lake Outlet and the Ottawa River. No organized stream is present in this sub-basin, although a fairly large volume of sub-surface flow enters Perch Creek in this area [V.7]. Water exits Perch Lake via the outlet, with a residence time of water in the lake of approximately 0.5 years. Therefore, radionuclides that reach the lake, and are not complexed in the sediments or the tissues of resident biota, can be transported out of the lake relatively rapidly.

V.9. Radionuclide Inputs to Perch Lake

Perch Lake has received radionuclides, including ^{90}Sr, ^{60}Co, ^{137}Cs and ^{3}H, since the mid-1950s. The radionuclides originate from atmospheric deposition from global nuclear weapons testing, as well as through the influx of historic waste from upstream Waste Management Areas (WMAs) [V.8, V.9].

Radionuclides from upstream sites infiltrate into groundwater and pass through a series of wetlands and streams before entering Perch Lake. These wetlands adjoin upstream swamp habitats, which are drained by a number of surface streams before entering Perch Lake via

Inlets 1 and 2. Most of the radionuclide inventory enters Perch Lake via the major surface streams, which are monitored at Inlets 1 and 2 of the lake, although a relatively small fraction of the radionuclides also enter the lake through diffuse sub-surface groundwater inflow [V.1, V.10]. Water flowing into the lake through Inlets 3, 4 and 5 does not contribute significantly to the radionuclide budget of the lake, since these streams drain areas with background radionuclide levels. Instead, these streams may serve to dilute radionuclides in the lake to varying extents depending upon their rates of discharge into the lake. Water that leaves the lake via the outlet passes through Perch Creek and is later discharged into the Ottawa River.

Radionuclide concentrations and rates of surface water discharge entering and leaving the lake are routinely measured at several locations in the Perch Lake watershed to monitor radionuclide transfer through the watershed and to ensure regulatory compliance is met. Routine monitoring of radionuclide activities in surface water entering and leaving Perch Lake was initiated in April 1956; however, early data were sporadic and sampling locations were not always well defined. A formal environmental monitoring program was established for the CRL site in 1964. Since then, water samples have been routinely collected at several locations at and near the CRL site (e.g. Figure V.3).

In most cases, water sampling in the Perch Lake watershed has focused on measurement of concentrations in the inflowing streams and at the outflow, but not in the lake itself; however, due to the small size of the lake, it is likely reasonable to assume that the lake is uniformly mixed and that radionuclide concentrations are representative of those present in the lake itself. Comparison of ^{90}Sr and ^{60}Co concentrations in surface waters collected at the outflow relative to the lake centre confirm this assumption.

It is important to note that, although measurable at the lake inlets, ^{60}Co concentrations in water collected at the lake outflow and in the lake itself often fall below analytical detection limits and therefore, cannot always be used to quantify the ^{60}Co levels. As a result, it was necessary to estimate ^{60}Co concentrations in Perch Lake surface waters for some time-points using mathematical modelling, as outlined in [V.9] (Figure V.4). Similarly, ^{137}Cs is rarely detectable in Perch Lake surface waters (although, it can be periodically detected in the inflowing waters), likely because ^{137}Cs tends to adsorb to sediment and soil particulates, causing most of it to remain in the system, as opposed to being transferred out of the lake via the outlet. As a result, as for ^{60}Co, in some cases, it was necessary to estimate ^{137}Cs levels in Perch Lake surface waters using a mass balance approach, as described in [V.9] (Figure V.5).

Currently, ^{90}Sr and ^{3}H represent the dominant radionuclides in the Perch Lake watershed (Figure V.4), although ^{60}Co levels were easily detectable in the past, but have been declining due to the lack of inputs combined with losses due to radiological decay (Figure V.5). Cesium-137 has been detectable in the lake during times of peak atmospheric deposition due to global weapons testing and is still detectable in Perch Lake sediments (Figure V.6), although ^{137}Cs concentrations in resident biota have not been historically measured. Radionuclide transfer to biota inhabiting Perch Lake is influenced by radionuclide partitioning between environmental compartments to which biota are exposed, as well as radionuclide transfer pathways. Archived radionuclide data and samples, which were collected in the Perch Lake system, can, therefore, be used to validate radionuclide transfer models that are being applied internationally for freshwater ecosystems. A summary of the available historical radiological data for each receptor species, as well as for environmental media is provided in TableV.14.

