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Application of ERICA Integrated Approach at case study sites

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ERICA (Environmental Risk from Ionising Contaminants: Assessment and Management) will provide an integrated approach to scientific, managerial and societal issues concerned with the environmental effects of contaminants emitting ionising radiation, with emphasis on biota and ecosystems. The project started in March 2004 and is to end by February 2007.

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Executive Summary

The application of (draft) outputs of the ERICA project to five different case study sites is described. The case study sites were:

- *Drigg Coast Sand Dunes, UK* (terrestrial ecosystem) – Coastal area protected under international and national environmental legislation. Contaminated indirectly by discharges from the Sellafield nuclear reprocessing plant.
- *Loire River, France* (freshwater ecosystem) – Receives discharges from a number of regulated sites.
- *Sellafield, UK* (marine ecosystem) – Regulated site discharging comparatively high levels of radioactivity into the Irish Sea.
- *Komi Republic, Russia* (terrestrial ecosystems) – Technologically enhanced natural radionuclides, radiation induced effects reported in biota.
- *Chernobyl exclusion zone, Ukraine* (terrestrial ecosystems) – Highly contaminated site with good datasets available for activity concentrations in biota, study conducted by ERICA participants to provide dose rate estimates for small mammal species.

The objectives of the case study applications were:

- to assess the applicability of December 2006 prototype software (the ERICA Tool) and draft of the final deliverable (D-ERICA);
- to compare predicted and observed activity concentrations in biota (and water/sediments for aquatic ecosystems);
- where possible, to compare measured doses and observed radiation induced effects with estimated doses and predicted effects;
- to make recommendations of improvements needed to the ERICA final outputs.

An important point to note before reading the case study reports is that they represent the authors' views on the performance of the prototype Tool and understanding of the accompanying draft documentation as issued for case study application (December 2006). Many of the issues raised have been addressed in the final release versions of D-ERICA and the ERICA Tool.

The case studies enabled different aspects of the ERICA Integrated Approach and associated ERICA Tool to be assessed. For instance:

- both the Sellafield Marine and Drigg Coast Dunes case studies facilitated a full application of the Integrated Approach at sites receiving discharges from a nuclear licensed site (participants playing the role of assessors);
- the Chernobyl case study provided data with which to compare external dose rate predictions;
- the Komi case study concentrated on natural radionuclides and included effects data with which to compare to the outputs of the effects database (FREDERICA) compiled during the ERICA project;
- the Loire River and Sellafield Marine case studies included comparisons of predictions from the screening transport models included within the Tool with bespoke models parameterised for the study sites.

Much of the feedback to the ERICA Consortium from the case study applications identified parts of the Integrated Approach which were poorly explained. As a consequence, both D-ERICA and the Tool Help file have been extensively rewritten.

Generally, there was an acceptable level of agreement between predicted and observed activity concentrations within biota. Where this was not the case, reasons could often be put forward to explain this. Similarly, predicted external dose rates compared well with measured dose rates. Areas of the ERICA Integrated Approach which require further testing were highlighted (e.g. degree of conservatism assumed at Tier 2). Responses of the ERICA Consortium to comments on the Integrated Approach resulting from the case study applications are given.

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1 Introduction

This deliverable describes the application of the ERICA Integrated Approach to five different case study sites. The case study sites were selected to test various components of the ERICA approach. The case study sites were:

Drigg Coast Sand Dunes, UK (terrestrial ecosystem) – Coastal area protected under international and national environmental legislation. Contaminated indirectly by discharges from the Sellafield nuclear reprocessing plant.

Loire River, France (freshwater ecosystem) – Receives discharges from a number of regulated sites.

Sellafield, UK (marine ecosystem) – Regulated site discharging comparatively high levels of radioactivity into the Irish Sea.

Komi Republic, Russia (terrestrial ecosystems) – Technologically enhanced natural radionuclides, radiation induced effects reported in biota.

Chernobyl exclusion zone, Ukraine (terrestrial ecosystems) – Highly contaminated site with comparatively good datasets available for activity concentrations in biota, study conducted by ERICA participants to provided dose rate estimates for small mammal species.

The objectives of the case study applications were:

- to assess the applicability of prototypes and drafts of ERICA outputs, namely the projects final deliverable (D-ERICA, Larsson et al., 2006) and software (the ERICA Tool);
- to compare predicted and observed activity concentrations in biota (and water/sediments for aquatic ecosystems);
- where possible, to compare observed radiation induced effects with estimated doses and predicted effects;
- to make recommendations to the ERICA consortium with regard to the projects' final outputs.

The case study applications were conducted concurrent to ongoing development of the ERICA outputs and their application by the ERICA end user group (see Zinger, 2006). As a consequence of interaction with the end user group, an early decision to radically revise the initial draft of D-ERICA was taken (Zinger, 2006). Subsequently, the emphasis of the case studies was placed on application and testing of the ERICA Tool and its underlying databases.

1.1 Deliverable description

Following an overview within this chapter of the ERICA Integrated Approach and the participation of the ERICA project in an international model comparison exercise, the application of the ERICA Tool to each case study is described in the subsequent five chapters. Observations on the performance of the ERICA Tool and recommendations arising from each application are reported within the individual case study reports. The final chapter summarises the findings and recommendations on the basis of all of the case studies. It also provides information on how the ERICA consortium will, or in some cases has already, responded to some recommendations arising from the case studies. Prior to the release of the case study test version of the ERICA Tool, participants trialled a number of pre-release versions to identify any functionality issues; some of the issues raised at this stage are retained within the recommendations within Annex A of this deliverable.

Whilst the editors have exercised some quality control on each of the case study chapters these remain the works, and views, of the chapter authors. **An important point to note before reading the case study report is that they represent the authors views on the performance of the Tool and understanding of the accompanying documentation as issued for case study application (December 2006). Many of the issues raised have been addressed in the final release versions of D-ERICA and the Tool (see Chapter 7 and Appendix A).**

1.2 Overview of the ERICA Integrated Approach

The ERICA Integrated Approach advises the user on problem formulation (involving stakeholders if appropriate), performing an impact assessment and evaluating data. It outlines the issues and options available to (and requiring decisions to be made by) the user before, during, and after an assessment. The following overview of the ERICA Integrated Approach and Tool is adapted from the final version of D-ERICA (Beresford et al. 2007) and not that available within the documentation supplied for case study application. The overview presented in this section in not comprehensive but should give readers an introduction to issues discussed within the subsequent case study reports (D-ERICA should be consulted for more detail).

The ERICA Integrated Approach is supported by the ERICA Tool, which is a software programme that guides the user through the assessment process, keeps records and performs the necessary calculations to estimate dose rates to selected biota. A detailed Help is provided to assist the user in making appropriate choices and inputs, as well as interpreting the outputs. The Tool interacts with a number of databases and other functions that help the assessor to estimate environmental media activity concentrations, activity concentrations in biota, and dose rates to biota. The ERICA Tool also interfaces with the FREDERICA radiation effects database, which is a compilation of the scientific literature on radiation effect experiments and field studies, organised around different wildlife groups and, for most data, broadly categorised according to four effect umbrella endpoints: morbidity, mortality, reproduction and mutation.

The databases of the ERICA Tool have been built around a number of reference organisms. Each reference organism has its own specified geometry (and default transfer data) and is representative of either terrestrial, freshwater or marine ecosystems. The approach is compatible with that used by ICRP; some of the geometries proposed for the ICRP 'reference animals and plants' are used as defaults in the ERICA Tool.

The assessment element of the ERICA Integrated Approach is organised in three separate tiers, where satisfying certain criteria in Tiers 1 and 2 allows the user to exit the assessment process while being confident that the effects on biota are low or negligible, and that the situation requires no further action. Where the effects are not shown to be negligible, the assessment should continue to Tiers 2 and 3. Situations of concern should be assessed further in Tier 3, by making full use of all relevant information available through the Integrated Approach or elsewhere.

1.2.1 Tier 1 assessment

The Tier 1 assessment is designed to be simple and conservative, requiring a minimum of input data and enabling the user to exit the process and exempt the situation from further evaluation, provided the assessment meets a predefined screening criterion. The default screening criterion in the ERICA Integrated Approach is an incremental dose rate of 10 μGy h^{-1}, to be used for all ecosystems and organisms. This value was derived from a species sensitivity distribution analysis performed on chronic exposure data in the FREDERICA database and is supported by other methods for determining predicted no effect values. However, the user can change the default screening dose rate within the ERICA Tool, For Tier 1, the predefined screening dose rate is back-calculated to yield Environmental Media Concentration Limits (EMCLs) for all reference organism/radionuclide combinations. The Tool derives a risk quotient (RQ) by dividing the most restrictive EMCL for each radionuclide by the input media concentrations, the radionuclide specific risk quotients are then summed to determine a total RQ. If the RQ is less than one, then the Tool suggests that the user should exit the assessment process. If the RQ is greater than one, the user is advised to continue with the assessment. It is envisaged that most situations will be screened out at Tier 1.

1.2.2 Tier 2 assessment

Tier 2 allows the user to be more interactive, to change the default parameters and to select specific reference organisms. The evaluation is performed directly against the screening dose rate, with the dose rate and RQs generated for each reference organism selected for assessment. A 'traffic light' system is used to indicate whether the situation can be considered to be:

I) Green: of negligible concern (with a high degree of confidence);

II) Amber: of potential concern, where more qualified judgements may need to be made and/or a refined assessment at Tier 2 or an in-depth assessment in Tier 3 performed;

III) Red: of concern, where the user is recommended to continue the assessment either at Tier 2 if refined input data can be obtained or at Tier 3.

Decisions to exit an assessment given outcomes (II) and (III) should be justified, for example by using information from FREDERICA provided in the Tool as 'look-up effects tables' for different wildlife groups.

1.2.3 Tier 3 assessment

Situations, which give rise to a Tier 3 assessment, are likely to be complex and unique, and it is therefore not possible to provide detailed or specific guidance on how the Tier 3 assessment should be conducted. Furthermore, a Tier 3 assessment does not provide a simple yes/no answer, nor is the ERICA-derived incremental screening dose rate of 10 µGy h^{-1} appropriate with respect to the assessment endpoint. The requirement to consider aspects such as the biological effects data within the FREDERICA database, or to undertake ecological survey work, is not straightforward and requires an experienced, knowledgeable assessor or consultation with an appropriate expert.

Tier 3 is a probabilistic risk assessment in which uncertainties within the results may be determined using sensitivity analysis. The assessor can also access up-to-date scientific literature (which may not be available at Tier 2) on the biological effects of exposure to ionising radiation in a number of different species. Together, these allow the user to estimate the probability (or incidence) and magnitude (or severity) of the environmental effects likely to occur and, by discussion and agreement with stakeholders, to determine the acceptability of the risk to non-human species.

1.2.4 Estimation of exposure

Exposure to radiation is estimated as the absorbed dose rate (the quantity of energy imparted by ionising radiation to a unit mass of an organism per unit time, with µGy h^{-1} used within the ERICA Tool). To determine this, the activity concentrations in both media and biota are required, together with the ability to convert these into estimates of external and internal exposure. Radionuclide activity concentration in media and/or biota may be known, or they may need to be estimated.

If media concentrations are not known, and the user does not have an appropriate model, the Tool includes a number of screening transport models adopted from IAEA (2001) (referred to in the following as the 'SRS-19 models').

If data are not available, whole-body activity concentrations of radionuclides in biota within the ERICA Tool are predicted from media activity concentrations using equilibrium concentration ratios (CRs). For aquatic environments, the distribution coefficient (K_d) is used to relate equilibrium activity concentrations in sediments with those in water. Concentration ratios and K_d for the ERICA ecosystems are defined in Box 1.1.

Box 1.1. Definitions of concentration ratios and K_d for the ERICA ecosystems.

Definition of CR and K_d

Terrestrial ecosystems

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

Exceptions are for chronic atmospheric releases of ^{3}H, ^{14}C, $^{32,33}P$ and ^{35}S where:

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

Aquatic ecosystems

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

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

The relationship between the activity concentration in an organism, or medium, and internal or external absorbed dose rates is described by the dose conversion coefficient (DCC; µGy h^{-1} per Bq kg^{-1} fresh weight). The method used to derive DCC values within the ERICA Tool are based on the methods described by Pröhl et al. (2003) and Ulanovsky and Pröhl, (2006). In addition to databases within the Tool containing DCC vaues for all reference organisms, the Tool allows the user to define their own organisms within certain size limitations.

1.3 ERICA participation in the IAEA EMRAS Biota working Group

The IAEA formed the *Biota Working Group* (BWG) as part of the EMRAS (Environmental Modelling for Radiation Safety) programme in November 2004. The primary objective of the BWG is: '*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'* (Beresford et al., 2005). This is being achieved through model intercomparison exercises and scenario testing at sites with available data. The ERICA project has participated within the BWG applying underlying databases from the ERICA Tool. The other models and approaches participating within the BWG included those being developed and, in some cases applied, in a regulatory context in the USA, Canada, France, Belgium, Russia, Lithuania and England and Wales. To date, the BWG has conducted two model-model comparisons and two scenario applications are currently underway.

The first model-model intercomparison exercise to compare unweighted absorbed whole organism dose conversion coefficients (DCCs) for both internal and external exposure for a selection of the Reference Animal and Plant geometries proposed by the ICRP (ICRP, 2005) has now been completed (Vives i Batlle et al., submitted). Overall, the exercise demonstrated that all 11 participating approaches compared favourably despite a wide range of assumptions being made. Of note to ERICA were:

I) External DCCs for ^{90}Sr as used within the ERICA Tool for terrestrial organisms were low (by 4-7 orders of magnitude) compared to other participating models and approaches. This was thought to be the consequence of the ERICA methodology assuming a shielding skin/fur layer which was not considered within the other approaches. This should perhaps be highlighted in the ERICA documentation as it will make considerable differences to external dose estimates compared to other models (see also section 6.4).

II) It was recommended that there was little justification for considering whole-body averaged external doses for low energy ß-emitters (such as ^{3}H and ^{14}C). The ERICA approaches was amongst those which already assume doses from such low energy ß-emitters to be zero.

The second model-model intercomparison compared predicted whole-body activity concentrations for selected radionuclides in a range of freshwater and terrestrial biota assuming 1 Bq per unit (kg, l or m^3) of media (soil, water and air respectively). Whilst this exercise is yet to be completed, participation led to changes in a number of ERICA concentration ratios (CRs) for mammals. Initial values included many data for reindeer, which, for some radionuclides, resulted in values which were unrealistically high for most mammals. Initial results also suggest that the guidance methodology used to derive default CR values within ERICA when empirical data are lacking provides conservative estimates (as intended) but less extreme than the initial methodology (Copplestone et al., 2003) from which it was developed. The exercise has also shown much greater variation in the estimation of whole-body activity concentrations by the participating approaches than for DCCs, demonstrating a need for transparency in methodology and data provenance within documentation of all methods with respect to deriving CR (and K_d) values.

Two scenarios to test model predictions against available data are underway (see http://www-ns.iaea.org/projects/emras/emras-biota-wg.htm for details). The first is a freshwater application for Perch Lake (Canada) for which ^{90}Sr, ^{137}Cs, ^{3}H and ^{60}Co activity concentration measurements are available for water, sediments and a wide range of biota types (including plants, fish, various invertebrates, mammals and reptiles). This has not progressed to such a stage where we can comment on the performance of the ERICA approach compared to other models. However, participation within the scenario demonstrated a need to be able to enter soil and sediment dry matter fractions within the ERICA Tool (which is now included). The second scenario is based on data from terrestrial ecosystems within the Chernobyl exclusion zone (see Chapter 6).

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2 Drigg Coast Sand Dunes, UK

(*W.A. Marshall, S. Jones, S. Watts (Westlakes Scientific Consulting) and M.D. Wood (University of Liverpool)*)

2.1 Introduction

An assessment was conducted in 2005 to evaluate the application of the FASSET methodology in assessing the environmental risks from ionising radiation on both agricultural and semi-natural ecosystems in the vicinity of the British Nuclear Group Sellafield Limited (BNGSL) Sellafield site (Beresford and Howard, 2005; Johnson and Marshall, 2005). The semi-natural ecosystem was represented by the sand dune and saltmarsh areas of the Drigg Coast Special Area of Conservation (SAC), a Natura 2000 designated conservation site. This assessment, together with associated stakeholder input and feedback, highlighted a number of deficiencies in the FASSET methodology. These included the absence of an amphibian in the FASSET list of terrestrial reference organisms, a lack of data to permit the assessment of internal dose to plants and an absence of concentration ratio data for birds. One major comment of stakeholders was the inability to assess impact on important protected species of amphibian such as the Natterjack Toad (*Bufo calamita*) (at this site).

An additional problem highlighted by the previous study was that, given considerable gaps in data, doubt remained as to the validity of the prediction of no effects on organisms of the Drigg Coast SAC from anthropogenic radionuclides.

The current report details the results of a second assessment of the Drigg Coast SAC, and the sand dune ecosystem in particular. This study utilises the ERICA Tool developed during the ERICA project and an up-to-date set of site specific data for radionuclide activity concentrations in soil and organisms.

2.2 Collection of site specific soil and biota data

Sampling was undertaken at the Drigg Coast Sand Dunes in 2005 and 2006. The sampling programme was designed to provide data necessary for the testing and validation of the ERICA Tool and, subsequently, other tools and approaches that have been developed to assess the impact of ionising radiation on wildlife. Samples collected included environmental media and various biota (Table 2.1).

Due to the protected nature of the Drigg Coast Sand Dune habitat and many of the biota present, approvals, consents and licences were obtained from the relevant people and organisations prior to commencing work at the dunes. These included a protected species licence from English Nature for the sampling of reptiles and the two protected amphibian species, the Great Crested Newt (*Triturus cristatus*) and the Natterjack Toad (*B. calamita*).

The sampling methodologies employed at Drigg Coast Sand Dunes are described in the following sections.

Table 2.1 Samples collected from the Drigg Coast Sand Dunes during 2005 and 2006.

Sample / Organism Group	Common name	Latin name
Amphibian	Common Toad	*Bufo bufo*
	Common Frog	*Rana temporaria*
	Great Crested Newt	*Triturus cristatus*
	Palmate Newt	*Triturus helveticus*
	Natterjack Toad	*Bufo calamita*
Bird	Mallard	*Anas platyrhynchos*
	Teal	*Anas crecca*
Invertebrate	Caterpillar	Non-specific
Mammal	Wood Mouse	*Apodemus sylvaticus*
	Field Vole	*Microtus agrestis*
Reptile	Common Lizard	*Lacerta vivipara*
	Adder	*Vipera berus*
	Slow Worm	*Anguis fragilis*
Plants	Marram Grass	*Ammophila arenaria*
	Red Fescue	*Festuca rubra*
	Heather	*Calluna vulgaris*
	Moss	Non-specific
	Lichen	Non-specific
Soil/Sediment	Soil/sediment from fore-dunes	
	Soil/sediment from rear of the dunes	

2.2.1 Environmental Media

Soil/sediment sampling

Three sampling transects were established at the Drigg Coast Sand Dunes to provide a good spatial coverage of the entire dune system (Figure 2.1) and to account for sea-to-land transfer, which is likely to be the principal route by which many anthropogenic radionuclides, especially actinides and other particle reactive radionuclides, reach the dunes. The transects were orientated in the prevailing wind direction and were located to maximise the number of dune habitat types that were covered by each transect.

Soil samples were collected from the front and back of each transect to a depth of 10 cm. A stainless-steel split-blade corer of 10cm diameter was used to extract the soil samples as described in Wood et al. (in press). The soil surface was cleared of vegetation and stones. Any surface litter was removed by gently scraping it away with a spatula. The corer was hammered into the soil to a 40 cm depth using a soft-headed mallet. A purpose designed A-frame and lifting tackle were used to extract the core. The corer was positioned on the ground and the upper blade retracted to expose the core (Figure 2.2). The core was cut at 10 cm below the upper surface using a stainless-steel knife and the bottom 30 cm section was returned to the hole from which the core was extracted. The upper 10 cm section was transferred to a labelled plastic sample bag and retained for analysis. Samples were packed into a cool box for transport to the analytical laboratory.

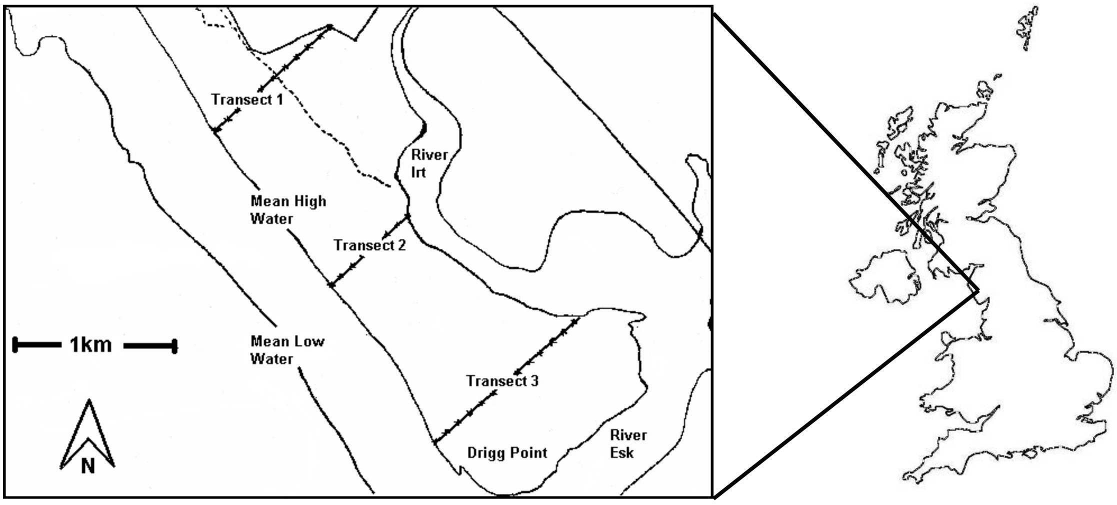

Figure 2.1. Location of the sampling transects at the Drigg Coast Sand Dunes.

Figure 2.2 Soil core (40cm) extracted from the rear of the sand dunes at the Drigg Coast Sand Dune.

2.2.2 Biota

Biota sampling was conducted in accordance with the requirements of national legislation, namely the Wildlife and Countryside Act 1981 (amended by the Environmental Protection Act 1980), the Conservation (Natural Habitats, & C.) Regulations1994 and the Animals (Scientific Procedures) Act 1986.

Amphibian sampling

Amphibian sampling was conducted using the methodologies described in Gent and Gibson (1998). The two methodologies that proved effective at Drigg were bottle trapping and torching (with hand capture).

Between 25-30 bottle traps were positioned at approximately 2m intervals around the perimeter of a pool. Each bottle trap was slowly submerged until two thirds of the trap was filled with water. The trap was then positioned at a 45 degree angle so that the mouth of the trap was lowest and there was an air pocket in the end of the trap. The air pocket ensured that trapped amphibians would have a sufficient oxygen supply until the bottle traps were retrieved. The traps were secured in this position using a garden cane and a length of elastic. The traps were deployed during late afternoon/early evening and were retrieved the morning after deployment and the animals collected.