V.10. Ecology of Perch Lake

Perch Lake is a littorally-dominated lake, with a littoral zone that represents approximately 15 ha or 33% of the entire lake surface area (Figures V.1 and V.2). The ecology of the lake and the trophic transfer of key radionuclides in the lake were extensively studied both historically, as well as more recently between the 1990s and the present.

In 1998, an inventory of the species occupying the lake and its watershed was conducted to characterize the ecological significance of the Perch Lake aquatic community (as summarized in Tables V.1 to V.6). Key aspects of the Perch Lake community are discussed in the context of selection of receptor species in the sections that follow. Relevant web-sites for each type of receptor species is provided in Table V.7.

V.10.1. Primary Producers

Surveys of the Perch Lake macrophyte community have been undertaken both historically and in the recent past. The plant community in the lake consists of free-floating macrophytes, floating-leafed species, completely or partially submergent species and emergent species. Plant growth is fairly dense in the littoral zone, but is sparse in deeper, offshore waters exceeding depths of approximately 1.5 m, although some water lilies, watershield and pondweed can be found in the inshore, as well as offshore areas of the lake.

A list of key Perch Lake primary producers for which radiological data exist is provided in Table V.1.

V.10.2. Plankton and Invertebrate Communities

The plankton community and its seasonal distribution in Perch Lake have been reported by Havlik and Ophel in 1970 [V.11]. Detailed characterization of the plankton community is beyond the scope of the current scenario. Instead, the plankton community has been sub-divided into zooplankton and phytoplankton for the purposes of this scenario.

Perch Lake also supports a diverse assemblage of invertebrates in addition to zooplankton. Many of these invertebrate species are benthic, although a number of species that occupy the water column or vegetation can also be found in the lake. The aquatic invertebrate community of Perch Lake is typical of small lakes and ponds, and littoral zones of larger lakes [V.12]. Macroinvertebrate species include diving beetles, giant water bugs, water boatmen, backswimmers, various species of dragonfly and damselfly larvae, mayfly larvae, caddisfly larvae, dobsonfly larvae, mosquito larvae, water striders, water scorpions, water mites, oligochaetes, chironomids and others. There are also various species of crustaceans, snails, freshwater mussels (Unionidae and Sphaeriidae), leeches and worms, which occupy various habitats in the lake, including surface sediments, vegetation and open water.

A summary and description of key invertebrate receptor species to be included in the *Perch Lake Freshwater Scenario* is provided in Table V.2.

V.10.3. Perch Lake Herpetofauna

Nine species of amphibians utilize Perch Lake during at least part of their life cycles. Of these, radiological data are available for two species, which include bullfrogs (*Rana catesbeiana*) and green frogs (*Rana clamitans*) (Table V.3).

Four reptilian species utilize Perch Lake, including common snapping turtles (*Chelydra serpentina*), painted turtles (*Chrysemys picta*), eastern garter snakes (*Thamnophis sirtalis*) and northern water snakes (*Nerodia sipedon*) (Table V.3). Turtles lay their eggs in gravel substrates along the roadside near Inlet 2 and Perch Lake outlet during the early summer. Predatory mammalian species, such as raccoons and skunks, will then prey on the eggs.

V.10.4. Avian Species

Numerous bird species also make direct or indirect use of the lake. These include the Canada goose (*Branta canadensis*), wood duck (*Aix sponsa*), American black duck (*Anas rubripes*), blue-winged teal (*Anas discors*), broad-winged hawk (*Buteo platypterus*), belted kingfisher (*Megaceryle alcyon*), common yellowthroat (*Geothlypis trichas*), swamp sparrow (*Melospiza georgiana*), and red-winged blackbird (*Agelaius phoeniceus*) (Table V.4). Common loons (*Gavia immer*) and other species periodically use the lake in the spring and fall as a migration stopover. A double-crested cormorant (*Phalacrocorax auritus*) was observed fishing in the lake on one occasion in 2001, although this species had not been documented in the lake in the past.

V.10.5. Mammalian Receptors

Mammalian species that utilize Perch Lake either on a regular basis or periodically include muskrat (*Ondatra zibethicus*), American water shrew (*Sorex palustris*), star-nose mole (*Condylura cristata*), moose (*Alces alces*), beaver (*Castor canadensis*), and other species. Radiological data have been collected for water shrews and star-nose moles, as summarized in Table V.5.

V.10.6. Recent Changes to the Perch Lake Fish Community

Historically, the Perch Lake fish community consisted of yellow perch (*Perca flavescens*), pumpkinseeds (*Lepomis gibbosus*), brown bullheads (*Ameiurus nebulosus*), and six cyprinid species, including pearl dace (*Margariscus margarita*), bluntnose minnow (*Pimephales notatus*), creek chub (*Semotilus atromaculatus*), fathead minnow (*Pimephales promelas*), blacknose shiners (*Notropis heterolepis*), and lake chub (*Couesius plumbeus*), were present in Perch Lake (Table V.6).

In the mid- to late-1980s, northern pike (*Esox lucius*) were inadvertently introduced into Perch Lake, resulting in a species replacement at the top of the food chain (whereby northern pike replaced yellow perch) and a corresponding cascade of effects that led to the decline and extirpation of a number of forage fish species (Table V.6), changes in the fish size structure and reduced somatic condition of both brown bullheads and northern pike, which led to the starvation of some individuals. These changes in the fish community not only provide an opportunity for model validation under the EMRAS programme, but also facilitate assessment of the impacts of changes in community structure on radionuclide dynamics in aquatic foodwebs.

Table V.14. Summary of available radiological data for use in the Perch Lake Freshwater Scenario.

Medium or Species	Radionuclide	1954	1955	1956	1957	1958	1959	1960	1961	1962	1963	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Abiotic Media:																																																						
Surface Water	^{90}Sr						×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×
	^{60}Co	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×
	^{137}Cs	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×
	Tritium												×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×	×
Sediments	^{90}Sr							×								×	×	×	×										×			×									×	×	×	×	×						×	(×)
	^{60}Co															×	×	×	×											×			×									×	×	×	×	×						×	(×)
	^{137}Cs						×									×	×	×	×											×			×									×	×	×	×	×						×	(×)
	Tritium																																																			×	×	(×)
Non-Human Biota:																																																						
Aquatic Primary Producers:																																																					
Free-floating (unrooted) Submergents	^{90}Sr															×	×		×																									×	×	×	×	×								
	^{60}Co															×	×																									×	×											
	^{137}Cs																																									×	×											
	Tritium																																																			×	×	
Rooted, Submergent Macrophytes	^{90}Sr																		×																										×	×		×								
	^{60}Co															×	×	×																										×	×		×								
	^{137}Cs																																										×	×		×								
	Tritium																																																					
Rooted, Floating-Leafed Macrophytes	^{90}Sr															×	×	×	×							×																	×	×	×		×	×							
	^{60}Co															×	×	×	×																									×	×	×									
	^{137}Cs															×	×	×	×																									×	×	×									
	Tritium																																																				
Emergent Macrophytes	^{90}Sr																	×	×																									×	×	×		×								
	^{60}Co																		×																									×	×	×		×								
	^{137}Cs																																										×	×	×		×								
	Tritium																																																				
Aquatic Invertebrates:																																																					
Zooplankton	^{90}Sr																																											×												
	^{60}Co																																											×												
	^{137}Cs																																										×												
	Tritium																																																				×	
Macroinvertebrates	^{90}Sr																	×																									×												
	^{60}Co																																											×												
	^{137}Cs																																										×												
	Tritium																																																					
Snails	^{90}Sr																																																						
	^{60}Co																	×																																						
	^{137}Cs																																																					
	Tritium																																																					