The bottle traps proved effective at trapping newts of all species, namely the Great Crested Newt (*Triturus cristatus*), the Palmate Newt (*Triturus helveticus*) and the Smooth Newt (*Triturus vulgaris*). They also caught some Common Frogs (*Rana temporaria*) and a few Common Toads (*Bufo bufo*).

Torching was useful for locating both toad species (Common Toad (*B. bufo*) and the Natterjack Toad (*B. calamita*)). Amphibians were identified in pools after dark using torches and individuals selected and captured by hand.

Bird sampling

The British Association for Shooting and Conservation were contacted to identify members that shoot in the vicinity of Drigg. Whilst there is no shooting permitted on the dunes themselves, the Egremont and District Wildfowlers Association were identified as having rights to shoot on the opposite side of the River Irt. This was the closest location to the dunes where shooting occurs and some species of wildfowl, especially resident Mallards (*Anas platyrhynchos*), were thought likely to feed on the dune heath. The Wildfowlers Association donated one Mallard (*A. platyrhynchos*) and two Teal (*Anas crecca*).

Invertebrate sampling

Invertebrate sampling was opportunistic. Many caterpillars were found in the vicinity of some of the reptile sampling locations (see Section 2.2.5) so samples of these animals were collected by hand. The animals were killed by immersion in 70% ethanol. They were kept in the ethanol to preserve them during transport to the analytical laboratory.

Mammal sampling

Small mammal sampling was undertaken using pre-fabricated live traps. Three types of trap were used: Longworth traps, Pipe traps and Trip traps. Traps were set out in a grid pattern, with approximately 10 m between traps, in an area that was determined to be likely to support small mammal populations. In total, five voles (*Microtus agrestis*) and two mice (*Apodemus sylvaticus*) were obtained.

Reptile sampling

Reptile sampling was conducted using the methodologies described in Reading (1996) and Gent and Gibson (1998). The two approaches that were used were hand capture and the use of artificial refuges.

Artificial refuges comprised sheets of corrugated iron cut into squares measuring 0.81 x 0.75 m. Reptiles are poikilotherms and are thus attracted to the refuges as they warm during daylight hours. Temperature measurements made during refuge searches at Drigg showed that the temperature difference between the ground surface immediately next to the refuge and the area underneath the refuge can exceed 16^{o}C.

Three locations, each associated with one of the three main transects, were identified for the establishment of artificial refuge arrays. Each array consisted of 19 individual refuges which were set out in a hexagonal pattern with an inter-refuge distance of 15 m.

The arrays were visited on 10 occasions during 2006. Each array was walked systematically by two field workers. One worker lifted the refuges while the other worker captured reptiles found underneath. All three arrays proved successful at attracting common lizards (*Lacerta vivipara*), which are abundant at Drigg. Slow worms (*Anguis fragilis*) were also found under the refuges but appeared less abundant than *L. vivipara*. None of the refuges were utilised by adders (*Vipera berus*) at any point during the study. For this reason, additional hand capture techniques had to be employed. Collection of *V. berus* was conducted by locating areas of snake activity, and targeting these areas for intensive searches at times when the snakes were most likely to be basking. Two male adders were collected for analysis in this way.

Vegetation sampling

Where possible, vegetation samples were collected on a species-specific basis from each transect. The methodology used for the collection of the vegetation samples was that described in Wood et al. (in press). The vegetation was clipped to 2-3 cm above the soil surface using garden shears. The cut vegetation was transferred to a labelled plastic sample bag. Once approximately 400 g (fresh weight) of material had been collected, the sample bag was sealed and a record made of the area of vegetation cover that had been cut. The labelled plastic bags were packed into a cool box for transport to the analytical laboratory.

2.3 Assessment of dose to biota using the ERICA methodology

2.3.1 Input data

Soil radionuclide activity concentrations

Soil and biota samples were analysed to determine gamma-emitting radionuclides; animal samples were analysed as whole-body minus the gastrointestinal tract and feather or fur as applicable. Results for anthropogenic gamma-emitting radionuclides which could be expected to be found as a consequence of Sellafield marine discharges and a range of naturally occurring radionuclides (soils only) were provided for the case study assessment (see Tables 2.3 and 2.5). Selected soil and biota samples were also analysed to determine activity concentrations of ^{239,240}Pu, ^{99}Tc and ^{90}Sr (see Tables 2.3 and 2.5).

To provide an estimate of background radiation levels for naturally occurring radionuclides, it was assumed that radionuclides of the ^{238}U and ^{232}Th series were in equilibrium in the soil. Data for ^{226}Ra, ^{214}Pb and ^{214}Bi were available for radionuclides in the ^{238}U decay chain and data for ^{228}Ac, ^{212}Bi and ^{212}Pb were available for radionuclides in the ^{232}Th decay chain (Table 2.2). These data were used to provide input data for the other natural radionuclides in these decay chains for which dose conversion coefficients were provided within the ERICA Tool (Table 2.2). The values cited in Table 2.2 are low for the range of activity concentrations of these radionuclides found in UK soils, but this is not surprising given the sandy nature of the soils at Drigg.

An initial assessment was conducted using Tier 1 in the ERICA Tool. At the Tier 1 stage for terrestrial ecosystems, data is required for soil activity concentrations in dry weight (DW) and the maximum recorded measurement from all soil samples taken was used for each anthropogenic radionuclide (Table 2.3). Where values were recorded as being below the limit of detection, the "less than" value was used to ensure a conservative Tier 1 assessment.

Following the Tier 1 assessment, two Tier 2 assessments were conducted, both utilised the mean soil activity concentrations from all soil samples for each anthropogenic radionuclide. In the first assessment all data were used including that below the limit of detection (Table 2.3), this also applied to biota activity concentrations. In the second Tier 2 assessment only data recorded as being at or above the limit of detection were used (Table 2.3). In both assessments activity concentrations of naturally occurring radionuclides remained at the background levels defined in Table 2.2. A problem arose in the second Tier 2 assessment in that good organism data were available for ^{134}Cs and ^{106}Ru in moss and lichen, but not in soil. The ERICA Tool would not allow organism data to be entered for radionuclides where no soil data had been entered. To continue the assessment, zeros had to be entered for the soil activity concentrations. Subsequent versions of the ERICA Tool allow the assessment to progress if soil activity concentrations are missing as long as a value is entered for at least one reference organism for a given radionuclide. Soil activity concentrations are then back calculated from CR value to enable total dose estimates. However, for the purposes of this assessment, in the case of lichens and moss the external dose rate is negligible (assessed by looking at the Tool DCC values[1]) and the omission of soil data has no effect on the total dose rate received by these organisms.

Table 2.2 Natural radionuclide concentrations observed in soil at the Drigg Coast Sand Dunes and data used as input for the ERICA assessment (where 'input' values were estimated by assuming radioactive equilibrium).

	Decay Chain				
	^{238}U Soil activity concentration (Bq kg^{-1} DW)			^{232}Th Soil activity concentration (Bq kg^{-1} DW)	
Nuclide	**Observed**	**Input**	**Nuclide**	**Observed**	**Input**
^{238}U		12	^{232}Th		12
^{234}Th		12	^{228}Ra		12
^{234}U		12	^{228}Ac	9.3	
^{230}Th		12	^{228}Th		12
^{226}Ra	11.7	12	^{212}Bi	12	
^{214}Pb	15.8		^{212}Pb	15.9	
^{214}Bi	12.1				
^{210}Pb		12			
^{210}Po		12			

[1] Note the lichen & bryophyte DCC in the release version of the Tool values were subsequently identified to be in error and have subsequently revised (increased; see Annex A) – this may impact on the conclusion.

Table 2.3 Soil activity concentrations used in the assessment of dose to biota on the Drigg Coast Sand Dunes using the ERICA methodology at Tier 1 and Tiers 2 and 3.

	Soil activity concentration (Bq kg^{-1} DW)		
	Tier 1	Tier 2 and 3	
Nuclide	Max. all samples	Mean all samples	Mean values, only samples ≥ LOD*
^{241}Am	100	49	49
^{144}Ce	2.7	2.5	
^{60}Co	0.69	0.41	
^{134}Cs	0.33	0.30	0
^{137}Cs	180	140	140
^{152}Eu	0.02	0.02	
^{154}Eu	0.91	0.72	
^{95}Nb	1.0	0.92	
^{238}Pu	11	5.2	
^{240}Pu	64	29	29
^{103}Ru	1.0	0.83	
^{106}Ru	2.7	2.5	0
^{125}Sb	1.4	1.1	
^{90}Sr	18	14	14
^{99}Tc	38	20	20
^{95}Zr	0.90	0.84	

*Where no value reported for mean >LOD then all samples had activity concentrations <LOD

Organisms of interest

The organisms of interest for this assessment were largely determined by the requirements of stakeholders following the FASSET assessment (Beresford and Howard, 2005; Johnson and Marshall, 2005). The main requirement was for the inclusion of amphibians but reptiles, which had not been included in the previous assessment were also included along with species that were previously assessed such as small mammals, vegetation and birds. The sampling programme itself was constrained by time and limits on the number of organisms that could be taken and the analysis programme by cost, therefore these factors also determined which organisms were included in the assessment. The aim was to limit the assessment to contemporary measurements thereby focusing it on current impacts, whereas the FASSET assessment was conducted with data spanning a number of decades. Organisms for which new data were available are listed in Table 2.1.

To best represent the organisms of interest in terms of size and ecology, default reference organism settings in the ERICA Tool were not appropriate for all organisms included in the assessment. Data were gathered on the dimensions and ecology of organisms from internet searches and compared to that of the reference organisms. In instances where there was a considerable disparity between the dimensions of a reference organism and the dimensions of its corresponding organism of interest, a new reference organism was created using the add organism function. Dimensions of new/custom organisms are given in Table 2.4 along with occupancy values. Where occupancy values sum to less than one, as is the case for the birds, this indicates that the organism spends time outside of the assessed area.

Table 2.4 Size and occupancy data used to represent organisms of interest from the Drigg Coast SAC.

Organism	Occupancy (fraction of time) In soil	At soil surface	In air	Dimensions (m) x	y	z	Mass (kg)
Heather	0	1	0	ERICA default shrub			
Lichen	0	1	0	ERICA default lichen bryophyte			
Marram grass	0	1	0	ERICA default grass/herb			
Mixed vegetation	0	1	0	ERICA default grass/herb			
Moss	0	1	0	ERICA default lichen bryophyte			
Red Fescue	0	1	0	ERICA default grass/herb			
Common Toad	0.25	0.75	0	ERICA default amphibian			
Common Frog	0.25	0.75	0	ERICA default amphibian			
Great Crested Newt	0	1	0	0.16	0.015	0.013	0.0085
Natterjack Toad	0.6	0.4	0	0.06	0.03	0.04	0.02
Palmate Newt	0	1	0	0.09	0.012	0.0135	0.004238
Mallard	0	0.3	0.25	ERICA default bird			
Teal	0	0.25	0.25	ERICA default bird			
Caterpillar	0	1	0	ERICA default flying insect			
Mouse	0.5	0.5	0	0.081	0.03	0.03	0.02
Vole	0.2	0.8	0	0.09	0.035	0.035	0.03
Common Lizard	0.4	0.6	0	0.14	0.01	0.02	0.01
Adder	0.75	0.25	0	0.65	0.02	0.02	0.0725
Slow worm	0.4	0.6	0	0.4	0.02	0.015	0.034

Radioecology parameters or concentration ratios were set at the default values for all the default reference organisms. Where new reference organisms were created, the default concentration ratio values for the corresponding organism group were used in all cases.

Organism radionuclide activity concentrations

In the Tier 2 assessment there is the opportunity to input organism activity concentrations where these are available. In the first Tier 2 assessment the mean of all available measurements for each organism/radionuclide combination was used. This included data that were indicated as below the limit of detection; limit of detection data were entered as the given value. This introduced a level of conservatism to this assessment. In the second Tier 2 assessment this level of conservatism was removed and only data at or above the limit of detection were used. Where more than one value was available for each radionuclide/organism combination the mean of the values was used. Organism activity concentrations used in the first Tier 2 assessment are given in Table 2.5, and those for the second in Table 2.6 for animals and Table 2.7 for plants.

Predicting organism activity concentrations

Using the ERICA Tool it is possible to derive predictions of radionuclide activity concentrations found within organisms based on observed soil activity concentrations. This exercise was carried out for organisms/nuclide combinations for which observed data were available at or above the limits of detection (Tables 2.6 and 2.7). Input data in these cases consisted of soil activity concentration and organism occupancy and morphology data only. Both Tier 2 and Tier 3 were used to obtain predictions. In Tier 3, the probabilistic assessment, predictions were based on the assumption of a log normal distribution of soil activity concentrations.

Table 2.5 Mean anthropogenic radionuclide activity concentrations in organisms of interest at the Drigg Coast SAC, including data below the limit of detection (identified by *). For animals the values represent whole-body activity concentrations.

Nuclide	Mean Activity Concentration (Bq kg^{-1} FW)								
	Marram Grass	Red Fescue	Moss	Lichen	Heather	Mouse	Vole	Teal	Mallard
^{110m}Ag		1.2*				4.3*	8.0*	0.20*	0.15*
^{241}Am	9.3	3.7	5.9	5.2	1.9	6.1*	10*	0.96	0.32
^{144}Ce	1.6*	3.3*	5.2*	0.44*	1.7*	28*	51*	0.83*	0.56*
^{60}Co	0.29*	0.75*	0.65*	0.10*	0.28*	3.4*	7.8*	0.20*	0.17*
^{134}Cs	0.29*	0.66*	1.1*	0.17*	0.31*	3.1*	5.7*	0.17*	0.13*
^{137}Cs	3.8	3.9	21	13	25	3.8	8.2	2.1	3.2
^{152}Eu	0.72*	0.85*	1.5*	3.0*	0.91*				
^{154}Eu	0.37*	0.29*	0.57*	0.11*	0.43*				
^{95}Nb	0.47*	4.0*	1.5*	0.10*	0.33*	84*	200*	0.99*	0.70*
$^{239+240}$Pu		1.3						0.42	0.18
^{103}Ru	0.57*	0.27*	1.1*	0.05*	0.36*				
^{106}Ru	4.4*	7.2*	25*	1.4*	1.4*	33*	62*	1.7*	1.3*
^{125}Sb	1.1*	1.7*	2.4*	0.24*	0.91*	8.3*	15*	0.45*	0.02*
^{90}Sr		4.3						0.58*	0.75*
^{99}Tc		1.4						2.7	0.52*
^{95}Zr	0.56*	3.0*	2.0*	0.16*	0.55*	31*	64*	0.76*	0.57*
Nuclide	**Caterpillar**	**Common Toad**	**Common Frog**	**Palmate Newt**	**Great Crested Newt**	**Natterjack Toad**	**Adder**	**Common Lizard**	**Slow worm**
^{110m}Ag	2.3*	2.0*	1.1*	2.0*	2.6*	1.7*	1.5*	2.8*	1.6*
^{241}Am	3.5*	2.8*	2.0*	3.3*	4.5*	3.0*	1.6*	4.3*	2.5*
^{144}Ce	14*	13*	6.9*	12*	16*	10*	8.9*	17*	9.4*
^{60}Co	2.9*	1.5*	1.6*	2.3*	3.4*	2.1*	2.0*	3.8*	1.7*
^{134}Cs	1.9*	1.4*	0.90*	1.7*	2.2*	1.4*	1.0*	2.2*	1.3*
^{137}Cs	2.8	2.3	2.1	2.1	12	7.9*	1.4*	7.3	17
^{152}Eu									
^{154}Eu									
^{95}Nb	11*	53*	4.4*	8.6*	12*	7.1*	40*	12*	6.4*
^{238}Pu									
^{240}Pu									
^{103}Ru									
^{106}Ru	20*	15*	9.3*	17*	24*	14*	12*	24*	13*
^{125}Sb	5.4*	3.8*	2.6*	5.0*	6.6*	3.9*	2.9*	6.4*	3.8*
^{90}Sr	0.98		8.3	1.9	12	7.7		1.7*	13
^{99}Tc	18		4.8	8.3	10	6.1		11	4.5
^{95}Zr	8.5*	16*	3.9*	7.2*	9.7*	5.9	13*	9.8*	5.3*

Table 2.6 Activity concentrations of anthropogenic radionuclides in animals at Drigg Coast Sand Dunes predicted by the ERICA Tool from measured soil activity concentrations compared with those observed in biota samples.

	Activity concentration (Bq kg^{-1} FW)					Observed / Predicted ratio
		Observed				
Organism	**Predicted**	**Mean**	**n**	**Min.**	**Max.**	
^{241}Am						
Teal	2.0	0.96	2	0.53	1.4	0.48
Mallard*	2.0	0.32	1	-	-	0.16
Common Frog	2.0	2.2	2	2.0	2.3	1.10
^{137}Cs						
Mouse	390	3.8	2	3.0	4.5	0.01
Vole	390	7.4	2	5.0	9.6	0.02
Teal	100	2.1	2	2.0	2.2	0.02
Mallard	100	3.2	1	-	-	0.03
Caterpillar	7.5	4.2	1	-	-	0.56
Common Toad	79	2.3	1	-	-	0.03
Common Frog	79	2.6	2	2.5	2.7	0.03
Palmate Newt	79	12	3	9.0	13	0.15
Great Crested Newt	79	7.9	3	4.6	13	0.10
Common Lizard	1200	7.9	2	7.8	7.9	0.01
Slow worm	1200	17	3	6.4	31	0.02
$^{239+240}$Pu						
Teal	0.68	0.42	2	0.41	0.44	0.62
Mallard[1]	0.68	0.18	1	-	-	0.26
^{90}Sr						
Mallard	8.0	0.75	1	-	-	0.09
Common Frog	16	8.3	3	6.7	10	0.54
Palmate Newt	16	12	3	5.2	21	0.76
Great Crested Newt	16	7.7	3	7.3	8.2	0.49
Natterjack Toad	16	2.5	1	-	-	0.16
Slow worm	6.7	13	3	4.0	23	1.90
^{99}Tc						
Teal	7.7	4.9	1	-	-	0.63
Natterjack Toad	7.7	4.8	1	-	-	0.62

*Note, the liver of the mallard was missing and therefore not included in the analysis, as a result up to 45% of the body burden of $^{239+240}$Pu and ^{241}Am may be unaccounted for.

Table 2.7 Activity concentrations of anthropogenic radionuclides in plants at Drigg Coast Sand Dunes predicted by Tier 2 of the ERICA Tool from measured soil activity concentrations compared with those observed in biota samples.

Organism	Predicted	Activity concentration (Bq kg^{-1} FW) Observed Mean	n	Min.	Max.	Observed/ predicted ratio
^{241}Am						
Marram grass	0.24	9.3	8	3.0	27	38
Red fescue	0.24	3.7	5	1.5	6.1	15
Mixed vegetation	0.24	4.7	1	-	-	19
Heather	0.24	1.9	1	-	-	7.8
Moss	5.1	5.9	1	-	-	1.2
Lichen	5.1	5.2	1	-	-	1.0
^{137}Cs						
Marram grass	94	3.8	8	0.66	14	0.04
Red fescue	94	4.6	4	2.2	9.3	0.05
Mixed vegetation	94	1.9	1	-	-	0.02
Heather	540	25	1	-	-	0.05
Moss	760	21	1	-	-	0.03
Lichen	760	13	1	-	-	0.02
$^{239+240}$Pu						
Red fescue	0.42	1.3	5	0.7	2.4	3.2
^{90}Sr						
Red fescue	3.0	4.3	5	0.84	9.2	1.4
^{99}Tc						
Red fescue	420	1.4	5	1.3	1.4	0.003

2.3.2 Results

Tier 1 assessment

In the Tier 1 assessment the ERICA Tool returns a series of unit-less risk quotients for each radionuclide included in the assessment. Radionuclides with a RQ greater than unity are highlighted in red, similarly if the sum of the risk quotients is greater than unity this is also highlighted in red. When a red highlight appears the ERICA Tool recommends proceeding to a Tier 2 assessment. The Drigg Coast assessment was flagged red for ^{210}Po (RQ=1.37) and ^{226}Ra only (RQ=2.5). The highest RQ applied to an anthropogenic radionuclide was 0.16 for ^{241}Am. On this basis it can be considered that anthropogenic radionuclides, considered on their own, do not pose a risk to the biota of the Drigg Coast. However, the Tool suggests that when considered with naturally occurring radionuclides the possibility of ionising radiation affecting the biota of the Drigg Coast cannot be excluded.

Tier 2 assessment

In the Tier 2 assessment a more detailed set of results is produced by the ERICA Tool. This includes details of the dose contribution from radionuclides both internal and external to the organism as well as a total dose for each organism/radionuclide combination. At Tier 2 the dose results are compared to the ERICA "no effects" screening value of 10 µGy h^{-1}. To provide an additional level of conservatism, in effect, a factor of 0.33 is applied to this[2], therefore for organisms calculated to

[2] Note at Tier 2 the Tool applies an uncertainty factor of three to the best estimate total dose rate (to estimate the 95th percentile dose rate) and then compares this to the screening dose rate to estimate the conservative RQ rather than dividing the screening dose rate by three.

receive a dose greater than 3.3 µGy h^{-1} the possibility of adverse effects cannot be precluded. In the first Tier 2 assessment neither the anthropogenic nor the natural radionuclides in isolation contribute a dose above 3.3 µGy h^{-1}. However, when the dose contributions from both sets of radionuclides are considered together a dose rate of 3.4 µGy h^{-1} is estimated for both lichen and moss (Table 2.8). The radionuclide contributing most significantly to this dose is ^{210}Po (2.3 µGy h^{-1}) with the natural radionuclides contributing a total of 3.0 µGy h^{-1}. Am-241 gives the highest contribution by an anthropogenic radionuclide, with 0.19 µGy h^{-1} and 0.17 µGy h^{-1} for moss and lichen respectively. The next highest total dose is received by Marram grass (0.61 µGy h^{-1}) and the animal with the highest dose from anthropogenic radionuclides is the vole (0.47 µGy h^{-1}).

Table 2.8 Predicted radiation dose rates for organisms from sand dune habitats at the Drigg Coast SAC derived from measured site specific data for anthropogenic radionuclides (including measurements below the limit of detection) and UK average background soil activity concentrations for naturally occurring radionuclides.