Medium or Species	Radionuclide	1954	1955	1956	1957	1958	1959	1960	1961	1962	1963	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Freshwater Mussels	^{90}Sr						×			×	×		×				×																																		×	×	
	^{60}Co																																																			×	
	^{137}Cs																																																			×	
	Tritium																																																				×	(×)
Fishes:																																																					
Forage Fishes:																																																					
Cyprinid Species	^{90}Sr						×	×	×	×	×		×					×	×				×																			×			×	×								
	^{60}Co																	×																									×	×									
	^{137}Cs																																																				
	Tritium																																																				
Pumpkinseeds	^{90}Sr									×	×		×						×				×					×													×		×	×		×						
	^{60}Co																×	×	×				×					×													×			×								
	^{137}Cs																																																				
	Tritium																																																				
Benthivorous Species:																																																					
Brown Bullhead	^{90}Sr									×	×	×	×						×				×																			×	×		×							
	^{60}Co												×				×	×	×				×																			×	×	×								
	^{137}Cs																																																				
	Tritium																																																			×	
Piscivorous Species:																																																					
Yellow Perch	^{90}Sr							×		×	×	×	×																																									
	^{60}Co												×				×	×	×																																			
	^{137}Cs																																																				
	Tritium																																																				
Northern Pike	^{90}Sr																																																				
	^{60}Co																																																				
	^{137}Cs																																																				
	Tritium																																																			×	
Amphibians:																																																				
Green Frogs	^{90}Sr																																																				
	^{60}Co																																																				
	^{137}Cs																																																				
	Tritium																																																				
Bullfrogs	^{90}Sr																	×																																				
	^{60}Co																																																				
	^{137}Cs																																																				
	Tritium																																																				
Reptiles:																																																				
Painted Turtle	^{90}Sr																																																				
	^{60}Co																		×																																		
	^{137}Cs																																																				
	Tritium																																																				

Medium or Species	Radionuclide	1954	1955	1956	1957	1958	1959	1960	1961	1962	1963	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Common Snapping Turtle	^{90}Sr ^{60}Co ^{137}Cs Tritium																	×																										×									
Aquatic Mammals:																																																					
Star-nose Mole	^{90}Sr ^{60}Co ^{137}Cs Tritium																																												×									
American Water Shrew	^{90}Sr ^{60}Co ^{137}Cs Tritium																																											×									

REFERENCES

[V.1] BARRY, P.J., (ed), Hydrological Studies on a Small Basin on the Canadian Shield: A Final Summary of the Perch Lake Evaporation Study 1965–1974 (Volume 2), Atomic Energy of Canada Limited AECL–5041/II (1975c).

[V.2] MCMAHON, J.W., Physical and chemical conditions in Perch Lake, Atomic Energy of Canada Limited Report, AECL-2831 (1966).

[V.3] GÀCS, I., Distribution of some major and minor elements in the Perch Lake system, In: Hydrological Studies on a Small Basin on the Canadian Shield: A Final Summary of the Perch Lake Evaporation Study 1965–1974, P.J. Barry (ed.), Atomic Energy of Canada Limited AECL–5041/I (1975).

[V.4] JAY, P.C., Morphology of Perch Lake, In: Hydrological Studies on a Small Basin on the Canadian Shield: A Final Summary of the Perch Lake Evaporation Study 1965–1974, P.J. Barry (ed.), Atomic Energy of Canada Limited AECL–5041/I, pp 131–138 (1975).

[V.5] BOYKO-DIAKONOW, M., TERASMAE, J., Palynology of Holocene sediments in Perch Lake, Chalk River, Ontario, In: Hydrological Studies on a Small Basin on the Canadian Shield: A Final Summary of the Perch Lake Evaporation Study 1965–1974 (Volume 1), Atomic Energy of Canada Limited AECL–5041/I, pp 189–220 (1975).

[V.6] MERRITT, W.F., RISTO, B.A., Calibration and history of the Perch Lake weirs, In: Hydrological Studies on a Small Basin on the Canadian Shield, P.J. Barry (ed.), Atomic Energy of Canada Limited Report, AECL-5041/1, 299–310 (1975).

[V.7] BARRY, P.J., Perch Lake. In: Hydrological Studies on a Small Basin on the Canadian Shield: A Final Summary of the Perch Lake Evaporation Study 1965–1974 (Volume 1), Atomic Energy of Canada Limited AECL–5041/I (1975a).

[V.8] EISENBUD, M., GESELL, T., Environmental Radioactivity from Natural, Industrial, and Military Sources, 4th Edition, Academic Press: Toronto, 656 pp (1997).

[V.9] YANKOVICH, T.L., KILLEY, R.W.D., KLUKAS, M.H., CORNETT, R.J., ZACH, R., LAFONTAINE, C.R., O'DONNELL, B.C., EVE, T.L., CHAPUT, T.J., BENZ, M.L., HAAS, M.K., The importance of environmental monitoring data in Environmental Risk Assessment: An ecosystem approach, Proceedings from the Canadian Nuclear Society Meeting, Ottawa, Ontario, 20 pp (2000).

[V.10] BARRY, P.J., (ed), Hydrological Studies on a Small Basin on the Canadian Shield: A Final Summary of the Perch Lake Evaporation Study 1965–1974 (Volume 1), Atomic Energy of Canada Limited AECL–5041/I (1975b).

[V.11] HAVLIK, B., OPHEL, I.L., Study of the plankton composition of four small lakes in Ontario, AECL Technical Report No. 03607 (1970).

[V.12] WETZEL, R.G., Limnology, Second Edition, Saunders College Publishing: New York, 767 pp (1983).

APPENDIX VI. CHERNOBYL SCENARIO INSTRUCTIONS

EMRAS BWG: Chernobyl Terrestrial Scenario – Instructions

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20th June 2007 (v3.1)

'Red Forest' soil (photograph: N.A. Beresford CEH)

VI.1. Ammendments within version 2.0 of scenario spreadsheet

We have now restricted estimation of unweighted absorbed dose rate to one example of each species with the exception of those data entries for which TLD measurements are available. Note we now have an additional set of TLD results and these have been included in the scenario (see CT42).

Version 1.1 was found to contain an error in estimation of soil activity concentrations for the CT39 - Version 2.0 contains corrected inputs.

VI.2. Ammendment to version 3.0 of scenario spreadsheet

CT41 Cs-137 and Sr-90 input soil activity concentrations corrected (they had been transposed in original sheet).

VI.2.1. The Chernobyl Exclusion Zone

The predominant terrestrial ecosystems within the Chernobyl exclusion zone are forests; areas previously used for agriculture are becoming scrubland. The majority of the (podzolic, soddy) soils within the exclusion zone are sandy in nature.

VI.2.2. Available data

Data for activity concentrations of radionuclides in biota within the Chernobyl exclusion zone have been compiled from the open literature, IRL data holdings and on-going collaborative studies by CEH and IRL (funded by the EC FP6 project ERICA). Activity concentrations in soil corresponding with these data are presented in the accompanying Excel sheet (mean, minimum and maximum values are given where possible). A random numbering system is used (e.g. CT1 etc.) so that individual data from the published literature cannot be identified easily. In the Excel sheet, where a data ID is suffixed with a lower case letter (e.g. CT29a, CT29b and CT29c) data are for the same site and time (i.e. the soil activity concentrations are the same).

A range of biota types have been selected for this exercise comprising: graminaceous vegetation; invertebrates; birds; wide range of mammal species; amphibians; a reptile. The majority of collated data are for ^{137}Cs and ^{90}Sr, although some data are available for actinide isotopes in small mammals and birds.

A few of the data entries for ^{137}Cs and ^{90}Sr span 1 or more years (e.g. CT12 – CT16). This is justified on the basis of an analyses of data presented by Gaschak et al. (2001) [VI.1]which reports no long-term temporal decline in either ^{90}Sr or ^{137}Cs activity concentrations of wild mammals in the Chernobyl exclusion zone over a similar time period as the collated data used here.

Results from TLDs attached to species of small mammals are available for five of the data entries (CT32a, CT33a, CT33b, CT34a, CT34b and CT42).

VI.2.2.1. Data manipulations for preparation of scenario

Not all biota data had associated soil activity concentrations. Where this was the case soil activity concentrations have been estimated using GIS based deposition maps and relationships between the radionuclides released [VI.2]. Where reported soil results were

given as Bq m^{-2} a soil bulk density of 1100 kg m^{-3} [VI.3] and sampling depth of 10 cm were assumed to estimate a soil activity concentration.

An output of the exercise will be whole-body activity concentrations. For larger animals reported results are often tissue specific. To generate wholebody activity concentrations (for comparison with the model outputs from this exercise) it has been assumed that:

- ^{137}Cs activity concentrations in muscle are equal to those in whole body;
- 90 % of the wholebody ^{90}Sr burden is in bone, and bone contributes 10 % and 7 % of the whole bodyweight of mammals and birds respectively.

Some small mammal results were available as dry matter activity concentrations only, a conversion factor of 0.25 was applied to generated fresh weight activity concentration values.