	Predicted Dose rate (µGy h^{-1})								
Nuclide	**Marram Grass**	**Red Fescue**	**Moss**	**Lichen**	**Heather**	**Mouse**	**Vole**	**Teal**	**Mallard**
Anthropogenic radionuclides									
^{110m}Ag	6.5E-03	9.7E-04	7.5E-06	4.0E-05	8.3E-03	9.1E-03	7.2E-03	4.7E-04	2.5E-03
^{241}Am	3.0E-01	1.2E-01	1.9E-01	1.7E-01	6.0E-02	1.9E-01	3.2E-01	3.0E-02	1.0E-02
^{144}Ce	8.7E-04	1.7E-03	1.3E-03	1.1E-04	9.1E-04	1.6E-02	3.2E-02	6.2E-04	4.2E-04
^{60}Co	2.2E-04	2.5E-04	3.6E-05	5.7E-06	2.0E-04	6.8E-04	1.1E-03	1.4E-04	1.5E-04
^{134}Cs	1.2E-04	1.6E-04	8.9E-05	1.4E-05	1.2E-04	5.5E-04	8.6E-04	8.1E-05	7.8E-05
^{137}Cs	1.6E-02	1.6E-02	2.3E-03	1.4E-03	1.9E-02	2.9E-02	2.2E-02	7.5E-03	8.5E-03
^{152}Eu	1.3E-04	1.5E-04	2.3E-04	4.5E-04	1.6E-04	7.2E-06	5.3E-06	1.9E-06	2.1E-06
^{154}Eu	2.4E-04	2.2E-04	7.6E-05	1.5E-05	2.3E-04	3.2E-04	2.3E-04	8.2E-05	9.0E-05
^{95}Nb	1.5E-04	2.8E-04	4.3E-05	2.8E-06	1.4E-04	3.6E-03	9.1E-03	1.5E-04	1.4E-04
^{238}Pu	2.4E-03	2.5E-03	1.8E-02	1.7E-02	5.3E-03	3.8E-03	3.8E-03	4.0E-03	3.9E-03
^{240}Pu	1.3E-02	4.0E-02	9.1E-02	9.1E-02	2.8E-02	2.0E-02	2.0E-02	1.3E-02	5.4E-03
^{103}Ru	1.2E-04	9.8E-05	7.4E-05	3.5E-06	1.0E-04	1.5E-04	1.1E-04	4.9E-05	5.3E-05
^{106}Ru	2.3E-03	3.8E-03	5.0E-03	2.8E-04	7.8E-04	2.0E-02	4.1E-02	1.4E-03	1.1E-03
^{125}Sb	1.6E-04	1.3E-04	1.4E-04	1.4E-05	1.5E-04	7.6E-04	1.2E-03	5.1E-05	4.9E-05
^{90}Sr	1.4E-03	2.2E-03	3.4E-02	3.4E-02	3.4E-04	1.2E-02	1.3E-02	3.7E-04	4.7E-04
^{99}Tc	2.2E-02	8.0E-05	2.1E-02	2.1E-02	2.2E-02	4.1E-04	4.1E-04	1.6E-04	3.0E-05
^{95}Zr	1.7E-04	3.5E-04	1.3E-04	1.0E-05	1.6E-04	2.7E-03	5.6E-03	1.6E-04	1.4E-04
Total	**0.36**	**0.19**	**0.36**	**0.33**	**0.15**	**0.31**	**0.47**	**0.06**	**0.03**
Naturally occurring radionuclides									
^{210}Pb	1.9E-04	1.9E-04	1.3E-02	1.3E-02	8.5E-04	1.2E-04	1.2E-04	1.9E-04	1.9E-04
^{210}Po	4.6E-02	4.6E-02	2.3E+00	2.3E+00	3.7E-02	1.0E-03	1.0E-03	1.0E-03	1.0E-03
^{226}Ra	6.9E-02	6.9E-02	3.5E-01	3.5E-01	4.3E-02	5.2E-02	5.0E-02	6.4E-02	6.4E-02
^{228}Ra	2.4E-03	2.4E-03	5.6E-04	5.6E-04	2.2E-03	4.2E-03	3.1E-03	1.3E-03	1.4E-03
^{228}Th	1.0E-01	1.0E-01	2.3E-01	2.3E-01	3.9E-02	6.7E-03	4.9E-03	2.5E-03	2.7E-03
^{230}Th	1.4E-02	1.4E-02	3.4E-02	3.4E-02	5.2E-03	4.1E-05	4.0E-05	1.3E-04	1.3E-04
^{232}Th	1.2E-02	1.2E-02	2.9E-02	2.9E-02	4.4E-03	3.4E-05	3.4E-05	1.1E-04	1.1E-04
^{234}Th	2.7E-04	2.7E-04	2.9E-04	2.9E-04	1.3E-04	9.6E-05	7.2E-05	2.8E-05	3.1E-05
^{234}U	4.9E-03	4.9E-03	2.4E-02	2.4E-02	2.4E-03	3.8E-05	3.7E-05	1.7E-04	1.7E-04
^{238}U	4.2E-03	4.2E-03	2.0E-02	2.0E-02	2.0E-03	3.3E-05	3.3E-05	1.4E-04	1.4E-04
Total	**0.25**	**0.25**	**3.00**	**3.00**	**0.14**	**0.07**	**0.06**	**0.07**	**0.07**
Natural and Anthropogenic radionuclides									
Total	**0.61**	**0.44**	**3.40**	**3.40**	**0.28**	**0.38**	**0.53**	**0.13**	**0.10**

Table 2.8 (Continued) Predicted radiation dose rates for organisms from sand dune habitats at the Drigg Coast SAC derived from measured site specific data for anthropogenic radionuclides (including measurements below the limit of detection) and UK average background soil activity concentrations for naturally occurring radionuclides.

				Predicted dose rate (µGy h^{-1})					
Nuclide	Caterpillar	Common Toad	Common Frog	Palmate Newt	Great Crested Newt	Natterjack Toad	Adder	Common Lizard	Slow worm
Anthropogenic radionuclides									
^{110m}Ag	5.0E-03	6.9E-03	1.4E-03	6.6E-04	6.1E-04	9.7E-03	1.1E-02	8.2E-03	8.1E-03
^{241}Am	1.1E-01	8.8E-02	6.4E-02	1.4E-01	9.5E-02	1.1E-01	5.1E-02	1.4E-01	7.9E-02
^{144}Ce	5.5E-03	8.7E-03	4.6E-03	7.5E-03	4.9E-03	7.0E-03	6.0E-03	8.7E-03	6.1E-03
^{60}Co	3.9E-04	4.5E-04	4.6E-04	2.5E-04	1.8E-04	6.1E-04	7.1E-04	6.3E-04	5.1E-04
^{134}Cs	2.8E-04	3.2E-04	2.5E-04	2.3E-04	1.6E-04	3.9E-04	3.6E-04	4.1E-04	3.3E-04
^{137}Cs	1.7E-02	2.2E-02	2.2E-02	3.2E-03	2.7E-03	3.1E-02	3.4E-02	2.7E-02	2.8E-02
^{152}Eu	4.1E-06	5.5E-06	5.5E-06	4.1E-07	4.1E-07	7.8E-06	8.6E-06	6.6E-06	6.5E-06
^{154}Eu	1.8E-04	2.4E-04	2.4E-04	1.8E-05	1.8E-05	3.4E-04	3.8E-04	2.9E-04	2.9E-04
^{95}Nb	4.8E-04	2.6E-03	4.0E-04	3.9E-04	2.6E-04	6.3E-04	2.5E-03	6.6E-04	5.3E-04
^{238}Pu	2.8E-03	3.9E-03	4.0E-03	3.8E-03	3.8E-03	3.8E-03	3.8E-03	3.8E-03	3.8E-03
^{240}Pu	1.5E-02	2.1E-02	2.1E-02	2.0E-02	2.0E-02	2.0E-02	2.0E-02	2.0E-02	2.0E-02
^{103}Ru	8.1E-05	1.2E-04	1.2E-04	1.5E-05	1.5E-05	1.6E-04	1.8E-04	1.4E-04	1.4E-04
^{106}Ru	7.2E-03	1.1E-02	6.6E-03	9.9E-03	6.7E-03	1.0E-02	8.7E-03	1.2E-02	9.0E-03
^{125}Sb	4.4E-04	4.1E-04	2.1E-04	4.4E-04	2.7E-04	5.4E-04	4.3E-04	5.9E-04	4.4E-04
^{90}Sr	4.1E-04	8.6E-03	4.9E-03	5.3E-03	3.7E-03	1.0E-03	3.7E-03	8.4E-04	7.6E-03
^{99}Tc	1.0E-03	4.1E-04	2.8E-04	5.8E-04	3.6E-04	4.8E-04	4.1E-04	6.4E-04	2.6E-04
^{95}Zr	7.2E-04	1.6E-03	5.0E-04	7.0E-04	4.5E-04	8.3E-04	1.5E-03	9.5E-04	6.6E-04
Total	**0.17**	**0.18**	**0.13**	**0.19**	**0.14**	**0.19**	**0.14**	**0.22**	**0.17**
Naturally occurring radionuclides									
^{210}Pb	1.6E-04	3.6E-04	3.6E-04	3.2E-04	3.3E-04	3.5E-04	1.9E-04	1.8E-04	1.9E-04
^{210}Po	1.0E-03	1.0E-03	1.0E-03	1.0E-03	1.0E-03	1.0E-03	1.0E-03	1.0E-03	1.0E-03
^{226}Ra	1.5E-01	6.4E-02	6.4E-02	6.0E-02	6.0E-02	6.8E-02	6.9E-02	6.7E-02	6.7E-02
^{228}Ra	2.6E-03	3.3E-03	3.3E-03	3.5E-04	3.5E-04	4.6E-03	5.1E-03	3.9E-03	3.9E-03
^{228}Th	2.3E-02	5.8E-03	5.8E-03	1.2E-03	1.2E-03	7.9E-03	8.7E-03	6.7E-03	6.7E-03
^{230}Th	2.9E-03	1.3E-04	1.3E-04	1.3E-04	1.3E-04	1.3E-04	1.3E-04	1.3E-04	1.3E-04
^{232}Th	2.4E-03	1.1E-04	1.1E-04	1.1E-04	1.1E-04	1.1E-04	1.1E-04	1.1E-04	1.1E-04
^{234}Th	9.0E-05	7.7E-05	7.7E-05	7.2E-06	7.4E-06	1.1E-04	1.2E-04	9.0E-05	8.9E-05
^{234}U	3.0E-03	1.7E-04	1.7E-04	1.7E-04	1.7E-04	1.7E-04	1.7E-04	1.7E-04	1.7E-04
^{238}U	2.6E-03	1.4E-04	1.4E-04	1.5E-04	1.5E-04	1.5E-04	1.5E-04	1.5E-04	1.5E-04
Total	**0.19**	**0.08**	**0.08**	**0.06**	**0.06**	**0.08**	**0.09**	**0.08**	**0.08**
Natural and Anthropogenic radionuclides									
Total	**0.36**	**0.25**	**0.21**	**0.26**	**0.20**	**0.27**	**0.23**	**0.30**	**0.24**

In the second Tier 2 assessment (Table 2.9), using only data with values above the limit of detection, there is no change in the outcome of the assessment and all figures quoted for the first Tier 2 assessment (above) apply to this assessment with one exception. This is the organism with the highest contribution from anthropogenic radionuclides, which is moss (0.36 µGy h^{-1}). The dose to moss was not elevated above that predicted in the first Tier 2 assessment, rather in the second Tier 2 assessment the dose to voles from anthropogenics was reduced to 0.12 µGy h^{-1}, mainly due to the exclusion of the vole organism data for ^{241}Am (which was below the limit of detection). This had the effect of reducing the dose contribution of ^{241}Am from 0.32 µGy h^{-1} to 0.064 µGy h^{-1}.

To assess the nature of the risk to the ecosystem due to the "high" dose received by lichen and moss, the radiological effects summary page in ERICA was investigated; this contained no relevant data.

Table 2.9 (a and b) Predicted radiation dose rates for organisms from sand dune habitats at the Drigg Coast SAC derived from measured site specific data for anthropogenic radionuclides where measurements were above the limit of detection only.

	Predicted Dose rate (μGy h^{-1})								
Nuclide	**Marram Grass**	**Red Fescue**	**Moss**	**Lichen**	**Heather**	**Mouse**	**Vole**	**Teal**	**Mallard**
^{241}Am	3.0E-01	1.2E-01	1.9E-01	1.7E-01	6.0E-02	6.4E-02	6.4E-02	3.0E-02	1.0E-02
^{134}Cs	-	-	8.9E-05	1.4E-05	-	-	-	-	-
^{137}Cs	1.6E-02	1.6E-02	2.3E-03	1.4E-03	1.9E-02	2.9E-02	2.2E-02	8.3E-03	7.7E-03
^{238}Pu	2.8E-03	2.8E-03	2.0E-02	2.0E-02	6.1E-03	4.4E-03	4.4E-03	4.6E-03	4.6E-03
^{240}Pu	1.3E-02	4.0E-02	9.1E-02	9.1E-02	2.8E-02	2.0E-02	2.0E-02	1.3E-02	5.4E-03
^{106}Ru	-	-	5.0E-03	2.8E-04	-	-	-	-	-
^{90}Sr	1.5E-03	2.2E-03	3.6E-02	3.6E-02	3.6E-04	1.3E-02	1.4E-02	5.0E-03	4.7E-04
^{99}Tc	2.4E-02	8.0E-05	2.3E-02	2.3E-02	2.4E-02	4.5E-04	4.5E-04	2.8E-04	4.5E-04
Total	**0.35**	**0.18**	**0.36**	**0.34**	**0.14**	**0.13**	**0.12**	**0.06**	**0.03**
Natural and Anthropogenic radionuclides									
Total	**0.61**	**0.43**	**3.40**	**3.40**	**0.27**	**0.20**	**0.18**	**0.13**	**0.10**

	Predicted Dose rate (μGy h^{-1})								
Nuclide	**Caterpillar**	**Common Toad**	**Common Frog**	**Palmate Newt**	**Great Crested Newt**	**Natterjack Toad**	**Adder**	**Common Lizard**	**Slow worm**
^{241}Am	2.0E-01	6.4E-02	6.9E-02	6.4E-02	6.4E-02	6.4E-02	6.4E-02	6.4E-02	6.4E-02
^{134}Cs	-	-	-	-	-	-	-	-	-
^{137}Cs	1.7E-02	2.2E-02	2.2E-02	3.2E-03	2.7E-03	4.2E-02	2.2E-01	2.7E-02	2.8E-02
^{238}Pu	3.3E-03	4.6E-03	4.6E-03	4.4E-03	4.4E-03	4.4E-03	4.4E-03	4.4E-03	4.4E-03
^{240}Pu	1.5E-02	2.1E-02	2.1E-02	2.0E-02	2.0E-02	2.0E-02	2.0E-02	2.0E-02	2.0E-02
^{106}Ru	-	-	-	-	-	-	-	-	-
^{90}Sr	3.8E-04	9.2E-03	4.9E-03	5.3E-03	3.7E-03	1.4E-03	4.0E-03	3.3E-03	7.6E-03
^{99}Tc	4.4E-04	4.5E-04	4.5E-04	4.4E-04	4.5E-04	2.8E-04	4.5E-04	4.5E-04	4.5E-04
Total	**0.23**	**0.12**	**0.12**	**0.10**	**0.10**	**0.13**	**0.31**	**0.12**	**0.12**
Natural and Anthropogenic radionuclides									
Total	**0.42**	**0.20**	**0.20**	**0.16**	**0.16**	**0.22**	**0.40**	**0.20**	**0.20**

Comparison of observed and predicted organism activity concentrations at Tier 2

The predictions of organism activity concentrations made by Tier 2 of the ERICA Tool based on observed soil activity concentrations were variable but generally gave higher values than measured observations (Tables 2.6 and 2.7).

Am-241 comparisons were favourable for animals with predictions within the range of observations for the Common Frog and about double the mean for Teal. Lichen and moss predictions for ^{241}Am were very close but were under-predicted for other vegetation by less than one order of magnitude for heather and over one order of magnitude but less than two for Marram grass, Red fescue and Mixed vegetation.

More observed versus predicted organism activity concentration comparisons could be made for ^{137}Cs than any other radionuclide. These showed that the ERICA Tool consistently over-predicted organism activity concentrations. In plants this was always by over one order of magnitude but less than two orders of magnitude. More variability was seen for animals with predictions for newts being 6.6 and 10 times the mean observed activity concentrations for Palmate and Great Crested Newts respectively. The largest over-prediction for ^{137}Cs was seen in mice and Common Lizards with values 103 and 164 times mean observed activity concentrations.

Sr-90 and $^{239+240}Pu$ predictions were all within an order of magnitude of mean measured data and mostly fell within the range of minimum and maximum records. Amongst the birds (Mallard and Teal) $^{239+240}Pu$ organism activity concentrations were slightly under-predicted. Whilst for plants (Red fescue) a slight under-prediction was seen with respect to the mean activity concentrations, this was also the case for ^{90}Sr.

Tc-99 comparisons were limited to single observations for Teal and Naterjack Toad and five for Red fescue. Both animal predictions were nearly double the corresponding observed activity concentrations, but the prediction for ^{99}Tc in Red fescue was 330 times the mean observed measurements, which showed little variation.

Comparison of observed and predicted organism activity concentrations at Tier 3

At Tier 3 the predicted organism activity concentrations were generally the same as those predicted at Tier 2. However, Tier 3 is able to provide additional information to Tier 2 by calculating the predicted range of expected organism activity concentrations. This provides a more robust means of comparing the observed data with predicted data (Tables 2.10 and 2.11; Figures 2.3 and 2.4).

Am-241 activity concentrations were mostly under-predicted at Tier 2 and at Tier 3 the corresponding predictions show only some small deviations from this. For animals, predictions were not greatly below the mean observed activity concentrations and all observations were between the 5th and 95th percentile of the predicted data. Predictions of organism activity concentrations for animals can be considered not to be significantly less than observed activity concentrations. For vegetation, predictions for moss and lichen were very close to observed data but in other vegetation ^{241}Am activity concentrations were under-predicted with minimum observed values being in excess of the Tier 3 95th percentile prediction.

In the case of ^{137}Cs the over-predictions by the ERICA Tool can be seen to be statistically significant for nearly all organisms for which comparisons were made, with respect to the mean observed activity concentrations. The exception is the caterpillar, the only organism to have a mean observed activity concentration within 90 % of the predicted range of activity concentrations (i.e. within the 5th and 95th percentile). If maximum and minimum observed activity concentrations are included, Red fescue can be brought within this 90 % range.

All mean observed data for activity concentrations of ^{90}Sr and $^{239+240}Pu$ fell between the 5th and 95th percentile of predicted data. With respect to the mean data the predictions are realistic, although the mean observed measurement for $^{239+240}Pu$ in Red fescue is the same as the 95th percentile of the predicted range (1.3 Bq kg^{-1}). The upper range of observed data for $^{239+240}Pu$ in Red fescue (2.4 Bq kg^{-1} maximum) is therefore significantly above the predicted mean.

Table 2.10 Activity concentrations of anthropogenic radionuclides in animals at Drigg Coast Sand Dunes predicted by Tier 3 of the ERICA Tool from measured soil activity concentrations compared with the activity concentrations observed in the samples.

Organism	Predicted activity concentration ($Bq\ kg^{-1}$ FW) Mean	5^{th} percentile	95^{th} percentile	Measured activity concentration ($Bq\ kg^{-1}$ FW) Mean	Min.	Max.
^{241}Am						
Teal	2.0	0.10	6.0	0.96	0.53	1.4
Mallard*	2.1	0.10	6.2	0.32	-	-
Common Frog	2.0	0.10	5.8	2.2	2.0	2.3
^{137}Cs						
Teal	100	5.0	360	2.1	2.0	2.2
Mallard	110	4.9	400	3.2	-	-
Caterpillar	7.1	0.11	29	4.2	-	-
Mouse	410	40	1400	3.8	3.0	4.5
Vole	400	33	1200	7.4	5.0	9.6
Common Toad	90	6.8	330	2.3	-	-
Common Frog	86	6.6	290	2.6	2.5	2.7
Palmate Newt	82	6.3	290	12	9.0	13
Common Lizard	460	14	1800	7.9	7.8	7.9
Slow worm	450	17	1700	17	6.4	31
$^{239+240}$Pu						
Teal	0.68	0.04	2.0	0.42	0.41	0.44
Mallard*	0.68	0.03	20	0.18	-	-
^{90}Sr						
Mallard	7.9	0.54	27	0.75	-	-
Common Frog	16	1.5	55	8.3	6.7	10
Natterjack Toad	16	1.2	52	2.5	-	-
Palmate Newt	16	1.4	52	12	5.	21
Slow worm	170	8.30	600	13	4.0	23
^{99}Tc						
Teal	7.7	0.39	23	4.9	-	-
Natterjack Toad	8.2	0.48	0.25	4.8	-	-

*Note, the liver of the mallard was missing and therefore not included in the analysis, as a result up to 45% of the body burden of $^{239+240}$Pu and ^{241}Am may be unaccounted for.

Table 2.11 Activity concentrations of anthropogenic radionuclides in plants at Drigg Coast Sand Dunes predicted by Tier 3 of the ERICA Tool from measured soil activity concentrations compared with the activity concentrations observed in the samples.

Organism	Predicted activity concentration ($Bq\ kg^{-1}$ FW) Mean	5th percentile	95th percentile	Measured activity concentration ($Bq\ kg^{-1}$ FW) Mean	Min.	Max.
^{241}Am						
Marram grass	0.25	0.004	0.67	9.3	3.0	27
Red fescue	0.24	0.04	0.70	3.7	1.5	6.1
Mixed vegetation	0.24	0.05	0.64	4.7	-	-
Heather	0.23	0.01	0.70	1.9	-	-
Moss	5.2	0.33	16	5.9^{+}	-	-
Lichen	5.0	0.31	14	5.2	-	-
^{137}Cs						
Marram grass	98	7.6	320	3.8	0.66	14
Red fescue	100	7.7	340	4.6	2.2	9.3
Mixed vegetation	100	8.4	370	1.9	-	-
Heather	580	74	1600	25	-	-
Moss	800	210	2000	21^{+}	-	-
Lichen	800	210	1900	13	-	-
^{240}Pu						
Red fescue	0.40	0.04	1.3	1.3	0.70	2.4
^{90}Sr						
Red fescue	2.8	0.004	10	4.3	0.84	9.2
^{99}Tc						
Red fescue	410	130	890	1.4	1.3	1.4

$^{+}$Note moss activity concentrations are reported on a dry weight basis

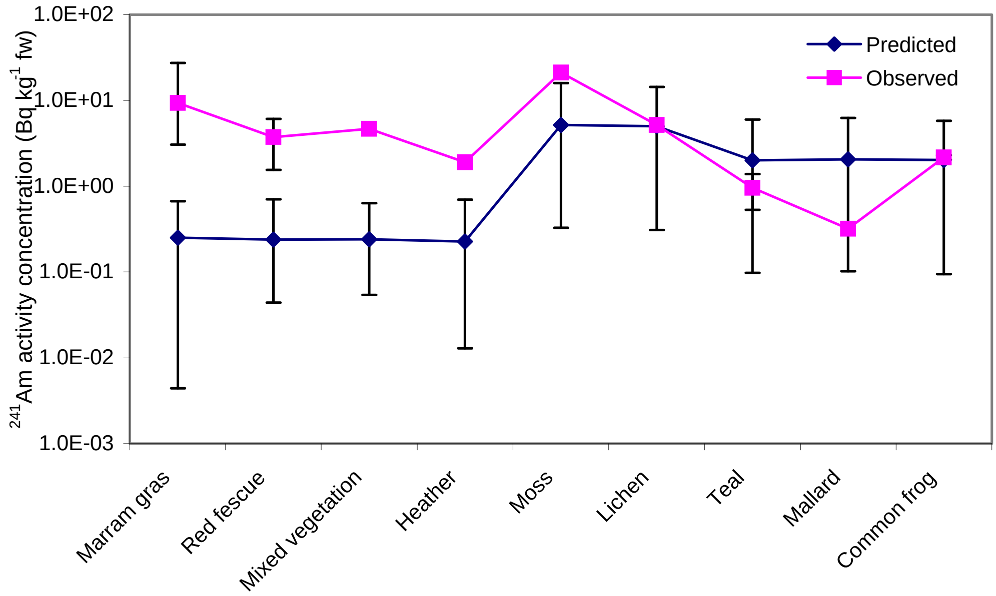

Figure 2.3 Mean predicted (with 5th and 95th percentile) and observed (with maximum and minimum) activity concentrations of ^{241}Am in organisms of the Drigg Coast Sand Dunes.