Results for plutonium isotopes (in soil and biota) are reported in available publications in a number of ways (e.g. ^{238,239,240}Pu, ^{239,240}Pu etc.). To determine isotope specific values, ratios in the release [VI.4] have been assumed to be applicable throughout the exclusion zone (^{238}Pu activities were corrected for decay).

VI.2.2.2. Other information

In addition to soil activity concentrations for some of the amphibian data, water activity concentrations are also available (CT5a, CT5b, CT6a and CT6b). These are presented in Table VI.1 to enable their use in the exercise if specific models/approaches require this.

For one of the *Microtus* (vole) species for which TLD results are available no definitive species name is available (CT32a); the median wholebody weight of these animals was 23 g.

Information on animal size and behaviour which may be required can be found from:

http://www.arkive.org/species/ARK/

http://en.wikipedia.org/wiki/Main_Page

http://www.rspb.org.uk/birds/guide/index.asp

http://genomics.senescence.info/species/

The most useful website for each species is indicated in the spreadsheet.

VI.3. Exercise instructions and reporting

The accompanying Excel sheet contains a number of worksheets providing the scenario soil activity concentrations and others for the recording of model parameters and assumptions, and model outputs.

Table VI.1. Caesium-137 waster activity concentrations at sites for which amphibian data are available (kBq m^{-3}).

BWG Data ID	Mean	Range
CT5a&b	0.37	0.13-0.64
CT6a&b	14	8.7-22

The first worksheet (*Input data*) contains soil activity concentrations, sampling dates and species names. The order of data entries and radionuclides in this worksheet is retained in all of the succeeding sheets.

The next two sheets are for recording: (i) the transfer parameters used (*Transfer parameter values*) and (ii) dose conversion coefficients used and geometry and habitat assumptions (*DCC values*). There is also opportunity to note how these parameters are derived in these worksheets. If your model does not use CR values use the 'Any other notes' column of the *Transfer parameter values* worksheet to note alternative approach used and any specific parameters as appropriate.

Estimated whole body (fresh weight) activity concentrations should be recorded in *Output activity concentrations*.

Unweighted absorbed dose rates should be recorded in *Output dose rates*.

The final worksheet (*Total output dose rates*) sums the individual contributions to dose for each radionuclide (this is done automatically). Predictions of TLD dose rate recordings should also be entered into this sheet.

All cells requiring an entry by the user (if applicable) are shaded green, cells which will automatically update are shaded apricot and cells which do not require an entry are shaded black.

Most worksheets have columns for comments or information on method of calculation etc.. Additional comment columns can be added **but only to the right of the sheet** (i.e. DO NOT ALTER ORDER OF INPUT PARAMETER OR OUTPUT COLUMNS).

Table VI.2. Requested results for the Chernobyl Scenario.

BWG Data ID	Species	Biota activity concentration				Dose rate				
		^{137}Cs	^{90}Sr	Pu	^{241}Am	^{137}Cs	^{90}Sr	Pu	^{241}Am	TLD
CT1a	Grassy vegetation	√	√			√	√			
CT1b	Lactera agilis	√	√			√	√			
CT1c	Clethrionomys glareolus	√	√							
CT1c	Apodemus flavicollis	√	√							
CT1d	Beetles	√	√			√	√			
CT2a	Grassy vegetation	√	√							
CT2b	Clethrionomys glareolus	√	√							
CT2c	Apodemus flavicollis	√	√							
CT3a	Grassy vegetation	√	√							
CT3b	Sicista betulina	√	√			√	√			
CT3c	Sorex araneus	√	√							
CT4a	Grassy vegetation	√	√							
CT4b	Clethrionomys glareolus	√	√							
CT4c	Apodemus flavicollis	√	√							
CT5a	Rana esculenta	√								
CT5b	Rana terrestris	√								
CT6a	Rana esculenta	√								