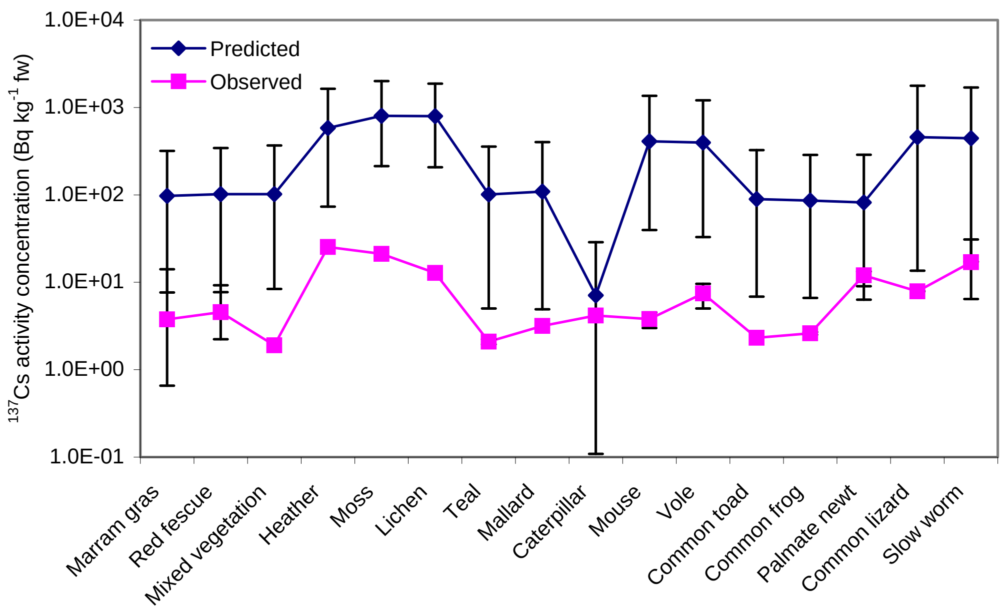

Figure 2.4 Mean predicted (with 5th and 95th percentile) and observed (with maximum and minimum) activity concentrations of ^{137}Cs in organisms of the Drigg Coast Sand Dunes.

The observed data for ^{99}Tc in Teal fell in the middle of the predicted distribution. However, the predictions for ^{99}Tc in Red fescue were significantly higher than the observed data with the 5th percentile of predicted data, 132 Bq kg^{-1}, compared to a maximum observed value of 1.4 Bq kg^{-1} FW.

2.3.3 Discussion

The impact of anthropogenic radionuclides on the sand dune ecosystem of the Drigg Coast SAC is well below the level where effects are likely to be seen in biota. Given the free draining nature of the substrate, radionuclides deposited on the dunes, either directly or by sea- to-land transfer, would be expected to have a short residence time in the soil compared to soils with higher clay or organic contents. Therefore, there may be a 'reduced opportunity' for accumulation of radionuclides by organisms. The combination of short residence time and 'reduced opportunity' for accumulation would be expected to result in comparatively low internal and external exposure rates.

Naturally occurring radionuclides were indicated by the assessment to be present in the soil in sufficiently high activity concentrations that adverse effects on biota cannot be excluded. Po-210 and ^{226}Ra were identified at Tier 1 as presenting a risk worthy of further investigation, while at Tier 2 ^{210}Po was identified as contributing significantly to dose rates in lichen and moss above levels at which effects can be excluded. The assessment was conducted assuming a soil activity concentration of 12 Bq kg^{-1} for each of the natural radionuclides. This is low in comparison to average soils for which activities of around 37 Bq kg^{-1} and 22 Bq kg^{-1} would be considered normal for radionuclides in the ^{232}Th and ^{238}U decay series respectively (Eisenbud and Gesell, 1997). As the assessment suggested that the risk of impact was only just above the screening value plus safety margin (10 µGy h^{-1} divided by 3), at 3.4µGy h^{-1}, it was considered that no further assessment was required and that the sand dune ecosystem at Drigg Coast SAC was not at risk of impacts from ionising radiation. Indeed, the maximum predicted dose rate was well below the 10 µGy h^{-1} ERICA screening value, which is cautious relative to the 40 µGy h^{-1} (chronic exposure) for terrestrial animals, below which it has previously been suggested that no measurable population effects would occur (IAEA, 1992; UNSCEAR, 1996) and which is used (for terrestrial animals) as a benchmark dose rate within the USDoE (2002) graded approach.

It is also relevant that the radiosensitivity of those organisms highlighted (lichen and moss) is likely to be low, although the ERICA Tool provided no data to support this.

That the ERICA Tool/methodology highlights such low activity concentrations of common natural radionuclides as presenting potential risks to natural ecosystems suggests a potential problem. Natural ecosystems have undoubtedly evolved and flourish in natural ionising radiation regimes several times more intense than that experienced at the Drigg Coast Sand Dunes. It could be counter-productive to raise the screening levels within ERICA to account for natural radionuclides as the sensitivity of the assessment to contributions from anthropogenic radionuclides would be reduced. However, the safety margin (uncertainty factor) placed on top of the 10 µGy h^{-1} screening value could be reviewed as it represents a considerable safety margin over other benchmarks such as that used by USDoE (2002).

Within this assessment, the ERICA Tool/methodology appears somewhat inconsistent in its prediction of organism activity concentrations. Whilst an over-prediction, as seen for ^{137}Cs at the Drigg Coast SAC, can be considered as introducing an element of conservatism, an under-prediction, as seen in the case of ^{241}Am in higher plants at the Drigg Coast SAC, is of more concern. The under-prediction of ^{241}Am in organisms at the Drigg Coast Sand Dunes may be a function of the nature of the soils at this site and the high particle reactivity of this radionuclide. Large grained sandy soils will present fewer binding sites with which ^{241}Am can react, making it more available for uptake by plants. Future adaptations of dose to biota methodologies could consider this and allow some degree of parameterisation of soil types. Whilst $^{239+240}$Pu, which are also highly particle reactive, were slightly over-predicted in higher plants, the effect was not as strong as observed for ^{241}Am and was similar to that for the non particle reactive ^{90}Sr, suggesting that applying a "soil type bias" should be investigated with caution. A more likely cause of the observed inconsistency in predictions of organism activity concentrations is the fact that the Drigg Coast Sand Dunes represent a site experiencing continued deposition via sea–to-land transfer; this process is most effective for particle reactive radionuclides such as ^{241}Am and $^{239+240}Pu$ (which are under-predicted by the ERICA Tool) and least effective for conservative radionuclides such as ^{99}Tc and ^{137}Cs (which are over-predicted by the Tool). The ERICA Tool and methodology is not constructed to deal with this situation and expects data to represent an equilibrium state between the activity present in the biota and the activity present in the soil substrate.

Comments on ERICA Tool and D-ERICA

The initial intent during this assessment was to follow the ERICA methodology and use the ERICA Tool as described in D-ERICA and the associated Help files within the ERICA Tool. D-ERICA was used by both assessors familiar with ERICA and novices. Both groups found that D-ERICA did not function as a user guide for the ERICA Tool or a guide through the assessment process. As such its use was terminated and the assessment conducted intuitively. The following points were noted in terms of general 'usability':

- An absence of information within the help files or D-ERICA on the parameterisation of the default reference organisms provides no indication to the assessor of circumstances when default reference organisms do not represent organisms of interest. It is therefore unclear when it is appropriate to create new "custom" reference organisms.
- When a new reference organism is created there are no means to review (check) or edit (correct) parameters. Additionally, as CRs for custom organisms would generally be those for the corresponding default reference organism type, these should be able to be entered via tick box in the *add organism* function to minimise data entry.
- Neither data entry or data retrieval are optimised within the ERICA Tool. The ability to cut and paste blocks of data into and out of the ERICA Tool would enhance its usability considerably as would the ability to carry data between Tiers.
- In Tier 1 the results are very limited and do not facilitate an adequate interpretation or understanding of the problems a site may have. Each isotope is listed with a risk quotient associated with a single reference organism. Where a failure is reported for a nuclide it is not clear what this means for any organism other than the "limiting organism" which are reported separately for each radionuclide.
- A clearer description of the pass/fail criteria for Tier 2 would be helpful – as referring to a 'probability of exceeding the screening dose rate' is confusing in what is seen to be a deterministic assessment.
- It is not entirely clear what Tier 3 adds to an assessment that has already 'failed' at Tier 2. The probabilistic assessment is likely to reduce the 95th percentile dose somewhat, below the 3-fold factor used at Tier 2, but not enough to negate any 'fails'. In this case, the ERICA Tool was unable to resolve the relatively minor 'fail' for moss and lichen at Tier 2, due to natural radioactivity, as no relevant effects data were provided.
- A likely outcome is that use of the ERICA Tool for decision-making at present may effectively exclude a situation that cannot achieve a clear 'pass' at Tier 2. This means (given the factor of 3 applied for uncertainty) that a 'de facto' dose limit of 3.3 $\mu Gy\ h^{-1}$ has been introduced.

Stakeholder input

This study, since the inception of its forebear the FASSET assessment, has been subject to a continued process of stakeholder engagement. This has included the identification of organisms of interest and review of the FASSET assessment at workshops by invitation and an open public meeting to explain the process and appraise opinions on it. The process of stakeholder engagement is ongoing and a review of the current ERICA assessment will mark its conclusion.

Input from the workshops has been useful in highlighting deficiencies in the FASSET methodology, most notably the exclusion of amphibians from the reference organism list, as well as eliciting general views of this type of assessment. The public meeting provided a broader base of individuals beyond those with a specific association with the Drigg Coast SAC. This meeting served to highlight and expand on many of the general perceptions of this type of assessment. The views were highly varied with opinions expressing concern over both complexity and over simplification within the methodology. In general, it was considered that the opacity of the methodology relative to the ability of lay people to understand it may lead to distrust of this type of assessment.

Developments incorporated in the ERICA Tool and methodology have addressed a number of the points raised by stakeholders, particularly expansion and rationalisation of the set of reference organisms and the ability to add ‘custom’ organisms into the assessment. The methods used to extend the matrix of available CRs avoids gaps in the provision of internal radionuclide activity concentrations, and internal doses are now calculated for plants. The method used to derive a ‘screening dose rate’ is consistent with that used for chemical pollutants, and appears to be adequately conservative (i.e. erring on the side of safety). However, the methodology, and the scientific reasoning that lies behind it, is complex.

An intelligent lay persons guide to the methodology and its assumptions was requested following the FASSET assessment and would be of benefit to both stakeholders and assessors.

Overall the opinion has been that stakeholder engagement is beneficial in fostering trust from the interested parties where radiological impact issues are concerned. This is an important adjunct to more formal engagement with statutory stakeholders, such as regulatory agencies.

This assessment has found the ERICA Tool to provide a useful arena for documenting the stakeholder engagement process in close association with the assessment inputs and outputs. However, in the options for methods of communications no option was provided for direct targeting of individuals (via mail, email or telephone).

2.4 Conclusions

- The ERICA Tool worked adequately in excluding the probability of adverse effects in biota from anthropogenic radionuclides in this relatively simple case study.
- At Tier 2 the risk quotient derived from assessments including site-specific soil data and predictions of organism activity concentrations were mostly greater than for the measured activity concentrations, the only exceptions to this were herbaceous plants (Table 2.12). In this specific instance, the ERICA methodology does effectively reduce conservatism with increasing data availability, justifying the Tiered Approach.
- To ensure uptake and effective use by practitioners and interpretation by stakeholders the provision of a simplified description and guidance associated with the ERICA Tool / methodology is essential.

Table 2.12 Comparison of risk quotients output from Tier 2 assessments driven by site specific soil and organism activity concentrations.

Organism	Risk quotient (unitless) for assessments driven by Site soil activity concentrations	Risk quotient (unitless) for assessments driven by Site organism activity concentrations	Organism driven / soil driven ratio
Marram grass	0.003	0.006	1.830
Red fescue	0.003	0.004	1.310
Mixed vegetation	0.003	0.005	1.390
Moss	0.035	0.034	0.986
Lichen	0.035	0.034	0.979
Heather	0.003	0.003	0.935
Mouse	0.003	0.002	0.773
Vole	0.002	0.002	0.759
Teal	0.002	0.001	0.684
Mallard	0.002	0.001	0.515
Caterpillar	0.004	0.004	0.997
Common Toad	0.002	0.002	0.945
Common Frog	0.002	0.002	0.948
Palmate Newt	0.002	0.002	0.936
Great Crested Newt	0.002	0.002	0.919
Natterjack Toad	0.002	0.002	0.969
Adder	0.004	0.004	0.999
Common Lizard	0.004	0.002	0.547
Slow worm	0.004	0.002	0.539

2.5 References

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3 Loire River case study

(P. Ciffroy (EDF))

3.1 Site study

The number of nuclear power plants (NPP) located on the Loire River and its tributaries has steadily increased since 1963; by 2000, there were fourteen reactors operating on five different sites (Figure 3.1). Although some dams are located on the upper Loire, all nuclear power plants are located below these on a stretch of the river some 350 km from its estuary. One of the Loires main tributaries, the Vienne River, receives radioactive releases from the Civaux NPP.

The Loire River is considered one of the last 'wild' rivers in Europe. Several local to international initiatives (Rivernet, 2004; MEDD, 2004a; 2004b) recognise and establish the important status of the Loire River and its valley, which in part is registered in the world patrimony (UNESCO, 2000). At 1010 km, the Loire is the longest river in France, its watershed covering a fifth of the national territory. The Loire provides a habitat to 103 plant species of natural heritage interest and 107 nationally protected animal species; the valley includes a number of Natura 2000 sites. As such, an accurate inventory of habitats and species is available for specific sites. A rapid summary of these data leads to the following conclusions: the high variability of the flow rate creates many micro-habitats, (shores, meanders, islands and islet etc.); the fluvial dynamic influences the vegetation near the river bed, generating a high diversity; the river is an important migratory pathway for fish (including salmon and lamprey); species of national importance are present (e.g. beaver, little and common tern).

3.1.1 Case study application

The application of the ERICA Tool has taken the form of a comparison of predictions using different levels of available input data and at different tiers.

For all the calculations within the case study application of the ERICA Tool actual releases of ^{3}H, ^{60}Co, ^{58}Co, ^{110m}Ag, ^{137}Cs and ^{54}Mn during the period 1994-1999 were used. These radionuclides were chosen because they are the main components of releases by French NPPs and because they are known to have contrasting biogeochemical behaviours. Most predictions were performed for one typical measurement station (Beaulieu) situated downstream of the Belleville nuclear site which is the most upstream site on the Loire (Figure 4.1). Some predictions were also performed for measurement station Blois situated downstream from the St. Laurent nuclear site.

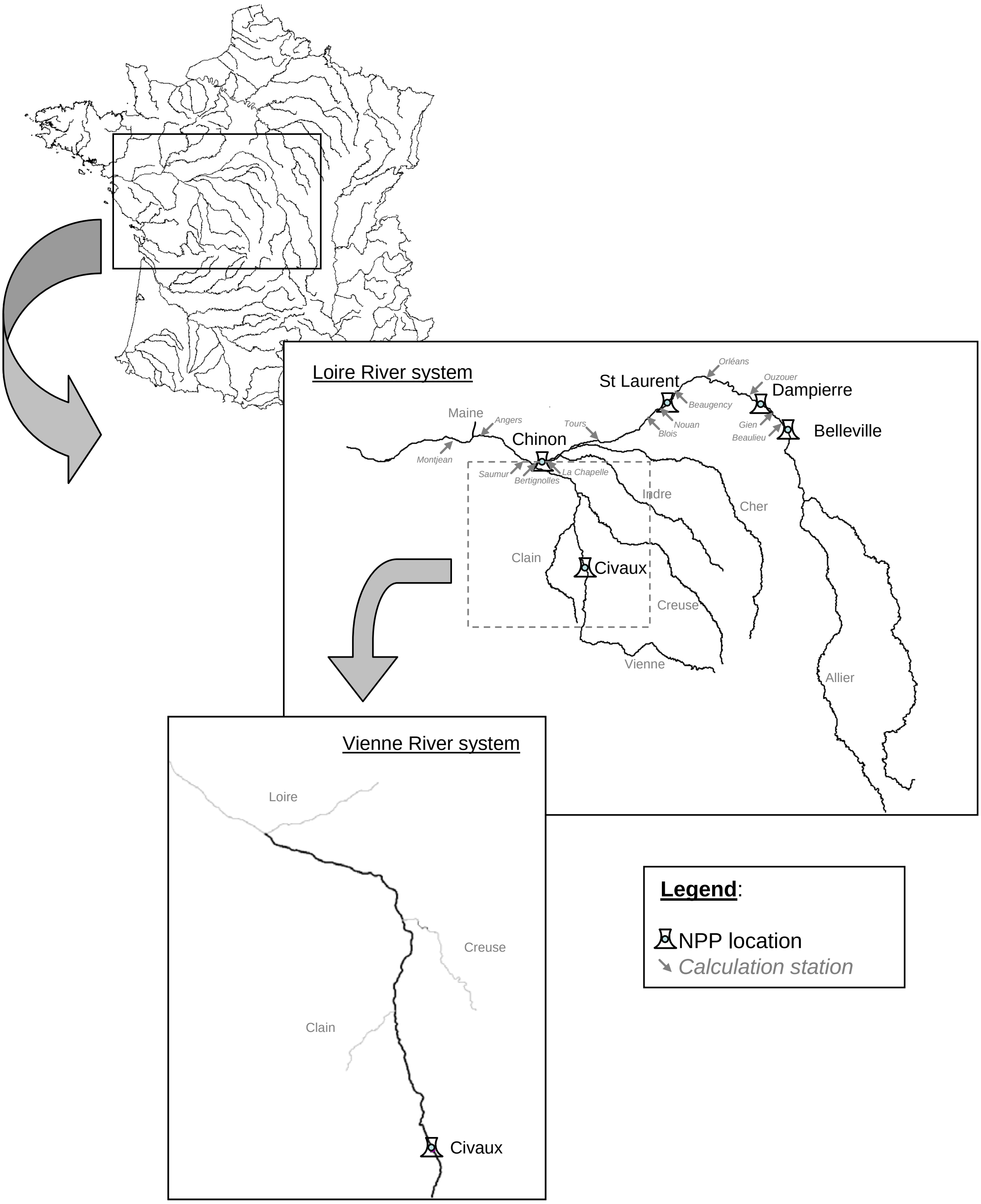

Figure 3.1 Nuclear installations on the Loire River.

3.2 Calculation 1 - Tier 1 using mean release data

3.2.1 Input data

Water concentrations were estimated for the Beaulieu sampling station using the SRS-19 river transport model available in the ERICA Tool. Input data were the mean annual releases from the Belleville NPP (in Bq s^{-1}) (Table 3.1), as well as the mean annual flow rate (342 m^3 s^{-1}), the depth (1.5 m) and width (255 m) of the river at Beaulieu. The most conservative annual conditions over the period 1994-1999 were considered (i.e. maximum annual releases). Concentrations in river water calculated by the ERICA Tool are presented in Table 3.1.

Table 3.1 Mean annual releases from the Belleville NPP.

Radionuclide	Mean annual releases (Bq s^{-1})	Predicted water activity concentrations (Bq l^{-1})
^{110m}Ag	72	$1.41x10^{-3}$
^{58}Co	61.9	$1.21x10^{-3}$
^{60}Co	99	$1.94x10^{-3}$
^{137}Cs	14.7	$2.90x10^{-4}$
^{3}H	1083	$2.12x10^{-2}$
^{54}Mn	6.3	$1.23x10^{-4}$

3.2.2 Results

The outputs of the Tier 1 assessment with the inputs as described above are presented in Table 3.2.

Table 3.2 Risk quotients and limiting reference organisms at Beaulieu predicted using Tier 1 of the ERICA Tool and assuming discharge data.

Radionuclide	Risk quotient	Limiting reference organism
^{110m}Ag	$2.7x10^{-3}$	Bivalve mollusc
^{58}Co	$9.1x10^{-4}$	Insect larvae
^{60}Co	$3.3x10^{-3}$	Insect larvae
^{137}Cs	$1.4x10^{-4}$	Mammal
^{3}H	$3.7x10^{-8}$	Insect larvae
^{54}Mn	$5.4x10^{-5}$	Bivalve mollusc
Sum	**$7.1x10^{-3}$**	

3.2.3 Discussion and comments

Except for ^{137}Cs, limiting reference organisms are benthic organisms (bivalve mollusc living at water-sediment interface and insect larvae living in sediments, as defined by the default occupancy factors in the Tool). This observation demonstrates the importance of the 'sedimentology' model used within the calculation of risk quotient.

In the case of the Beaulieu station, more sophisticated models (see LIDO-TRACER discussed below) taking into account daily conditions (flow rate and suspended matter concentration), as well as specific river parameters (particles settling velocity, critical deposition shear stress) predict that no sediment deposits. The ERICA approach based on a generic hydraulic (SRS-19) and sedimentological (i.e. equilibrium K_d values) models is therefore very conservative in this specific case.

3.3 Calculation 2 - Tier 1 assuming mean concentrations in river water and sediment

3.3.1 Input data

Input data were provided using a time-dependent fully 1D hydraulic model named LIDO-TRACER. Siclet et al. (2002) provide a full description of the LIDO-TRACER model with examples of validation. Data on the bathymetry of the river (i.e. section profiles) and flow regimes were collected, and tracer experiments were conducted in the Loire to calibrate the model parameters (e.g. friction and dispersion coefficients). Actual releases and flow rates of the river at an hourly time step were considered as input data to the LIDO-TRACER model for this assessment. Such an approach allowed the calculation of the total concentration of radionuclides at the investigated station on an hourly time step and this was further averaged to an annual time step. The maximum annual averaged concentration over the period 1994-1999 was used as the input for this calculation. Hence, the result equated to the same input data as for calculation 1 except that input activity concentrations were calculated using a site specific and parameterised model rather than the generic SRS-19 model. At the Beaulieu station, the LIDO-TRACER model predicted that no sediment would be found. Consequently, a sediment activity concentration of zero was assumed for all radionuclides. Input data entered into the ERICA Tool are presented in Table 3.3.

Table 3.3 Averaged annual concentrations in river water at Beaulieu calculated by the LIDO-TRACER model.

	Activity concentration	
Radionuclide	**Concentration in water (Bq l^{-1})**	**Concentration in sediment (Bq kg^{-1})**
^{110m}Ag	$5.0x10^{-4}$	0
^{14}C	$1.15x10^{-3}$	0
^{58}Co	$2.0x10^{-4}$	0
^{60}Co	$3.9x10^{-4}$	0
^{137}Cs	$6.0x10^{-5}$	0
^{3}H	5	0
^{54}Mn	$3.6x10^{-5}$	0

3.3.2 Results

The outputs of the Tier 1 assessment with the inputs as described above are presented in Table 3.4.

Table 3.4 Risk quotients and limiting reference organisms at Beaulieu predicted using Tier 1 of the ERICA Tool and assuming water and sediment inputs estimated by LIDO-TRACER.