BWG Data ID	Species	Biota activity concentration				Dose rate				
		^{137}Cs	^{90}Sr	Pu	^{241}Am	^{137}Cs	^{90}Sr	Pu	^{241}Am	TLD
CT6b	Rana terrestris	√								
CT7	Hirundo rustica	√	√			√	√			
CT8	Perdix perdix	√	√							
CT9	Perdix perdix	√	√			√	√			
CT10	Sturnus vulgaris	√	√			√	√			
CT11	Canis lupus	√	√			√	√			
CT12	Canis lupus	√	√							
CT13	Capreolus capreolus	√	√			√	√			
CT14	Capreolus capreolus	√	√							
CT15	Capreolus capreolus	√	√							
CT16	Capreolus capreolus	√	√							
CT17	Capreolus capreolus	√	√							
CT18	Capreolus capreolus	√	√							
CT19	Capreolus capreolus	√	√							
CT20	Sus scofa	√	√							
CT21	Sus scofa	√	√							
CT22	Sus scofa	√	√							
CT23	Sus scofa	√	√							
CT24	Sus scofa	√	√							
CT25	Sus scofa	√	√							
CT26	Sus scofa	√	√							
CT27	Sus scofa	√	√							
CT28	Sus scofa	√	√			√	√			
CT29a	Clethrionomys glareolus	√	√							
CT29b	Microtus oeconomus	√	√							
CT29c	Sorex araneus	√	√							
CT30a	Microtus arvalis	√	√							
CT30b	Microtus oeconomus	√	√							
CT31a	Apodemus sylvaticus	√	√			√	√			
CT31b	Microtus arvalis	√	√			√	√			
CT32a	Microtus spp.	√	√	√	√	√	√	√	√	√
CT32b	Sorex araneus	√	√	√	√					
CT33a	Clethrionomys glareolus	√	√	√	√	√	√	√	√	√
CT33b	Apodemus flavicollis	√	√	√	√	√	√	√	√	√
CT33c	Sorex araneus	√	√	√	√					
CT34a	Clethrionomys glareolus	√	√	√	√	√	√	√	√	√
CT34b	Apodemus flavicollis	√	√	√	√	√	√	√	√	√
CT34c	Sorex araneus	√	√	√	√	√	√	√	√	
CT35	Parus major	√	√							
CT36a	Parus major	√	√	√		√	√	√		
CT36b	Aegithalos caudatus	√		√				√		
CT37	Erithacus rubecula	√	√			√	√			
CT38	Erithacus rubecula	√	√							
CT39	Clethrionomys glareolus		√	√	√					
CT40	Rana terrestris	√	√			√	√			
CT41	Rana terrestris	√	√							
CT42	Microtus oeconomus	√	√			√	√			√

REFERENCES

[VI.1] GASCHAK, S., CHIZHEVSKY, I., ARKHIPOV, A., BERESFORD, N.A., BARNETT, C.L., The transfer of Cs-137 and Sr-90 to wild animals within the Chernobyl exclusion zone In: International Conference on the Protection of the Environment from the effects of Ionizing Radiation, IAEA-CN-109, Stockholm (2003) pp 200–202.

[VI.2] BERESFORD, N.A., WRIGHT, S.M., BARNETT, C.L., WOOD, M.D., GASCHAK, S., ARKHIPOV, A., SAZYKINA, T.G., AVILA, R., A case study in the Chernobyl zone – Part 1: predicting radionuclide transfer to wildlife, Radioprotection – colloques, 40, (2005) S291–297.

[VI.3] UKRAINIAN RESEARCH INSTITUTE FOR AGRICULTURAL RADIOLOGY, Contamination of the ChNPP 30-km zone, CD v2, UIAR (2001), Chabany (2005).

[VI.4] SMITH, J., BERESFORD, N.A., Chernobyl Catastrophe and Consequences. Chichester: Praxis Publishing/Springer (2005).

PUBLICATION LIST OF THE EMRAS BIOTA WORKING GROUP

BERESFORD, N.A., BALONOV, M., BEAUGELIN-SEILLER, K., BORRETZEN, P., BROWN, J., CHENG, J., COPPLESTONE, D., DOI, M., GASCHAK, S., GOLIKOV, S., HORYNA, J., HOSSEINE, A., HOWARD, B.J., JASSERAND, F., KAMBOJ, S., NEDVECKAITE, T., OLYSLAEGERS, G., SAZYKINA, T., VIVES I BATLLE, J., YANKOVICH, T., YU, C., Models and approaches available to estimate the exposure of non-human biota: An international comparison of predictions, In: The 2nd International Conference on Radioactivity in the Environment (Eds. P. Strand, P. Børretzen and T. Jølle) 2–6 October, Nice, NRPA: Østerås (2005) p 146–149.