Radionuclide	Risk quotient	Limiting reference organism
^{110m}Ag	$1.3x10^{-3}$	Bivalve mollusc
^{14}C	$7.3x10^{-5}$	Bird, mammal
^{58}Co	$2.0x10^{-4}$	Insect larvae
^{60}Co	$8.9x10^{-4}$	Insect larvae
^{137}Cs	$3.8x10^{-5}$	Mammal
^{3}H	$1.2x10^{-5}$	Insect larvae
^{54}Mn	$2.1x10^{-5}$	Bivalve mollusc
Sum	**$2.5x10^{-3}$**	

3.3.3 *Discussion and comments*

The input of concentrations in water and sediments estimated using the LIDO-TRACER model resulted in a decrease in risk quotients at the Beaulieu station for all the radionuclides, except for ^{3}H, for which large differences were observed between the two calculations as a consequence of the SRS-19 model predicting lower water activity concentrations than LIDO-TRACER. This result shows that the generic approach proposed in calculation 1 for calculating mean annual activities for ^{3}H, that shows high temporal variations in the releases, must be used with caution.

It was also a surprise that, although sediment concentrations were entered as zero, limiting reference organisms are those living in the sediment (bivalve mollusc and insect larvae).

3.4 Calculation 3 - Tier 1 assuming mean concentrations in river water and sediment at a site with sedimentation

3.4.1 *Input data*

Input data were generated by the same methodology as for calculation 2, but at a measurement station (Blois) where sediment may deposit during low flow rate periods. Input data entered into the ERICA Tool are reported in Table 3.5. Calculations were limited to ^{110m}Ag, ^{58}Co and ^{137}Cs.

Table 3.5 Averaged annual concentrations in river water and sediments at Blois calculated by the LIDO-TRACER model.

	Activity concentration	
Radionuclide	**Water (Bq l^{-1})**	**Sediments (Bq kg^{-1})**
^{110m}Ag	1.0×10^{-3}	6.7×10^{-2}
^{58}Co	1.0×10^{-3}	3.4×10^{-2}
^{137}Cs	1.3×10^{-4}	1.15×10^{-3}

3.4.2 *Results*

The outputs of the Tier 1 assessment with the inputs as described above are presented in Table 3.6.

Table 3.6 Risk quotients and limiting reference organisms at Blois predicted using Tier 1 of the ERICA Tool and assuming water and sediment inputs estimated by LIDO-TRACER.

Radionuclide	Risk quotient	Limiting reference organism
^{110m}Ag	2.6×10^{-3}	Bivalve mollusc
^{58}Co	1.0×10^{-3}	Insect larvae
^{137}Cs	8.5×10^{-5}	Mammal
Sum	**3.7×10^{-3}**	

3.4.3 *Discussion and comments*

For ^{110m}Ag and ^{137}Cs, risk quotients are in the same order of magnitude for the Beaulieu and Blois stations respectively even though both river water and sediment concentrations were higher at Blois.

For instance for ^{58}Co, for which a benthic organism is the limiting reference organism, risk quotients are about 5 times higher at Blois (which is the same as the difference in water activity concentrations).

3.5 Calculation 4 - Tier 2 assuming mean concentrations in river water for a site with no sedimentation

3.5.1 Input data

Input data for river water at Beaulieu , where LIDO-TRACER predicts that no sedimentation will occur, were the same as those presented in Table 3.3. Two simulations were conducted: (i) sediment concentration was not entered as input data and therefore calculated by the ERICA Tool; (ii) sediment concentrations were entered as zero.

3.5.2 Results

The outputs of the Tier 2 assessment with the inputs as described above are presented in Table 3.7.

Table 3.7 Best estimate risk quotients and limiting reference organisms at Beaulieu predicted using Tier 2 of the ERICA Tool and assuming water and sediment inputs estimated by LIDO-TRACER.

Species	Risk quotient (calculated sediment concentration)	Risk quotient (sediment concentration = 0)
Amphibian	3.9×10^{-5}	3.9×10^{-5}
Benthic fish	3.4×10^{-4}	3.4×10^{-5}
Bird	4.0×10^{-5}	4.0×10^{-5}
Bivalve mollusc	4.7×10^{-4}	2.6×10^{-4}
Crustacean	2.9×10^{-4}	6.8×10^{-5}
Gastropod	3.8×10^{-4}	1.8×10^{-4}
Insect larvae	5.4×10^{-4}	1.0×10^{-4}
Mammal	5.2×10^{-5}	5.2×10^{-5}
Pelagic fish	3.5×10^{-5}	3.5×10^{-5}
Phytoplankton	2.7×10^{-6}	2.7×10^{-6}
Vascular plant	2.5×10^{-4}	3.7×10^{-5}
Zooplankton	5.1×10^{-5}	5.1×10^{-5}

3.5.3 Discussion and comments

For organisms that are assumed to live in sediment or at the water-sediment interface (i.e. benthic fish, bivalve mollusc, crustacean, gastropods, insect larvae, vascular plants), the contribution of sediment contamination to the risk quotient may be significant (e.g. benthic fish, crustacean). Consequently, a good estimation of sediment contamination is important to ensure a realistic degree of conservatism of the ERICA approach. However, in this instance, it should be underlined that calculating doses to benthic organisms when no sediment is present is unlikely to be relevant.

3.6 Calculation 5 - Tier 2 assuming mean concentration in river water and sediments for a site with sedimentation

3.6.1 Input data

Input data for river water and sediment activity concentrations at Blois (where sediments are present) are those presented in Table 3.5. Two simulations were conducted: (i) sediment concentration was not entered as input data and therefore calculated by the ERICA Tool; (ii) sediment concentrations were entered as presented in Table 3.5.

3.6.2 Results

The outputs of the Tier 2assessment with the inputs as described above are presented in Table 3.8 (N.B. risk quotients were calculated taking into account ^{110m}Ag, ^{58}Co and ^{137}Cs only).

Table 3.8 Best estimate risk quotients and limiting reference organisms at Blois predicted using Tier 2 of the ERICA Tool and assuming water and sediment inputs estimated by LIDO-TRACER.

Species	Risk quotient with ERICA Tool calculated sediment concentrations	Risk quotient with LIDO-TRACER calculated sediment concentrations
Amphibian	2.3×10^{-5}	2.3×10^{-5}
Benthic fish	2.2×10^{-4}	3.3×10^{-5}
Bird	2.0×10^{-5}	2.0×10^{-5}
Bivalve mollusc	6.7×10^{-4}	4.6×10^{-4}
Crustacean	3.2×10^{-4}	9.0×10^{-5}
Gastropod	6.3×10^{-4}	3.1×10^{-4}
Insect larvae	5.4×10^{-4}	1.8×10^{-4}
Mammal	4.4×10^{-5}	4.4×10^{-5}
Pelagic fish	2.9×10^{-5}	2.9×10^{-5}
Phytoplankton	2.3×10^{-7}	2.3×10^{-7}
Vascular plant	2.7×10^{-4}	4.7×10^{-5}
Zooplankton	7.0×10^{-5}	7.0×10^{-5}

3.6.3 Discussion and comments

For organisms that are assumed to live in sediment or at the water-sediment interface (*i.e.* benthic fish, bivalve mollusc, crustacean, gastropods, insect larvae, vascular plants), the contribution of sediment contamination to the risk quotient may be significant (*e.g.* benthic fish, crustacean). The calculation of the sediment concentration by the ERICA Tool is, in this specific case, conservative. This example showed that a way to limit undue conservatism in the assessment is to determine (robust) site-specific concentrations in the sediment, by taking into account the hydraulic and morphological characteristics of the investigated river.

3.7 Calculation 6 - Tier 2 assuming mean concentrations in river water and modified database

3.7.1 *Input data*

Input data for river water at Beaulieu were the same as those presented in Table 3.3. Two simulations were conducted: (i) the ERICA default K_d database was used; (ii) K_ds values (best estimate values) were modified according to the database in preparation under the IAEA-EMRAS programme (Durrieu et al., 2006, Ciffroy et al., pers. comm.) to update IAEA (1994). The two sets of K_d values used are compared in Table 3.9.

Table 3.9 ERICA default and proposed EMRAS K_d values used in the simulations.

Radionuclide	ERICA K_d	EMRAS K_d
^{110m}Ag	10^3	$8.5x10^4$
^{58}Co	$5.0x10^3$	$4.3x10^4$
^{60}Co	$5.0x10^3$	$4.3x10^4$
^{137}Cs	10^3	$8.5x10^3$
^{54}Mn	10^3	$1.3x10^5$

3.7.2 *Results*

Risk quotients using Tier 2 of the Tool with the two sets of K_d values are presented in Table 3.10.

Table 3.10 Best estimate risk quotients calculated using ERICA default and proposed EMRAS K_ds values.

Species	Risk quotient with ERICA K_ds and calculated sediment concentration	Risk quotient with EMRAS K_ds and calculated sediment concentration
Amphibian	$1.1x10^{-5}$	$1.1x10^{-5}$
Benthic fish	$2.0x10^{-4}$	$4.4x10^{-3}$
Bird	$1.1x10^{-5}$	$1.1x10^{-5}$
Bivalve mollusc	$4.4x10^{-4}$	$4.9x10^{-3}$
Crustacean	$2.6x10^{-4}$	$5.1x10^{-3}$
Gastropod	$3.6x10^{-4}$	$5.1x10^{-3}$
Insect larvae	$5.1x10^{-4}$	$1.0x10^{-2}$
Mammal	$2.3x10^{-5}$	$2.3x10^{-5}$
Pelagic fish	$1.5x10^{-5}$	$1.5x10^{-5}$
Phytoplankton	$1.6x10^{-7}$	$1.6x10^{-7}$
Vascular plant	$2.3x10^{-4}$	$5.0x10^{-3}$
Zooplankton	$3.5x10^{-5}$	$3.5x10^{-5}$

3.7.3 *Discussion and comments*

For organisms assumed to live completely in the water column, the choice of K_d values does not affect the calculation of risk quotients. However, for organisms living at the water-sediment interface or within the sediment, the K_d values used influenced the resultant risk quotient by up to 20-fold; the proposed EMRAS values resulting in the higher estimates. K_d values in the ERICA database were of the same order of magnitude for Ag, Co, Cs and Mn. However, these elements are known to show different behaviour at the water-particle interface. The K_d values proposed by the EMRAS programme are based on an exhaustive literature review of experimental K_ds, and on a rigorous statistical treatment of these data. By contrast, some values in the ERICA database are not for freshwaters (*e.g.* the K_d for Ag is taken from an old IAEA report (1985) dealing with saltwater). It is recommended that the ERICA K_d database is re-examined taking into account recent work in this area.

3.8 Other comments on the ERICA Tool

- Tier 1, Input values: it is suggested that the user should 'enter the maximum modelled media concentration'. The concept of 'maximum concentration' depends on the time step used for averaging modelled concentrations in water (hourly, daily or annual modelled values). This should be clarified, at least in associated attached document to the ERICA approach.
- Plots: Log-scales would be preferable (if possible, users should be allowed to change scales of plots).
- In Tier 2: Meaning of 'Uncertainty factor' is unclear.
- Tier 2: Radioecology parameters: the colour coding is a good idea, but, without any explanation concerning the signification of these colours, it is not useful. Is it planned to explain what these colours mean?
- Tier2: Effects summary tables: A first range 0-10 $\mu Gy\ h^{-1}$ would be more relevant relative to the screening value. Table not easy to view as effects column is right justified.
- Tier 3: it would be useful to propose a PDF for effect to organisms to provide uncertainty analysis on risk quotients.
- Tier 3: sensitivity analysis: Some outputs were found to be correlated to inputs without any justification (e.g. total dose to gastropods was found to be correlated to CR of pelagic fish (3rd most sensitive parameter)). When several radionuclides and several organisms are chosen for a given simulation, some meaningless results can be obtained because the Tool does not filter parameters that are actually used for a given output.

3.9 References

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4 Sellafield marine

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4.1 Introduction

This case study has been conducted to test the prototype ERICA Tool and the draft D-ERICA guidance in relation to its application in an assessment concerning regulated discharges to the marine environment. WSC and the NRPA have run the Tool as potential regulators assessing the impacts of radioactive liquid discharges from the Sellafield reprocessing site on the organisms that inhabit the Cumbrian coastline and the Western Irish Sea. The ERICA Tool has been used to assess the discharges of radionuclides for two different years to represent discharges from the present day (2005) and those from 1980, when the radioactive discharges were substantially higher. The primary purpose of the study was to assess the functionality of the ERICA Tool and D-ERICA guidance, the consistency of radionuclide transport and uptake models contained within it, and the usefulness of the Tool and its outputs in decision making. It was envisaged that this would be achieved through comparison with output from site-specific models and empirical data sets based on monitoring reports. To achieve this objective, the Marine Environment Advection and Dispersion (MEAD) computer model (Goshawk and Clark, 1999; Smith et al*.*, 2003), which has been developed at WSC specifically to predict the behaviour of radionuclides discharged into the Irish Sea, has been used to calculate concentrations of radionuclides in seawater and sediments resulting from the discharges from Sellafield during 1980 and 2005. The radiation doses to biota can be assessed from these calculated radionuclide concentrations using the ERICA Tool.

This report should not be interpreted as an authoritative assessment of the potential impact on Irish Sea biota of discharges from the Sellafield site. The time to complete this report was highly constrained by project timescales and any authoritative assessment would require substantially more detailed attention than it has been possible to apply to the assessment reported here.

4.2 The Irish Sea

The following section provides relevant information on the assessment and has drawn on the guidance provided in the Tool Help. This information could be used to complete "Problem formulation" text boxes including a detailed description of the assessment and consideration of the transfer pathways and assessment endpoints.

4.2.1 Hydrodynamics of the Irish Sea

The behaviour of radionuclides discharged into the Irish Sea is strongly influenced by water movements within the Irish Sea and therefore a short summary of its hydrodynamics has been provided.

The Irish Sea is a semi-enclosed body of water, bounded by the eastern coast of Ireland and the coasts of Wales, northwest England and southwest Scotland. It connects with the Atlantic Ocean through St. George's Channel in the south and the North Channel, Clyde Sea and Malin Shelf Sea to the north. Water enters the Irish Sea both through St George's Channel and the North Channel, although the former dominates, and leaves via the North Channel.

The eastern Irish Sea between the Isle of Man and the coast of northwest England and north Wales is quite shallow, with water depths not exceeding 50 metres. The dominant feature of the western Irish Sea is a deep trough extending from St George's Channel in the south through to the North Channel.

Water depths here extend to 100-200 metres throughout the trough, with depths exceeding 200 metres in the North Channel itself (Figure 4.1).

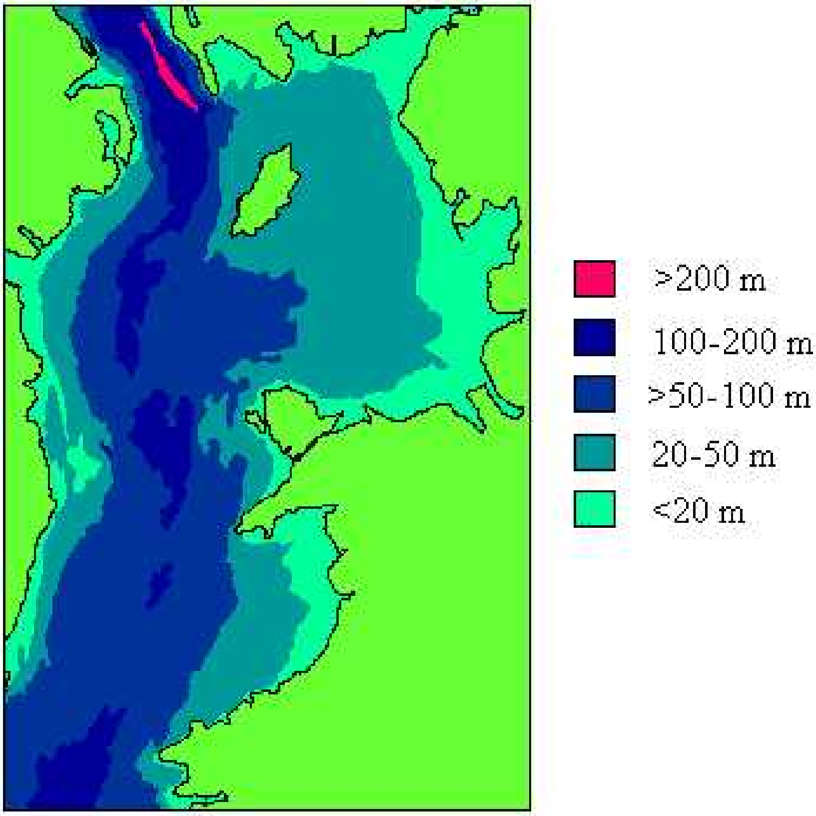


Figure 4.1 Bathymetry of the Irish Sea basin.

Within the Irish Sea, tides are transmitted around the sea in a complex wave-like motion. Although the motion is complex, the behaviour is determined largely by the amplitude and timing of tidal oscillations in the Atlantic and by the bathymetry of the Irish Sea basin. Wind strength and direction will also influence the tides. The tides cause elevations of water levels on a twice-daily cycle, which in turn result in movement of water by creating tidal currents. The direction of tidal currents of course reverses as the tide moves from flow to ebb, so water moved by the tidal currents tends to oscillate, moving forwards and backwards on the twice-daily tidal cycle. However, these forward and backward oscillations do not necessarily balance, leaving a net movement as a result of each tidal cycle - which is much smaller than the amplitude of the tidal oscillation. In addition to the imbalance in tidal oscillations, the effects of the wind on surface water can also induce net movements of seawater. This overall net movement, averaged over many tidal cycles, is known as the residual current and is very important in determining the transport of pollutants in seawater.

Residual current vectors (Goshawk et al., 2003) show clearly the general south to north movement of seawater; the throughput of water through the Irish Sea (*i.e.* the flow rate out through the North Channel) is about $5x10^9$ m^3 d^{-1} and the mean residence time of water in the northern Irish Sea is about one year (Jefferies et al., 1982). The vectors also show areas with low residual currents, notably off the Cumbrian coast, in Liverpool Bay and in Dublin Bay.

It must be emphasised that these residual currents are annually averaged. Sub-annual hydrodynamic processes in the Irish Sea, such as the gyre in the western Irish Sea (Hill et al., 1997) can affect the circulation of seawater in a localised area over a sub-annual timescale. It is important to be aware of such processes when one is attempting to describe seasonal variations. However, when investigating long-term behaviour, the importance of strongly seasonal processes is substantially diminished.

4.2.2 Sedimentology of the Irish Sea

Marine sediments - both those deposited on the seabed, and those suspended in the water column - play an important role in the behaviour of radionuclides in the marine environment.

In relation to radionuclide behaviour, the distribution of bed sediments according to grain size is of particular importance.

Sediment accumulation in particular areas is controlled by the particle size. Sediments are conventionally classified according to particle size: major grain-size classes include Clay (<2 µm particle diameter), Silt (2-60 µm), Fine sand 60-200 µm, Medium sand (200-600 µm) Coarse sand (600-2000 µm) and Gravels (>2000 µm) (BS1377).

The finest sediments are easily suspended in seawater and take some time to settle out, so only accumulate in areas with low current velocities. The two main areas of muddy (silt plus clay) sediments, one extending southwards from St. Bees Head towards Morecambe Bay, and the other lying between the Isle of Man and the northeast coast of Ireland, clearly correlate with the areas of low residual current velocity (see Figure 4.9 for location map).

Some authors have considered that both the main areas of muddy sediment in the Irish Sea are actively accreting whilst others consider that the net accumulation rate is zero or very low; this latter view is the current consensus (Kershaw, 1986; Kershaw et al., 1988). Despite the low overall accumulation rate, the sediment is by no means static. Extensive vertical mixing of bed sediments occurs through the action of biota (Kershaw et al., 1983), principally the Echiuran worm *Maxmülleria lankesteri* and the Thalassinid shrimp *Callinassa subterranea*. Sediment resuspension arises from localised erosion, bioturbation and human activities such as trawling and is balanced, on average, by deposition. These processes have mixed anthropogenic radionuclides and chemical pollutants down to a depth of 0.5 to 1 metre into the seabed.

In the intertidal region both wave energy and current velocities influence sediment distribution. Fine sediments are typically found in estuaries, on saltmarshes (where vegetation is an efficient trap for fine suspended particulate) and in harbours (McDonald and Jones, 1997). Beaches are a much higher energy environment, largely because of wave action, and their sediments are characterised by sands and gravels.

4.3 Discharges from Sellafield

Low level liquid effluents arising from a number of sources on the Sellafield site are discharged to the Irish Sea via pipelines which extend about 2.5 km from the high water mark. The first two such pipelines were laid in 1950.

4.3.1 Historical background to Sellafield site discharges

The historical background to the generation and discharge of low level liquid effluent from Sellafield into the Irish Sea has been described in some detail by Gray et al. (1995). Below is a brief account of this historical background.

The site at Sellafield (formerly Windscale) on the west coast of Cumbria comprises the largest nuclear complex in the UK, associated since the inception of the UK nuclear programme with spent fuel reprocessing, waste management operations and nuclear electricity generation. Discharges of radioactive effluents from the site to the environment have taken place since commencement of operations at the site in 1951. Currently, authorisation to discharge under the Radioactive Substances

Act 1993 is granted to British Nuclear Group Sellafield Limited (BNGSL) by the Environment Agency.

Discharges to the Irish Sea from the site increased from the 1950s to the early to mid 1970s, as the amount of radioactivity handled on the site increased. However, since the mid 1970s many major developments have taken place on the site to upgrade effluent treatment and waste management arrangements, and discharges have in consequence been substantially reduced.

During the operation of the first reprocessing plant, medium active liquors were subject to chemical treatment and delay storage prior to discharge to sea. With the commencement of the second reprocessing plant in 1964, facilities for the concentration of medium active liquor by evaporation were introduced permitting longer delay storage of concentrates prior to discharge. The discharge of medium active concentrate to sea was terminated in 1980; the Enhanced Actinide Removal Plant (EARP), which commenced active commissioning in 1994, treats the backlog of stored concentrates together with continuous arisings of these and other reprocessing effluents. Commissioning of the EARP plant has resulted in further substantial reductions in the discharge of isotopes of plutonium and americium. However, the presence of ^{99}Tc in the stored medium activity concentrates resulted in an increase of discharges of this radionuclide during the period when the backlog has been treated, since the EARP plant process was not designed to remove this radionuclide from the effluent. Since 2003 the discharges of ^{99}Tc have decreased significantly due to the addition of tetraphenylphosphonium bromide as a routine operation.

Historically, certain other salt bearing liquors from reprocessing were discharged to sea after delay storage. Introduction of the salt evaporator in 1985 allowed concentration and storage of these liquors, also pending treatment in EARP, substantially reducing the discharge of various short lived fission products such as ^{106}Ru, ^{95}Zr and ^{95}Nb.

The other major effluent stream is water from the fuel storage ponds. During the early to mid 1970s, the radioactivity content of this stream increased significantly due to the increased storage time and consequent corrosion of Magnox fuel. Temporary measures for removal of radioactivity from pond purge water were introduced in the late 1970's pending operation from May 1985 of the site ion exchange effluent plant (SIXEP).These measures proved highly successful in reducing discharges of ^{90}Sr, ^{134}Cs and ^{137}Cs.

Discharges of plutonium and americium have decreased following operation of a flocculation precipitation treatment facility from the mid 1970s, the termination of discharge of concentrates to sea in 1980, and the commissioning of the Salt Evaporator in 1985, and as explained above, the commissioning of EARP in 1994. Since 2004, the discharges of all actinides have declined due to improved effluent management in Fuel Handling Plant and SIXEP.

In 1988, the THORP Receipt and Storage area was commissioned and the pond water from the storage of oxide fuel in this facility was monitored and discharged directly to sea; it contains only a small proportion of the overall activity discharged to sea from the site. In 1994, the main THORP reprocessing plant commenced operation. THORP was designed to produce much lower effluent discharges than the older Magnox reprocessing plants, and, in general, the contribution on a nuclide-by-nuclide basis of THORP to site discharges of liquid effluent, with the exception of ^{3}H, is small.

4.3.2 Collation of Sellafield discharge data

Four separate sets of discharge figures have been used to calculate the water and sediment concentrations to the Cumbrian Coastline, the Eastern Irish Sea and Western Irish Sea.

- The total discharges from the Sellafield site, based on actual discharges from 1952 to 1980.
- The total discharges from the Sellafield site, based on actual discharges from 1952 to 2005.
- Fifty years discharge at 1980 rates with no prior discharges.

- Fifty years discharge at 2005 rates with no prior discharges.

BNFL Sellafield historic liquid discharge data, from 1952 to 2005, were collated from various sources (Gray et al., 1995; BNFL/BNG Annual Reports on Radioactive Discharges; Jackson et al., 2000; NRPB, 1995) and are shown in Table 4.1.

Historic discharges for the radionuclides ^{95}Zr and ^{95}Nb were provided as a combined total. To enable these radionuclides to be modelled the combined total was split into ^{95}Zr and ^{95}Nb components using a 56% to 44% split as calculated from the relative proportions for the years 1986 to 2005. Any discharges for plutonium-alpha were split into ^{238}Pu and ^{239}Pu components using a 26 % to 74 % split as had been previously applied to the historic data for the years 1993 to 1996.

The data for total discharges from the site reflect clearly the historical developments described in Section 4.3.1, with discharges of most radionuclides rising to a peak between the late 1960s and mid 1970s and subsequently reducing considerably as advances in technology, and substantial investment, allowed the introduction of improved effluent treatment and more efficient process plants on the site.

Table 4.1 Liquid effluent discharges from Sellafield, 1952 to 2005.

	T Bq y^{-1}											
Year	^{3}H	^{60}Co	^{90}Sr	^{95}Zr	^{95}Nb	^{95}Zr+ ^{95}Nb	^{99}Tc	^{106}Ru	^{129}I	^{134}Cs	^{137}Cs	^{144}Ce
1952	250	0	33	88	112	200	8	91	0.02	1.20	46	66
1953	250	0	36	88	112	200	8	180	0.02	1.20	46	66
1954	250	0	19	88	112	200	8	270	0.02	1.20	46	66
1955	250	0	9	106	134	240	8	210	0.02	0.52	21	35
1956	250	0	71	106	134	240	8	1600	0.02	4.10	160	47
1957	250	0	61	118	150	268	8	990	0.02	3.50	140	96
1958	250	0	93	141	179	320	8	1600	0.02	5.70	230	220
1959	250	0	57	246	314	560	8	1300	0.02	2.70	73	260
1960	250	0	19	141	179	320	8	1500	0.02	1.40	34	33
1961	250	0	18	145	185	330	8	930	0.02	1.80	40	80
1962	250	0	38	84	106	190	8	850	0.02	3.90	74	89
1963	140	0	20	62	78	140	8	1200	0.02	6.10	85	52
1964	290	0	36	704	896	1600	8	910	0.02	6.10	100	120
1965	330	0	56	792	1008	1800	8	750	0.02	10	110	140
1966	460	0	34	616	784	1400	8	920	0.02	16	180	250
1967	5180	0	52	748	952	1700	8	640	0.02	15	150	510
1968	507	0	50	1056	1344	2400	8	900	0.02	48	370	370
1969	5660	0	110	1012	1288	2300	8	850	0.02	62	440	500
1970	11700	0	230	312	398	710	40	1000	0.10	220	1200	460
1971	5480	0	460	572	728	1300	40	1400	0.10	240	1300	640
1972	1770	0	560	792	1008	1800	40	1100	0.10	220	1300	500
1973	740	0	280	704	896	1600	40	1400	0.10	170	770	540
1974	1200	0	390	158	202	360	40	1100	0.10	1000	4100	240
1975	1400	0	470	141	179	320	40	760	0.10	1100	5200	210
1976	1200	0	380	150	190	340	40	770	0.13	740	4300	150
1977	910	0	430	130	165	295	40	820	0.11	600	4500	150
1978	1000	1.0	600	101	129	230	179	810	0.07	400	4100	100
1979	1200	0.52	250	70	90	160	43	390	0.12	240	2600	83
1980	1300	0.78	350	70	90	160	57	340	0.24	240	3000	37
1981	2000	0.74	280	145	185	330	5.8	530	0.19	170	2400	17
1982	1800	1.1	320	229	291	520	3.6	420	0.10	140	2000	22
1983	1800	1.7	200	264	336	600	4.4	550	0.20	89	1200	24
1984	1600	1.3	72	207	263	470	4.3	350	0.10	35	430	9.0
1985	1100	2.3	52	18	28	46	1.9	81	0.10	30	330	5.0
1986	2200	1.5	18	8.5	6.2	15	6.6	28	0.12	1.3	18	3.3
1987	1400	1.4	15	8.9	4.5	13	3.6	22	0.10	1.2	12	3.9
1988	1700	0.96	10	5.2	4.6	9.8	4.2	24	0.13	0.95	13	3.2
1989	2100	0.17	9.2	6.5	4.6	11	6.1	25	0.17	1.7	29	3.8
1990	1700	0.17	4.2	4.2	2.6	6.8	3.8	17	0.11	1.2	24	2.0
1991	1800	0.09	4.1	4.2	5.0	12	3.9	19	0.15	0.76	16	1.7
1992	1200	0.07	4.2	7.4	3.3	10	3.2	13	0.07	0.83	15	1.7
1993	2300	0.09	17	7.0	3.4	9.7	6.1	17	0.16	1.2	22	2.5
1994	1700	0.11	29	2.1	1.2	3.3	72	6.7	0.16	0.61	14	0.84
1995	2700	1.3	28	0.34	0.40	0.74	190	7.3	0.25	0.51	12	1.1
1996	3000	0.43	16	0.52	0.63	1.2	150	9.0	0.41	0.27	10	0.78
1997	2560	1.47	37	0.18	0.18	0.36	84	9.8	0.52	0.30	8.0	0.49
1998	2300	2.4	17	0.30	0.35	0.65	53	5.6	0.55	0.32	7.5	0.76
1999	2520	0.89	31	0.10	0.08	0.18	69	2.7	0.48	0.34	9.1	0.60
2000	2300	1.2	20	0.1	0.09	0.19	44	2.7	0.47	0.23	6.9	0.55
2001	2600	1.2	26	0.13	0.14	0.27	79	3.9	0.63	0.48	9.6	0.79
2002	3320	0.89	20	0.17	0.25	0.42	85	6	0.73	0.49	7.7	0.97
2003	3900	0.43	14	0.14	0.16	0.3	37	12	0.55	0.39	6.2	0.88
2004	3200	0.78	18	0.13	0.10	0.23	14	4.4	0.65	0.40	9.7	0.82
2005	1600	0.70	13	0.09	0.07	0.16	7	1.8	0.30	0.16	6	0.54

4.4 Behaviour of discharged radionuclides

Radionuclides discharged into the Irish Sea from Sellafield are distributed and dispersed by the actions of the tides and currents. Dispersal in the longer term (several tidal cycles) is determined largely by the 'residual currents' which represent systematic imbalance in current strength and/or direction between the ebbing and flowing tides.

Some radionuclides (commonly referred to as *conservative* radionuclides), for example ^{90}Sr, ^{99}Tc, ^{137}Cs, remain almost entirely dissolved in seawater and concentrations of these radionuclides in seawater throughout the Irish Sea respond quite rapidly to changes in the levels of discharge from Sellafield. Thus, an increase or decrease, in discharge rates from Sellafield results quite rapidly in a rise, or fall, in concentrations along the Irish coast. However, concentrations along the Irish coast are substantially lower, due to dilution and dispersion, than those along the coast of north-west England and south west Scotland. The majority of the discharged inventory of these radionuclides is removed from the Irish Sea, and dispersed further, by the outward flow of water through the North Channel. Discharges of ^{137}Cs from Sellafield peaked in the early to mid 1970s and have subsequently reduced considerably. Concentrations of radionuclides in seawater have responded rapidly to these changes, as indicated by the distribution contours shown in Figure 4.2.

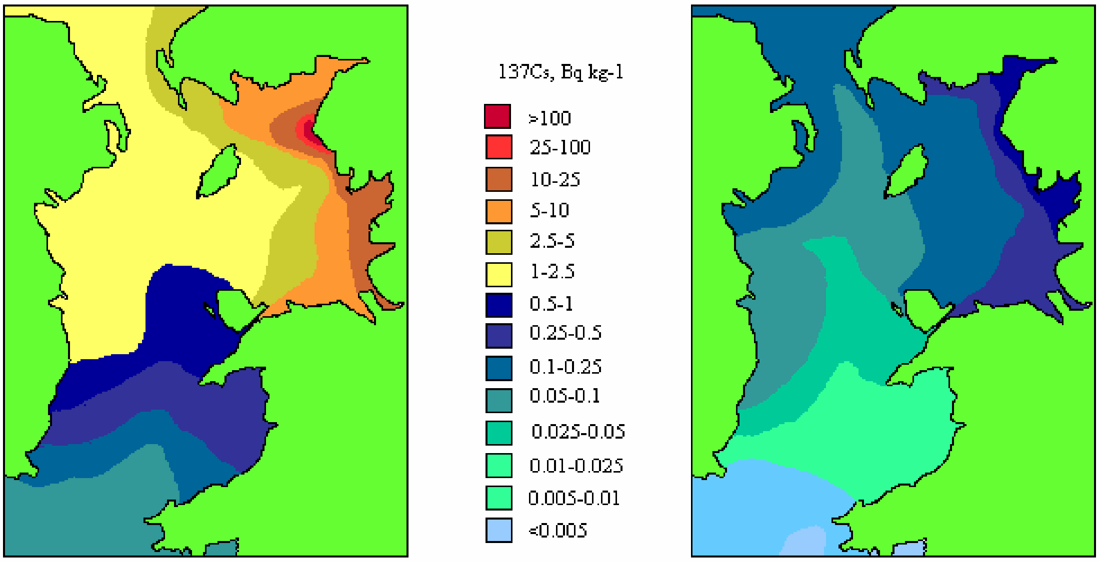


Figure 4.2 Contours of concentrations of ^{137}Cs in the water of the Irish Sea. Data from surveys made by the UK Ministry of Agriculture, Fisheries and Food in 1977 (left) and 1988 (right).

Other radionuclides - in particular, plutonium and americium isotopes - are rapidly adsorbed by suspended sediments and deposited onto the seabed. Transport across the Irish Sea is thereby restricted, and the concentration gradients across the Irish Sea are much greater than for the conservative nuclides referred to above. For these radionuclides, a large proportion of the discharged inventory is retained on seabed sediments within the Irish Sea, particularly in the eastern Irish Sea (Figure 4.3). The deposits of fine grained bed sediments between the Cumbrian coast and the Isle of Man are a significant 'reservoir' or 'sink' for these radionuclides. Through resuspension and desorption of the incorporated plutonium and americium, these bed sediments comprise a continuing and low level, but geographically extended, source of input into seawater and hence marine biota. Consequently, concentrations in the environment respond only slowly to changes in emission rates from Sellafield, whether upwards or downwards. This 'damping' of the response to changes in discharge rate becomes more pronounced with distance from Sellafield.

Marine biota may accumulate radionuclides by: direct absorption from seawater, ingestion, filtration or external adherence of sediment.

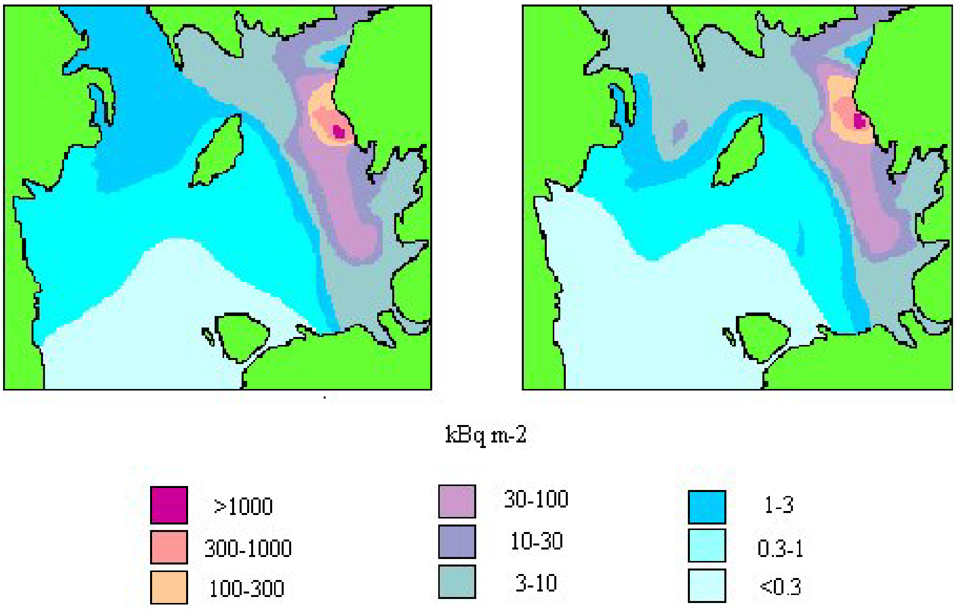


Figure 4.3 Contours of the inventory of plutonium and americium in bed sediments of the Irish Sea. Data from surveys made by the UK Ministry of Agriculture, Fisheries and Food between 1977 and 1979.

4.5 Modelling the distribution and uptake of radionuclides in the Irish Sea

The development and performance of the MEAD model has previously been described by Goshawk and Clarke (1999), Goshawk et al. (2003) and Smith et al. (2003). A brief description of the functions of the model and validation exercises is given below.

The Marine Environment Advection Dispersion (MEAD) model is a hydrodynamic simulation Tool that determines the long-term transport of radioactivity in shelf sea environments. In relation to MEAD, the phrase “long-term” refers to tens of years and to perform simulations on this time scale all quantities used in the model are annually averaged. It is assumed that the radioactivity can be present in the following three phases: dissolved in the seawater, attached to sediment that is suspended in the seawater and attached to sediment on the seabed. Transport is assumed to occur directly in the dissolved and suspended phases through advection and dispersion. In addition, there is indirect transport through chemical and kinematic exchanges between the three phases. The main transport processes represented in MEAD are shown schematically in Figure 4.4.

In the Irish Sea there are many specific issues but three of initial importance for a model are the range of nuclides discharged, the time and space scales of interest and the changing discharge regime from the Sellafield site. Different radionuclides can exhibit a wide range of behaviour, particularly in regard to their interaction with sediment. Some nuclides remain predominantly in the dissolved phase (conservative), such as technetium, whereas others adhere strongly to sediments (non-conservative), such as plutonium. A model must be able to simulate radionuclide transport across this response range requiring a good representation of physical oceanography and sediment interactions. Radionuclide transport models must be able to operate over decades as these are the timescales involved in transport in the Irish Sea. The historical legacy of discharges from BNFL Sellafield creates another interesting scenario. The peak discharges of ^{137}Cs and ^{239}Pu were in the mid 1970's, since then discharges have reduced dramatically to a point now where the seabed of the Irish Sea is, in relative terms, an important source of radionuclides to the water column (Hunt and Kershaw, 1990; Cook et al., 1997). Therefore, to simulate a time series of radionuclide concentrations, a model must be able to simulate the sorption and desorption processes to and from bed sediment.

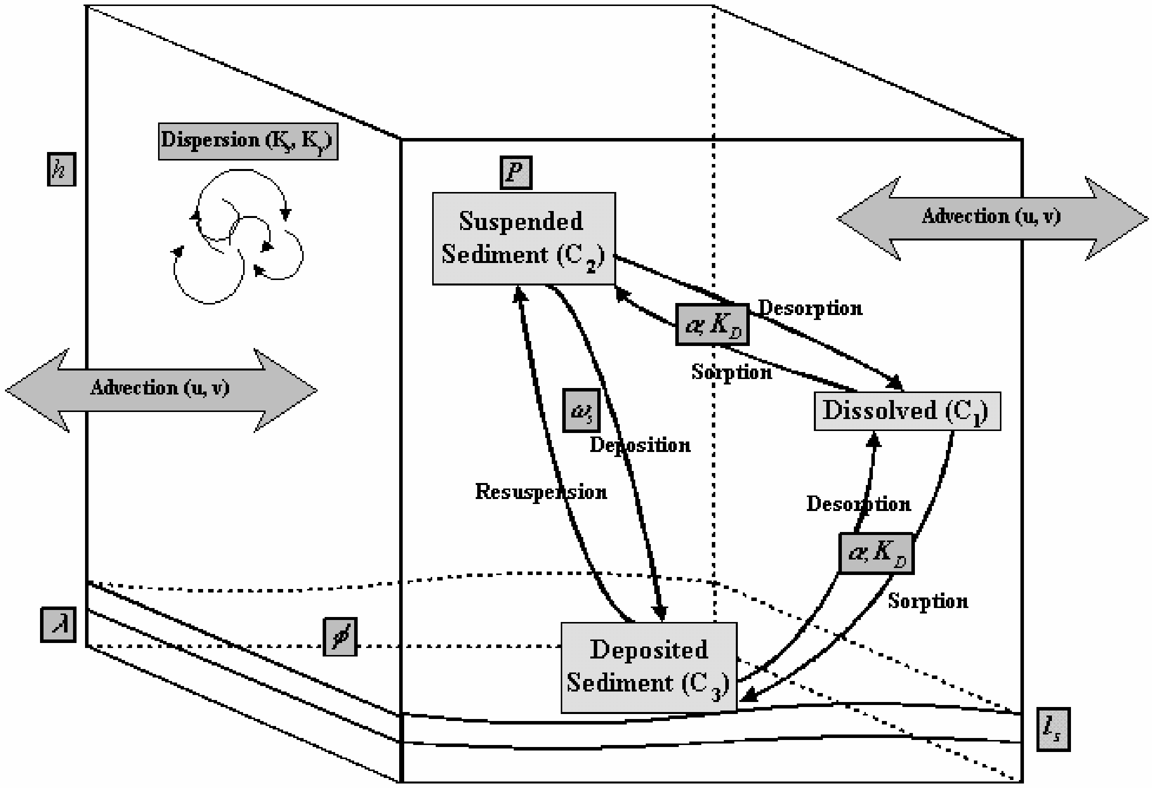


Figure 4.4 The main transport processes represented in MEAD.

A residual current flow field (see Section 4.2.1) has been used for MEAD in preference to a tidally resolving flow field to enable simulations in the order of tens of years to be performed. The flow field used in MEAD was generated using a tidally resolving hydrodynamic model MIKE21 (DHI, 1996). The maximum residual velocity in the flow field used in MEAD is 0.09 m s^{-1} which is consistent with previous work reported by Proctor (1982) where a maximum residual velocity of the order of 0.06 m s^{-1} was derived.

Model validation has been performed using concentrations of ^{137}Cs (conservative) and ^{239}Pu (particle reactive) radionuclides in Irish Sea waters (Smith et al., 2003). The good prediction of seawater concentrations of radionuclides is paramount for the successful performance of MEAD since many subsequent calculations are derived from seawater concentrations (CRs, K_ds).

4.6 Application of the ERICA methodology and Tool

4.6.1 Scenarios modelled

Runs of the MEAD model were executed to simulate radionuclide concentrations in water and sediments for:

a) The principal fishing grounds in the Western Irish Sea (WIS) for the years 1980 and 2005, as a result of discharges from Sellafield up to those times;

b) The waters immediately adjacent the West Cumbrian coast (WC), for the years 1980 and 2005, as a result of discharges from Sellafield up to those times;

c) The waters immediately adjacent the Cumbrian coast, up to the 50th year of continuous discharges at the 1980 rates (with no prior discharges);

d) The waters immediately adjacent the Cumbrian coast, up to the 50th year of continuous discharges at the 2005 rates (with no prior discharges);

e) The principal fishing grounds in the Western Irish Sea up to the 50th year of continuous discharges at the 1980 rates (with no prior discharges);

f) The principal fishing grounds in the Western Irish Sea up to the 50th year of continuous discharges at the 2005 rates (with no prior discharges).

4.7 Testing of the Tools models and parameters

4.7.1 Water concentrations: inter-model comparison

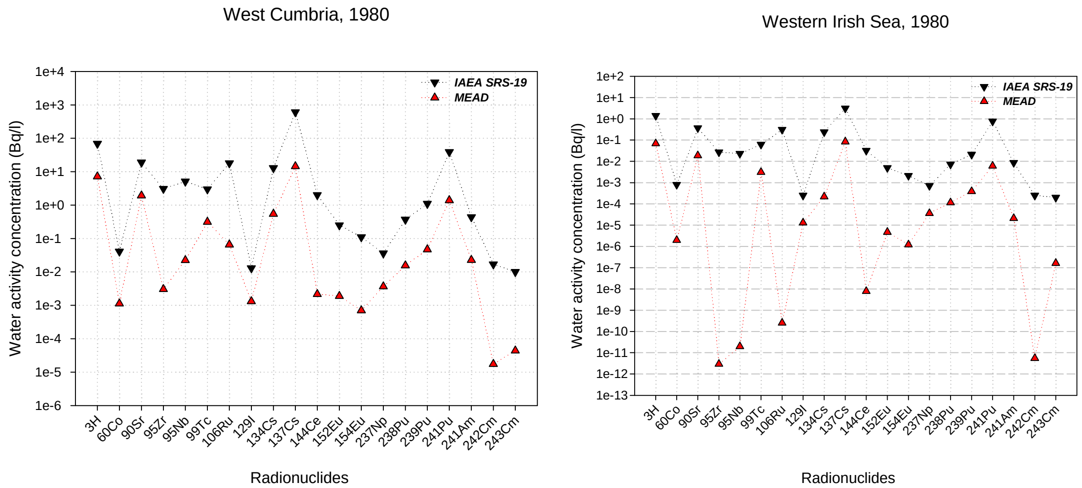
A test was undertaken to compare the results from the default model used in the Tool, i.e. the IAEA SRS-19 (IAEA, 2001) coastal model, with results from the MEAD model described above (Section 4.5). The inter-comparison was conducted at Tier 1 but could equally well have been conducted at Tier 2. The intention behind this approach was to establish whether the default model provided realistic output for a specific case and to identify whether any problems were evident in the default model application. Discharge data for the years 1980 and 2005 (Table 4.1) and appropriate parameters (Table 4.2) have been used to estimate water activity concentrations for sea areas WC and WIS.

Table 4.2 Values used in parameterising the IAEA SRS-19 model at Tier 1.