BERESFORD, N.A., BARNETT, C.L., BROWN, J., CHENG, J.J., COPPLESTONE, D., FILISTOVIC, V., HOSSEINI, A., HOWARD, B.J., JONES, S.R., KAMBOJ, S., KRYSHEV, A., NEDVECKAITE, T., OLYSLAEGERS, G., SAXEN, R., SAZYKINA, T., VIVES I BATLLE, J., VIVES-LYNCH, S., YANKOVICH, T., YU, C., Inter-comparison of models to estimate radionuclide activity concentrations in non-human biota. In: International conference on environmental radioactivity – from measurements and assessments to regulation, Vienna, 23–27 April 2007, Book of extended synopses, IAEA-CN-145, IAEA, Vienna (2007) 135–136.

VIVES I BATLLE, J., BALONOV, N., BEAUGELIN-SEILLER, K., BERESFORD, N.A., BROWN, J., CHENG, J-J., COPPLESTONE, D., DOI, M., FILISTOVIC, V., GOLIKOV, V., HORYNA, J., HOSSEINI, A., HOWARD, B.J., JONES, S.R., KAMBOJ, S., KRYSHEV, A, NEDVECKAITE, T., OLYSLAEGERS, G., PRÖHL, G., SAZYKINA, T., ULANOVSKY, A., VIVES LYNCH, S., YANKOVICH, T., YU, C., Inter-comparison of unweighteed absorbed dose rates for non-human biota, Radiation and Environmental Biophysics, **46** (2007) 349–373.

BERESFORD, N.A., BALONOV, M., BEAUGELIN-SEILLER, K., BROWN, J., COPPLESTONE, D., HINGSTON, J.L., HORYNA, J., HOSSEINI, A., HOWARD, B.J., KAMBOJ, S., NEDVECKAITE, T., OLYSLAEGERS, G., SAZYKINA, T., VIVES I BATLLE, J., YANKOVICH, T.L., YU. C., An international comparison of models and approaches for the estimation of the radiological exposure of non-human biota, Applied Radiation and Isotopes, 66 (2008). 1745–1749. doi:10.1016/j.apradiso.2008.04.009

BERESFORD, N.A., BARNETT, C.L., BEAUGELIN-SEILLER, K., BROWN, J.E., CHENG, J-J., COPPLESTONE, D., GASCHAK, S., HINGSTON, J.L., HORYNA, J., HOSSEINI, A., HOWARD, B.J., KAMBOJ, S., KRYSHEV A., NEDVECKAITE, T.,OLYSLAEGERS, G., SAZYKINA, T., SMITH, J.T., TELLERIA, D., VIVES I BATLLE, J., YANKOVICH, T.L., HELING, R., WOOD, M.D., YU, C., Findings and recommendations from an international comparison of models and approaches for the estimation of radiological exposure to non-human biota, extended abstract, In: Proceedings International conference on Radioecology and environmental radioactivity, Posters presentations, Part 1, Eds. P. Strand, J. Brown and T. Jølle (2008) pp 234–237.

BERESFORD, N.A., BARNETT, C.L., BROWN, J., CHENG, J-J. COPPLESTONE, D., FILISTOVIC, V., HOSSEINI, A., HOWARD, B.J., JONES, S.R., KAMBOJ, S., KRYSHEV, A., NEDVECKAITE, T., OLYSLAEGERS, G., SAXÉN, R., SAZYKINA, T., VIVES I BATLLE, J., VIVES-LYNCH, S., YANKOVICH, T., YU, C., Inter-comparison of models to estimate radionuclide activity concentrations in non-human biota, Radiation and Environmental Biophysics, 47 (2008) 419–514 doi:10.1007/s00411-008-0186-8.

BERESFORD, N.A., BARNETT, C.L., BEAUGELIN-SEILLER, K., BROWN, J.E., CHENG, J-J., COPPLESTONE, D., GASCHAK, S., HINGSTON, J.L., HORYNA, J., HOSSEINI, A., HOWARD, B.J., KAMBOJ, S., KRYSHEV A., NEDVECKAITE, T.,OLYSLAEGERS, G., SAZYKINA, T., SMITH, J.T., TELLERIA, D., VIVES I BATLLE, J., YANKOVICH, T.L., HELING, R., WOOD, M.D., YU, C., Findings and recommendations from an international comparison of models and approaches for the estimation of radiological exposure to non-human biota, Radioprotection (In Press).

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