Parameter	West Cumbria	Western Irish Sea
Water depth (m)	20	20
Distance between release point and shore (m)	2500	2500
Distance between release point and receptor (m)	3500	100000
Coastal current (m/s)	0.015	0.015

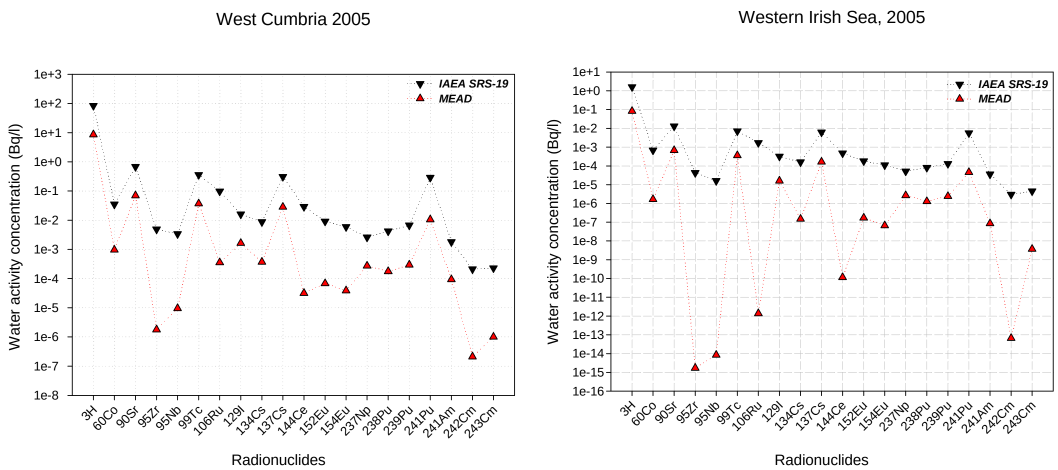
In some cases discharge data are reported as the sum of 2 radioisotopes – a format that is not compatible with Tool data entry options. Therefore, in the case of $^{239+240}Pu$ it was assumed that the total discharge is due to ^{239}Pu only and for $^{243+244}Cm$ that the total discharge rate is due solely to ^{243}Cm.

The output of IAEA SRS-19 model runs for 1980 and 2005 and for sea areas WC and WIS are presented in Figures 4.5 and 4.6. These estimated water activity concentrations have been compared with the corresponding results from the MEAD model (shown on the corresponding figures). The presented results from MEAD have been obtained for a continuous discharge period of 30 years based on the scenarios c-f in Section 4.6.1 above. The selection of a 30 year continuous discharge period is compatible with the discharge period specified by IAEA and therefore the inter-comparison is appropriate.



(a) (b)

Figure 4.5 IAEA SRS-19 and MEAD model outputs using Sellafield discharge data for 1980. The outputs are reported in activity concentrations in water (Bq l^{-1}) for sea area (a) West Cumbria and (b) Western Irish Sea.



(a) (b)

Figure 4.6 IAEA SRS-19 and MEAD model outputs using Sellafield discharge data for 2005. The outputs are reported in activity concentrations in water (Bq l^{-1}) for sea area (a) West Cumbria and (b) Western Irish Sea.

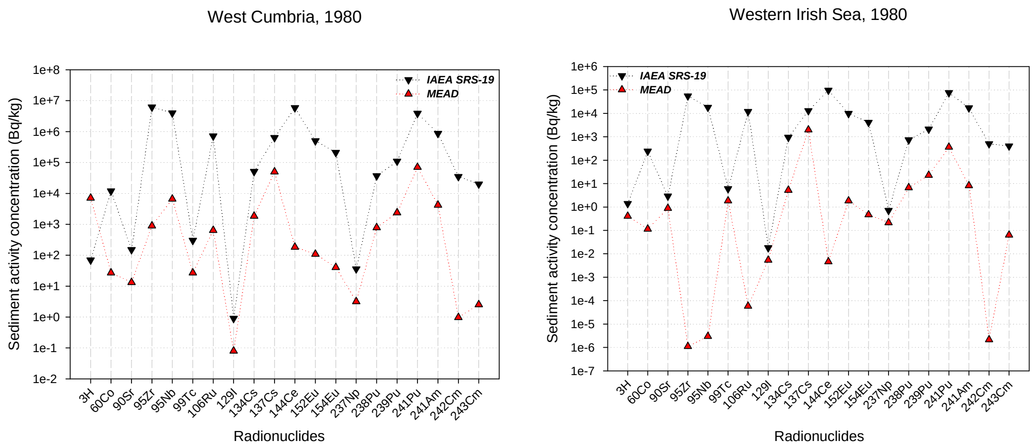
Generally, SRS-19 tends to estimate high water activity concentrations relative to MEAD. Whereas the magnitude of discrepancy in the WC area is within 3 orders of magnitude and, in many cases, much less than this (between 1-2 order of magnitudes for all radionuclides except ^{95}Zr, ^{144}Ce and ^{242}Cm), the difference between model output for the WIS can be substantial. In the worst case the IAEA SRS-19 predicts water activity concentrations that are 10 orders of magnitude higher than the MEAD output.

The large discrepancies in the case of WIS might be attributed to the fact that for this sea area an underlying assumption of the SRS-19 model is not applicable. SRS-19 prompts the user to enter a longitudinal distance, interpreted as a distance between the release point and a potential receptor in the direction of the coastal current. The longitudinal distance was selected to be 100 km for the WIS. This is a good estimation defined purely in terms of the distance between the release point and the receptor but is a poor representation of the corresponding distance if defined along the direction of the coastal current. The residual current vectors show clearly the general south to north movement of seawater (see Section 4.2.1). The intended application of the SRS-19 model appears to be related to distances on a coastal current "downstream" of the discharge point and in this particular case study it is clear that the most appropriate parameter may not have been entered. Furthermore, the mathematical formulation underlying the SRS-19 model does not include longitudinal dispersion, a fact that will also lead to conservative estimates of radionuclide concentrations relative to models that account for this phenomenon.

Comment: There is some question over the applicability of SRS-19 in situations where a distance relates to a source-receptor separation that is not in the direction of the long-shore current. The SRS-19 marine coastal model prompts for a longitudinal distance, interpreted as a distance in the direction of the current. The more complex current patterns observed under natural systems make parameterisation difficult. Consequently, there are reservations over the applicability of the SRS-19 model for long-distances and we recommend that some words to this effect are included in the Help.

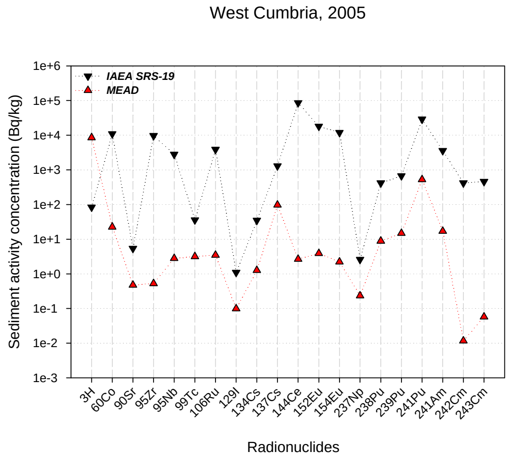
4.7.2 Sediment concentrations: inter-model comparison

A second set of inter-model comparisons were undertaken through analyses of sediment concentration data predicted from water concentrations estimated by the IAEA SRS-19 model and the Tools default K_ds values, and that predicted by the MEAD model. It was necessary to perform the Tool calculations at Tier 2 because predictions of water activity concentrations based on inputs of sediment concentrations are not available at Tier 1. Again, the simulations were made for 1980 and 2005 and for the sea areas WC and WIS using the input data described above (Tables 4.1 and 4.2). Furthermore, the MEAD simulations were conducted at continuous discharge rates at 1980 and 2005 levels for 30 years to allow appropriate inter-comparison. The results from these inter-comparisons are shown in Figures 4.7 and 4.8 for 1980 and 2005 respectively.

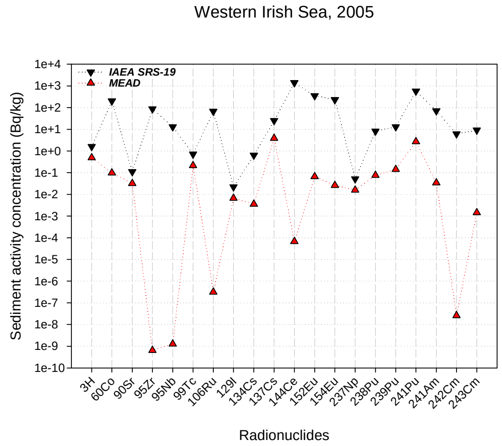


(a) (b)

Figure 4.7 IAEA SRS-19 (in conjunction with Tool default K_ds) and MEAD model outputs using Sellafield discharge data for 1980. The outputs are reported in activity concentrations in sediment (Bq kg^{-1}) for sea area (a) West Cumbria and (b) Western Irish Sea.



(a)



(b)

Figure 4.8 IAEA SRS-19 (in conjunction with Tool default K_ds) and MEAD model outputs using Sellafield discharge data for 2005. The outputs are reported in activity concentrations in sediment (Bq kg^{-1} DW) for sea area (a) West Cumbria and (b) Western Irish Sea.

For the WC, the SRS-19 model, in conjunction with the Tool default K_ds, predicted sediment concentrations that lay less than 4 orders of magnitude above the corresponding output from the MEAD model. This approximately reflects the differences observed during the comparison of water concentrations generated by the two models. One notable exception was ^{3}H for which MEAD predicted a higher activity concentration in sediment for WC than the value predicted by the SRS-19 model, this is presumably due to the choice of K_ds in the two simulations.

The comparison of sediment concentrations from the SRS-19 and MEAD outputs for WIS varies from close correlations in some few examples to a high degree of dissimilarity, reaching in excess of 10 orders of magnitude in the worst cases. An overview of the comparison is provided in Table 4.3.

Table 4.3 Comparison of sediment activity concentration outputs for the IAEA SRS-19 model (+ Tool K_ds) and the MEAD model.

Difference in order of magnitude	WC		WIS	
	1980	2005	1980	2005
1	Sr, Tc, I, Cs, Np	Sr, Tc, I, Cs, Np, Pu	H, Sr, Tc, I, Cs, Np	H, Sr, Tc, I, Cs, Np
2	H, Pu, Am	H, Pu, Am	Pu	Cs, Pu
3	Co, Nb, Ru, Eu*, Cm†	Co, Nb, Ru,	Co, Eu*,	Am, Cm††
4	Zr, Ce, Eu**, Cm††	Zr, Ce, Eu, Cm†	Eu**, Am, Cm††	Co, Eu
>7			Zr, Nb, Ru, Ce,Cm†	Zr, Nb, Ru, Ce, Cm†

* ^{152}Eu, **^{154}Eu, †^{242}Cm, ††^{243}Cm

Looking at this table, it is possible to infer that the major discrepancies between the results of the two models occur for particle-reactive radionuclides. This may reflect the fact that the SRS-19 model does not deal with the effects of sediment adsorption explicitly (for the sake of simplicity and to ensure that conservative estimations are made with respect to activity concentrations in the water column). MEAD, on the other hand, simulates the interaction of radionuclides with suspended and deposited sediments and has been calibrated with Irish Sea site-specific parameters.

The results from the comparison of sediment outputs from the two models support the conclusions of the initial inter-comparison for water that application of SRS-19 to distant locations should be viewed with some caution.

4.7.3 Water concentrations: Tool predicted versus in situ data

A third test was made on the robustness of the prognoses generated by the Tool by comparing water activity concentrations predictions calculated by the Tool, using empirical sediment concentrations as input data, with water activity concentrations measured *in situ*. The test was performed at Tier 2 in the Tool. This is really a test of whether the default K_ds used by the Tool can be appropriately applied in the Irish Sea case study. The simulations were performed for seven locations on the Cumbrian and Lancashire coast for the radionuclides ^{137}Cs, ^{241}Am and ^{239,240}Pu. The predicted water activity concentrations were compared with measurement data extracted from contour maps presented in MAFF (1982) and Kershaw et al. (1992) for ^{137}Cs (1980) and for ^{241}Am and ^{239,240}Pu (1979), respectively. This comparison is shown in Table 4.4.

Table 4.4 A comparison of observed water concentrations (from MAFF (1982) and Kershaw et al. (1992) with predictions using the ERICA Tool Kd values and observed sediment concentrations (from MAFF (1982)).

Location	Radionuclide	Observed sediment concentration (Bq kg^{-1} DW)	Estimated water concentration (Bq l^{-1})	Observed water concentration (Bq l^{-1})
Maryport	^{137}Cs	5700	1.4	3.8
	^{239}Pu	1400	1.4E-02	7.5E-03
	^{241}Am	1300	6.4E-04	1.0E-03
Whitehaven	^{137}Cs	7100	1.8	10
	^{239}Pu	1500	1.5E-02	1.0E-02
	^{241}Am	1200	6E-04	2.5E-03
Newbiggin	^{137}Cs	7700	1.9	2.5
	^{239}Pu	4200	4.2E-02	5.0E-03
	^{241}Am	2800	1.4E-03	5.0E-04
Walney Island	^{137}Cs	2200	0.55	2.5
	^{239}Pu	3.0*	3E-05	5.0E-03
	^{241}Am	800	4E-04	5.0E-04
Heysham	^{137}Cs	2000	0.5	2.5
	^{239}Pu	240	2.4E-03	3.8E-03
	^{241}Am	239	1.2E-04	3.8E-04
Fleetwood	^{137}Cs	150	3.8E-02	2.5
	^{239}Pu	3.0*	3E-05	2.5E-03
	^{241}Am	6.1	3E-06	3.5E-04
Blackpool	^{137}Cs	180	4.5E-02	2.5
	^{239}Pu	5.8	5.8E-05	2.5E-03
	^{241}Am	6.0	3E-06	3.5E-04

* This is an estimated value which is equal to the half of the minimum observed value

Comparison shows that predicted and measurement values lie within one order of magnitude for locations relatively near to the discharge point along the Cumbrian coast (the top three locations in the table). However, analysis of the complete data set, including values from more distant locations, leads to the observation that the maximum differences lie within two orders of magnitude. The Tool tends to underestimate the activity concentration in water (in all cases for ^{137}Cs and numerous cases for ^{239,240}Pu and ^{241}Am) which infers that the default K_ds may be higher than the corresponding site-specific K_ds: water concentrations have been predicted by dividing *in situ* sediment concentrations by the Tool default K_d, predictions more closely corresponding to measurements would have been obtained by using lower K_d values. However, there is some inherent uncertainty associated with empirical water concentrations as these are extracted from concentration contour maps with poor resolution; this may contribute to the discrepancy between predicted and measured values.

4.7.4 Biota concentrations: Tool predicted versus in situ data

The next test on the robustness of Tool predictions concerned the applicability of default CR values for the site-specificity of the Irish Sea. In this case, the input data were selected as water activity concentrations from 1979 and 1980 based on empirical data sets and extracted from appropriate reports, i.e. Cs (MAFF, 1982), Pu and Am (Kershaw et al., 1992). The predictions made by the Tool were compared with activity concentrations in biota from actual measurements in the Irish Sea, i.e. (MAFF, 1982; Woodhead, 1986). Typical sampling locations are demonstrated by the example of mollusc in Figure 4.9. The results from this comparison are shown in Table 4.5.

Figure 4.9 Sampling locations (pink circles) and landing points (blue circles) for molluscs in 1980 from information in MAFF (1982).

In general, it can be concluded that the activity concentrations derived using the Tool are in reasonable agreement with the corresponding values from field measurement, i.e. in many cases the discrepancy is within a factor of 10. In some cases, more substantial differences are observed. For example, in the case of ^{239}Pu and ^{241}Am in benthic and pelagic fish the discrepancy between predicted and measured values exceeds one order of magnitude and in one case is greater than 2 orders of magnitude. For Pu, the difference might be partly explained by the fact that monitoring data were not converted to whole-body equivalent values, rendering them incompatible with the values derived using default CRs where such a correction factor was applied (N.B. a correction factor was not applied to derive CR values for Am because of a lack of appropriate data). The fact that whole-body CRs are considerably higher for Pu than fish muscle CRs may partly account for the Tool over-prediction. The least satisfactory comparison between modelled and monitoring data is observed for ^{137}Cs in seabird for which the predicted value is almost 500 times greater than the measured value. The magnitude of this discrepancy is not easily dismissed as the default radiocaesium CR used in the Tool is based on a fairly large empirical data set (n=70). A possible explanation may lie in the peculiarities of the field data used for this particular comparison which are based on a single species of gull living within an estuarine environment. The transfer via foodstuffs derived predominately from saltmarsh and, presumably, surrounding terrestrial environments might be poorly represented by (ERICA-default) CRs derived for seabirds from more open coastal environments where the link to transfer from sea-water is more prominent. For ^{137}Cs, the Tool tends to slightly over-predict activity concentrations in most cases whereas for ^{239}Pu the Tool, with the exception of benthic and pelagic fish, tends to under-predict *in situ* values.

Table 4.5 Predicted activity concentrations in biota from the ERICA Tool using seawater activity concentrations as input data against measured activity concentrations in biota (primarily edible parts).

Biota	Radionuclide	Estimated activity concentration (Bq kg^{-1} FW) Mean	Observed activity concentration (Bq kg^{-1} FW) Mean	Min.	Max.
Macroalgae	Cs-137	744	326	30	1600
	Pu-239	70	84	56	113
Crustacean	Cs-137	340	280	11	600
	Pu-239	2.1	3	0.5	8
Molluscs	Cs-137	504	230	14	470
	Pu-239	10	50	0.2	150
	Am-241	16	42	0.04	100
Benthic fish	Cs-137	480	365	17	670
	Pu-239	29	0.1	0.003	0.12
	Am-241	0.11	0.05	0.004	0.1
Pelagic fish	Cs-137	676	517	2.6	2000
	Pu-239	36	0.03	0.002	0.08
	Am-241	0.13	0.03	0.002	0.08
Seabird	Cs-137	4600	10	6.7	15
	Pu-239	1.5	2.4		

4.8 Testing of the Add-organism functionality at Tier 2

The *Add organism* function in the Tool was tested by constructing a geometry based on a black-headed gull. The rationale for the selection of this particular organism relates to the fact that a detailed study concerning the activity concentrations of radionuclides in the bodies and environment of these birds allowing whole-body dose rates to be derived, was conducted for the period 1980-1983 (Woodhead, 1986). This provided an excellent opportunity to determine whether the ERICA methodology could be applied to assess dose rates to a specific organism in an equally robust manner to that achieved in the original study and thereafter to evaluate whether results from the two studies correspond.

In constructing the black-headed gull geometry, information was taken from the original study (Woodhead, 1986) where it was stated that the average weight of black-headed gulls was 275 g (range 116 - 390 g for 492 birds). Furthermore, the solid-tissue equivalent ellipsoid for dosimetric modelling was assigned the major dimensions: 5.8 x 8.8 x 11.6 cm. Information in this format is easily entered into the Tool's *Add organism* wizard for the subsequent derivation of dose conversion coefficients

(DCC). In practice, some consideration was afforded the choice of ecosystem, this is a required data entry on the *organism details* dialogue screen. Although the black-headed gull might be strictly considered as being associated with the marine ecosystem, it was deemed more appropriate to assign the gull to the terrestrial ecosystem so that external dose rates could be derived from contamination levels in soil/sediment as oppose to water and (submerged) sediment. Furthermore, the selection of the terrestrial bird allows allocation of an "in-air" occupancy factor, an option that is not available for the marine bird.

Comment – this particular example is illustrative of the fact that it is not always self-evident which ecosystem might be best selected when constructing a new organism geometry. It may be worth adding a few lines of advice in the Help function explaining that the assessor should give some thought to the requirements of the subsequent assessment when parameterising the module for DCC calculations. In cases, for example, where in air geometries are required, a terrestrial ecosystem may be more suitably selected even if the organism under study would be normally considered as occupying marine or freshwater ecosystems. In this case available sediment (or water) concentrations could be used as a proxy for the requested soil concentrations in the terrestrial configuration.

Having calculated the DCCs for the black-headed gull, the new geometry was then selected for the calculation of dose rates using the information in Woodhead (1986) relating to activity concentrations in the gull's body and data from MAFF (1982) for activity concentrations in sediment from Newbiggin in the Ravenglass Estuary for 1980 (Table 4.6). The occupancy factors were also derived from Woodhead (1986) where it was reported that gulls spend approximately two hours each day flying and the remaining period of time is spent on the ground. The height above ground was arbitrarily selected to be 5 m for the in-air configuration.

Table 4.6 Activity concentration for black-headed gulls and their environment.

Sample type	Radionuclide	Activity Concentration (Bq kg^{-1} DW)
gull – whole-body	Cs-137	8.5#
gull – whole-body	Nb-95	1.9#
gull – whole-body	Pu-239,240*	2.4#
silt	Co-60	68
silt	Sb-125	330
silt	Ru-106	11000
silt	Zr-95 + Nb-95**	3000
silt	Ce-144	3500
silt	Cs-134	490
silt	Cs-137	7700
silt	Eu-154	280
silt	Pu-238	960
silt	Pu-239,240	4200
silt	Am-241	2800
silt	Cm-242	48
silt	Cm-243,244***	17

*Assumed to be Pu-239 only; **Assumed activity ratio = 1; ***Assumed to be Cm-243 only; #Fresh weight

Radiation weighting factors were assigned values of unity for both low beta and alpha radiation types to allow comparison with the absorbed dose rates calculated by Woodhead (1986). Furthermore, the internal component of the dose was calculated only for those radionuclides for which direct measurements were available (using the Tool, whole-body concentrations could be derived through the use of CR data for birds if such data were required) again for the sake of compatibility with the earlier study.

The total external dose rate had to be derived manually (the Tool reporting external dose rates individually for each radionuclide).

Comment: There should be a row in the Results, Tables section that reports the total external (and also internal) dose rate.

4.8.1 *Occupancy factors*

Woodhead (1986) provides a quite detailed breakdown of the gull's occupancy at various locations within it's habitat including the period of time spent roosting and feeding over various sediment types (silt/mud, sand and saltmarsh) and time flying. This level of detail is not compatible with allowed input data entry for the Tool which is restricted to occupancy "in air" and "on soil" and to the association of a single contamination level for sediment/soil with these target-source configurations. This limitation might be circumvented by using a modified version of the guidance provided in the "Assessments of adjacent ecosystems" in the Help function. Although it was originally envisaged that this would be applied to assessments where the organism occupied more than one of the ERICA-defined ecosystems, *e.g.* freshwater and terrestrial, it could equally be applied in subdividing areas within individual ecosystems. In this particular case, the calculation could be run first for silt/mud, then for sand followed by saltmarsh entering appropriate occupancies factors and concentrations for these sediment types in turn. The occupancy factors would need to be weighted according to the fraction of time spent within each of these locations ensuring that the total occupancies of the three separate calculation runs did not exceed one. Alternatively, a "weighted" sediment concentration could be derived according to the time spent by the organism in different locations.

Note: Additional comment could be provided within the Help guidance, i.e. the adjacent ecosystem approach can be modified and applied to complex assessments where great detail is provided with regard to the time spent by the organism at different locations within its habitat with different levels of contamination.

Woodhead (1986) used measurements of γ-ray dose rate, i.e. gamma-air kerma rates, modified by a factor of 1.5 to account for the increased exposure of the bird owing to proximity to the substrate in the action of feeding and roosting, instead of deriving these values indirectly from activity concentrations of β,γ-emitting radionuclides in soil through the use of dose conversion coefficients. In cases where gamma-air kerma rate data are available for the study area, it would be quite natural to attempt to use them. A functionality allowing the entry of this type of information, is unfortunately not currently available for the Tool but should be seriously considered for post-ERICA versions of the software.

The total external dose rate was calculated to be approximately 2 $\mu Gy\ h^{-1}$ using the Tool. This compares to 0.83 $\mu Gy\ h^{-1}$calculated by Woodhead (1986). Considering that in providing input data for the Tool, it was assumed that the gull occupied the most contaminated environment represented by silt, whereas, in the calculations made by Woodhead (1986) consideration was given to the time spent in less contaminated environments, the higher estimate provided by the Tool are not surprising. The similarity in estimation suggest that the Tool is providing sensible results for external dose rates.

4.9 Application of the ERICA Tool in a role-play regulatory context

In view of the fact that the site-specific MEAD model provides more realistic predictions of the distribution of radionuclides in the Irish Sea compared to the generic IAEA SRS-19 model, activity concentrations predicted by MEAD were selected as the primary input data for use in the assessment Tool at Tier 2. Initially, scenarios (a) and (b) from Section 4.6.1 were run with MEAD water and sediment concentrations as the input data.

For the Western Irish Sea (scenario 4.6.1 a), both the ‘best estimate’ and ‘conservative’ risk quotients, based on the selected dose rate screening value of 10 μGy h^{-1} recommended by ERICA Integrated Approach, fell below unity for both 1980 and 2005 environmental concentrations. On this basis it can be concluded that impacts on biota are negligible for this particular area under the specified conditions. From a regulatory perspective, the level of environmental protection (accepting the use of the ERICA criteria) would be deemed satisfactory and the requirement for further analyses (by moving, for example, to the next tier) removed. In line with this conclusion, the assessment was not taken further for this sea area.

For the Cumbrian coast (scenario 4.6.1 b) at 1980 concentrations, six of the ten reference organisms were found to have a ‘conservative’ risk quotient in excess of unity and four a ‘best estimate’ risk quotient in excess of unity. For the 2005 concentrations phytoplankton alone showed risk quotients in excess of unity, for both ‘conservative and ‘best estimate’ values (best estimate dose rate 22 μGy h^{-1}). Water and sediment input data for these runs are given in Table 4.7, and the Tier 2 assessment results in Table 4.8.

As a next step, available measured environmental concentrations for 1980 (MAFF, 1982) were entered to increase the realism of the assessment (Table 4.9). This did not alter the general result, risk quotients exceeding unity for numerous reference organisms. Indeed, for some organisms the risk quotients increased, due to higher concentrations of some radionuclides in sediments (Table 4.10). Finally, calibrated concentration ratios based on seawater concentrations from the 1980 model runs and environmental measurements for sediments and biota were entered in the Tool to improve the assessment of the 2005 environmental concentrations (Table 4.11). Again, this did not alter the outcome, phytoplankton alone exceeded the risk quotient, with the best estimate dose rate being unaltered (as there are no available measured concentrations of radionuclides in phytoplankton).

Note: A more comprehensive assessment would have used a whole time series of environmental data to derive empirical CRs, rather than relying on figures from a single year.

Finally, it was recognised that environmental concentrations in 1980 may not have been in equilibrium with discharge rates at that time, as discharges in the mid 1970s had been substantially higher than those in 1980. If a regulator had been considering whether 1980 discharge rates were acceptable, the environmental concentrations resulting from discharges at those rates into the future would have been a relevant consideration. Therefore, the MEAD predictions for concentrations in the 50th year of discharges at the 1980 rates, using empirical CRs derived as above, were used as inputs to a prospective assessment. A similar assessment was made for continuous discharges at the 2005 rates. For discharges at the 1980 rates, the results were similar to those for the actual 1980 environmental concentrations. As such they do not greatly assist with the ‘simulated decision making’ discussed below. For continuous discharges at the 2005 rates, risk quotients were below unity in all cases, with doses to phytoplankton being reduced by an order of magnitude to c. 2 μGy h^{-1}.

Table 4.7 MEAD predicted water and sediment concentrations for the Cumbrian coast.

Radionuclide	1980, based on historic discharges to that time		2005, based on historic discharges to that time	
	Water Bq l^{-1} dissolved phase	Bed sediment Bq kg^{-1} DW	Water Bq l^{-1} dissolved phase	Bed Sediment Bq kg^{-1} DW
^{3}H	7.13E+00	6.12E+00	8.80E+00	7.55E+00
^{60}Co	7.53E-04	6.43E+00	1.15E-03	9.86E+00
^{90}Sr	1.92E+00	1.32E+01	7.15E-02	4.91E-01
^{95}Zr	1.14E-03	6.37E+01	1.73E-06	9.65E-02
^{95}Nb	5.73E-03	1.28E+02	6.86E-06	1.53E-01
^{99}Tc	3.14E-01	2.69E+01	3.98E-02	3.41E+00
^{106}Ru	9.82E-02	3.28E+03	9.12E-04	3.04E+01
^{129}I	1.31E-03	7.89E-02	1.68E-03	1.01E-01
^{134}Cs	7.22E-01	2.48E+03	6.34E-04	2.19E+00
^{137}Cs	1.41E+01	4.87E+04	2.43E-01	8.12E+02
^{144}Ce	3.31E-03	2.80E+02	3.75E-05	3.16E+00
^{152}Eu	1.13E-03	6.41E+01	8.25E-05	4.71E+00
^{154}Eu	1.96E-03	1.13E+02	6.10E-05	3.48E+00
^{237}Np	2.81E-03	2.42E+00	2.87E-04	2.46E-01
^{238}Pu	1.83E-02	1.59E+02	6.55E-04	5.55E+00
^{239}Pu	3.64E-02	3.15E+02	1.36E-03	1.14E+01
^{240}Pu	3.64E-02	3.15E+02	1.36E-03	1.14E+01
^{241}Pu	2.33E+00	2.01E+04	3.63E-02	3.09E+02
^{241}Am	1.38E-02	8.01E+02	1.46E-03	8.24E+01
^{242}Cm	1.79E-05	1.00E+00	2.38E-07	1.33E-02
^{243}Cm	2.22E-05	1.26E+00	2.96E-06	1.69E-01
^{244}Cm	2.17E-05	1.23E+00	2.44E-06	1.40E-01

Table 4.8 Tier 2 results for total dose (µGy h^{-1}), based on 1980 scenario with MEAD water and sediment activity concentration data as inputs.

Radionuclide	Benthic fish	Crustacean	Mammal	Pelagic fish	Phytoplankton	Zooplankton	(Wading) bird	Benthic mollusc	Macroalgae	Polychaete worm
H-3	5.9E-05	5.9E-05	5.9E-05	5.9E-05	8.3E-05	5.9E-05	5.9E-05	5.9E-05	5.9E-05	5.9E-05
Co-60	4.9E-03	4.5E-03	2.9E-04	8.0E-04	1.1E-04	2.0E-04	9.1E-05	4.9E-03	4.6E-03	9.5E-03
Sr-90	2.7E-02	1.6E-02	1.8E-03	2.7E-02	5.0E-02	3.3E-03	1.8E-03	1.3E-01	3.8E-02	1.9E-03
Zr-95	1.2E-02	1.2E-02	2.9E-05	1.1E-05	1.9E-03	1.6E-03	1.3E-05	1.4E-02	1.4E-02	2.7E-02
Nb-95	2.6E-02	2.5E-02	1.3E-04	3.7E-05	1.5E-04	3.5E-03	4.4E-05	2.8E-02	2.8E-02	5.5E-02
Tc-99	5.7E-04	4.0E-01	4.4E-04	5.7E-04	7.1E-05	1.8E-03	5.7E-04	1.6E-01	5.4E-01	4.0E-01
Ru-106	3.6E-01	2.9E-01	2.3E-03	2.0E-03	1.2E+00	6.8E-01	2.0E-03	5.6E-01	8.6E-01	1.2E+00
I-129	6.2E-07	5.6E-07	5.5E-08	2.6E-07	5.6E-05	1.8E-04	5.7E-08	1.4E-06	2.6E-04	1.9E-06
Cs-134	1.0E+00	9.9E-01	9.0E-02	1.2E-02	5.5E-03	7.3E-03	7.4E-02	1.1E+00	1.1E+00	2.2E+00
Cs-137	7.5E+00	7.2E+00	9.8E-01	2.2E-01	1.3E-01	1.9E-01	1.2E+00	7.9E+00	8.5E+00	1.6E+01
Ce-144	1.6E-02	1.7E-02	3.1E-04	2.8E-04	3.0E-02	5.0E-03	2.9E-04	2.6E-02	4.8E-02	6.3E-02
Eu-152	2.0E-02	2.0E-02	2.7E-04	1.2E-04	9.1E-03	6.6E-04	1.3E-04	2.2E-02	2.2E-02	4.4E-02
Eu-154	3.8E-02	3.8E-02	4.8E-04	2.1E-04	1.5E-02	1.0E-03	2.3E-04	4.2E-02	4.2E-02	8.3E-02
Np-237	9.3E-05	7.7E-03	3.1E-05	7.7E-05	1.1E-02	1.3E-03	3.1E-04	3.2E-02	4.1E-03	3.3E-02
Pu-238	2.1E+00	9.4E-02	1.6E-01	2.1E+00	7.0E+01	4.6E+00	8.8E-02	6.4E-01	2.4E+00	8.8E-01
Pu-239	3.8E+00	1.8E-01	3.1E-01	3.8E+00	1.3E+02	8.5E+00	1.6E-01	1.2E+00	4.5E+00	1.6E+00
Pu-240	3.8E+00	1.8E-01	3.1E-01	3.8E+00	1.3E+02	8.5E+00	1.6E-01	1.2E+00	4.5E+00	1.6E+00
Pu-241	6.6E-02	3.0E-03	5.3E-03	6.6E-02	2.3E+00	1.5E-01	2.8E-03	2.1E-02	7.7E-02	2.8E-02
Am-241	3.0E-02	5.7E-01	1.2E-01	2.5E-02	9.2E+01	1.8E+00	6.6E-02	3.6E+00	3.7E-01	3.6E+00
Cm-242	6.3E-05	8.2E-04	1.8E-04	6.3E-05	1.7E-01	4.9E-03	9.4E-05	2.0E-02	7.5E-03	9.4E-04
Cm-243	1.2E-04	9.9E-04	2.1E-04	7.3E-05	2.0E-01	5.8E-03	1.1E-04	2.4E-02	8.9E-03	1.2E-03
Cm-244	7.2E-05	9.3E-04	2.0E-04	7.2E-05	1.9E-01	5.6E-03	1.1E-04	2.3E-02	8.6E-03	1.1E-03
Total	**1.9E+01**	**1.0E+01**	**2.0E+00**	**1.0E+01**	**4.3E+02**	**2.4E+01**	**1.8E+00**	**1.7E+01**	**2.3E+01**	**2.8E+01**
Conservative RQ	5.6E+00	3.0E+00	5.9E-01	3.0E+00	1.3E+02	7.3E+00	5.4E-01	5.0E+00	6.9E+00	8.5E+00

Table 4.9 1980 measured sediment (Bq kg^{-1} DW) and organism (Bq kg^{-1} FW) activity concentration data used to calibrate assessments by modifying CR values.

Radionuclide	Sediment	Benthic Fish	Crustacean	Pelagic Fish	Wading Bird	Benthic Mollusc	Macroalgae
H-3							
Co-60	68		0.2			8.3	15
Sr-90			6.5			39.9	17
Zr-95			6.65			166	70
Nb-95			6.65		1.9	166	170
Tc-99							
Ru-106	11000		170				4367
I-129							
Cs-134	490	36.5	31	61.6		25.7	54.3
Cs-137	7700	618	430	912	8.5	340	613
Ce-144	3500		36			133	89.5
Eu-152							
Eu-154	280		1.0			5.7	0.7
Np-237							
Pu-238	960	0.01	2.3	0.02		25.8	23.0
Pu-239	2100	0.03075	4.15	0.041	1.2	47.9	42.3
Pu-240	2100	0.03075	4.15	0.041	1.2	47.9	42.3
Pu-241							
Am-241	2800	0.06		0.08		65.6	
Cm-242	48	0.0005	0.4	0.0013		0.9	0.7
Cm-243	8.5	0.00016	0.049	0.00015		0.29	0.15
Cm-244	8.5	0.00016	0.049	0.00015		0.29	0.15

Table 4.10 Tier 2 results for total dose rates (μGy h[-1]), based on 1980 scenario with actual measured environmental concentrations.

Radionuclide	Benthic fish	Crustacean	Mammal	Pelagic fish	Phytoplankton	Zooplankton	(Wading) bird	Benthic mollusc	Macroalgae	Polychaete worm
H-3	5.9E-05	5.9E-05	5.9E-05	5.9E-05	8.3E-05	5.9E-05	5.9E-05	5.9E-05	5.9E-05	5.9E-05
Co-60	4.5E-02	4.4E-02	2.9E-04	8.0E-04	1.1E-04	2.0E-04	9.1E-05	4.8E-02	4.9E-02	9.6E-02
Sr-90	2.7E-02	4.3E-03	1.8E-03	2.7E-02	5.0E-02	3.3E-03	1.8E-03	2.4E-02	9.2E-03	1.9E-03
Zr-95	1.2E-02	1.3E-02	2.9E-05	1.1E-05	1.9E-03	1.6E-03	1.3E-05	2.7E-02	1.8E-02	2.7E-02
Nb-95	2.6E-02	2.6E-02	1.3E-04	3.7E-05	1.5E-04	3.5E-03	1.7E-04	3.5E-02	3.4E-02	5.5E-02
Tc-99	5.7E-04	4.0E-01	4.4E-04	5.7E-04	7.1E-05	1.8E-03	5.7E-04	1.6E-01	5.4E-01	4.0E-01
Ru-106	1.2E+00	1.0E+00	2.3E-03	2.0E-03	1.2E+00	6.8E-01	2.0E-03	1.6E+00	4.7E+00	3.8E+00
I-129	6.2E-07	5.6E-07	5.5E-08	2.6E-07	5.6E-05	1.8E-04	5.7E-08	1.4E-06	2.6E-04	1.9E-06
Cs-134	2.1E-01	2.0E-01	9.0E-02	1.2E-02	5.5E-03	7.3E-03	7.4E-02	2.2E-01	2.3E-01	4.5E-01
Cs-137	1.3E+00	1.2E+00	9.8E-01	1.7E-01	1.3E-01	1.9E-01	5.6E-03	1.3E+00	1.4E+00	2.9E+00
Ce-144	1.9E-01	1.4E-01	3.1E-04	2.8E-04	3.0E-02	5.0E-03	2.9E-04	3.5E-01	6.0E-01	7.1E-01
Eu-152	2.0E-02	2.0E-02	2.7E-04	1.2E-04	9.1E-03	6.6E-04	1.3E-04	2.2E-02	2.2E-02	4.4E-02
Eu-154	9.3E-02	9.0E-02	4.8E-04	2.1E-04	1.5E-02	1.0E-03	2.3E-04	9.9E-02	1.0E-01	2.0E-01
Np-237	9.3E-05	7.7E-03	3.1E-05	7.7E-05	1.1E-02	1.3E-03	3.1E-04	3.2E-02	4.1E-03	3.3E-02
Pu-238	5.7E-04	7.4E-02	1.6E-01	6.4E-04	7.0E+01	4.6E+00	8.8E-02	8.3E-01	7.4E-01	8.8E-01
Pu-239	1.0E-03	1.3E-01	3.1E-01	1.2E-03	1.3E+02	8.5E+00	3.6E-02	1.4E+00	1.3E+00	1.6E+00
Pu-240	1.1E-03	1.3E-01	3.1E-02	1.2E-03	1.3E+01	8.5E-01	3.6E-02	1.4E+00	1.3E+00	1.7E-01
Pu-241	6.6E-02	3.0E-03	5.3E-03	6.6E-02	2.3E+00	1.5E-01	2.8E-03	2.1E-02	7.7E-02	2.8E-02
Am-241	1.9E-02	5.8E-01	1.2E-01	2.5E-03	9.2E+01	1.8E+00	6.6E-02	2.1E+00	3.9E-01	3.6E+00
Cm-242	2.3E-05	1.4E-02	1.8E-04	4.6E-05	1.7E-01	4.9E-03	9.4E-05	3.2E-02	2.5E-02	9.7E-04
Cm-243	2.9E-04	1.9E-03	2.1E-04	4.8E-06	2.0E-01	5.8E-03	1.1E-04	1.0E-02	5.3E-03	1.7E-03
Cm-244	6.3E-06	1.6E-03	2.0E-04	4.8E-06	1.9E-01	5.6E-03	1.1E-04	9.7E-03	5.0E-03	1.1E-03
Total	**3.2E+00**	**4.1E+00**	**1.7E+00**	**2.8E-01**	**3.1E+02**	**1.7E+01**	**3.1E-01**	**9.8E+00**	**1.1E+01**	**1.5E+01**
Conservative RQ	9.5E-01	1.2E+00	5.1E-01	8.5E-02	9.3E+01	5.0E+00	9.4E-02	2.9E+00	3.4E+00	4.5E+00

Table 4.11 Tier 2 results for total dose rate (μGy h^{-1}), based on 2005 scenario with CRs calibrated to 1980 measurement data.

Radionuclide	Benthic fish	Crustacean	Mammal	Pelagic fish	Phytoplankton	Zooplankton	(Wading) bird	Benthic mollusc	Macroalgae	Polychaete worm
H-3	7.3E-05	7.3E-05	7.3E-05	7.3E-05	1.0E-04	7.3E-05	7.3E-05	7.3E-05	7.3E-05	7.3E-05
Co-60	7.5E-03	6.5E-03	4.5E-04	1.2E-03	1.7E-04	3.1E-04	1.4E-04	8.1E-03	8.5E-03	1.5E-02
Sr-90	1.0E-03	1.6E-04	6.7E-05	1.0E-03	1.9E-03	1.2E-04	6.6E-05	8.8E-04	3.4E-04	7.2E-05
Zr-95	1.9E-05	1.9E-05	4.3E-08	1.7E-08	2.9E-06	2.5E-06	1.9E-08	2.8E-05	2.3E-05	4.1E-05
Nb-95	3.1E-05	3.0E-05	1.6E-07	4.4E-08	1.7E-07	4.2E-06	9.3E-08	3.7E-05	3.7E-05	6.6E-05
Tc-99	7.2E-05	5.1E-02	5.5E-05	7.2E-05	9.0E-06	2.2E-04	7.2E-05	2.1E-02	6.8E-02	5.1E-02
Ru-106	3.4E-03	3.7E-03	2.1E-05	1.8E-05	1.1E-02	6.3E-03	1.9E-05	5.2E-03	2.6E-02	1.1E-02
I-129	7.9E-07	7.2E-07	7.0E-08	3.3E-07	7.2E-05	2.4E-04	7.3E-08	1.8E-06	3.3E-04	2.4E-06
Cs-134	9.0E-04	8.7E-04	7.9E-05	9.6E-06	4.8E-06	6.4E-06	5.7E-07	9.6E-04	9.9E-04	1.9E-03
Cs-137	1.2E-01	1.2E-01	1.7E-02	3.4E-03	2.2E-03	3.3E-03	9.6E-05	1.3E-01	1.4E-01	2.7E-01
Ce-144	1.8E-04	4.0E-04	3.5E-06	3.2E-06	3.4E-04	5.6E-05	3.3E-06	1.2E-03	9.7E-04	7.1E-04
Eu-152	1.4E-03	1.4E-03	2.0E-05	8.6E-06	6.7E-04	4.8E-05	9.7E-06	1.6E-03	1.6E-03	3.2E-03
Eu-154	1.2E-03	1.1E-03	1.5E-05	6.6E-06	4.6E-04	3.2E-05	7.2E-06	1.3E-03	1.3E-03	2.5E-03
Np-237	9.5E-06	7.9E-04	3.2E-06	7.9E-06	1.1E-03	1.4E-04	3.2E-05	3.3E-03	4.2E-04	3.3E-03
Pu-238	1.8E-05	2.5E-03	5.9E-03	2.5E-05	2.5E+00	1.6E-01	6.9E-04	2.8E-02	2.5E-02	3.1E-02
Pu-239	3.5E-05	4.8E-03	1.1E-02	4.8E-05	4.9E+00	3.2E-01	1.3E-03	5.5E-02	4.9E-02	6.1E-02
Pu-240	3.5E-05	4.8E-03	1.1E-02	4.8E-05	4.9E+00	3.2E-01	1.3E-03	5.5E-02	4.9E-02	6.1E-02
Pu-241	3.8E-07	3.5E-05	8.2E-05	3.4E-07	3.5E-02	2.3E-03	9.6E-06	3.9E-04	3.5E-04	4.4E-04
Am-241	7.0E-04	6.6E-04	1.3E-02	2.6E-04	9.7E+00	1.9E-01	6.9E-03	2.2E-01	3.9E-02	3.8E-01
Cm-242	1.2E-07	8.1E-05	2.3E-06	2.4E-07	2.3E-03	6.5E-05	1.3E-06	2.1E-04	1.5E-04	1.3E-05
Cm-243	6.9E-06	9.5E-04	2.8E-05	2.8E-06	2.7E-02	7.7E-04	1.5E-05	2.5E-03	1.8E-03	1.6E-04
Cm-244	1.1E-06	7.8E-04	2.3E-05	2.3E-06	2.2E-02	6.3E-04	1.2E-05	2.0E-03	1.5E-03	1.2E-04
Total	**1.4E-01**	**2.0E-01**	**5.9E-02**	**6.2E-03**	**2.2E+01**	**1.0E+00**	**1.1E-02**	**5.4E-01**	**4.1E-01**	**8.9E-01**
Conservative RQ	4.2E-02	6.0E-02	1.8E-02	1.9E-03	6.6E+00	3.0E-01	3.3E-03	1.6E-01	1.2E-01	2.7E-01

4.10 Simulated decision making - how might these results have been used in a 'real' assessment?

4.10.1 1980 concentrations

The results for 1980 environmental concentrations would, clearly, present a decision maker with some difficulties. Many organisms show conservative risk quotients in excess of one, and in many cases have 'best estimate' dose rates in excess of 10 µGy h^{-1}. The ERICA methodology is therefore indicating that the possibility of adverse effects on biota cannot be excluded. The methodology then guides the user into trying to determine what sort of effects (if any) may occur in the various organism types, and judge whether those may be acceptable. This may be done by referring to the effects 'look-up' tables in the Tier 2 results, or proceeding to a Tier 3 assessment.

The effects summaries at Tier 2 prove unhelpful for a number of reasons. Two examples are discussed here.

For benthic fish, the look-up table shows a substantial number of entries for the relevant dose range of 0-50 µGy h^{-1} (Figure 4.10). There is no overall summary of the expected effects for this range, and many of the summaries are either less than clear or potentially conflicting with one another.

Note: We have found no way to save or print out the effects summaries at Tier 2 or Tier 3 – this further hinders their usefulness.

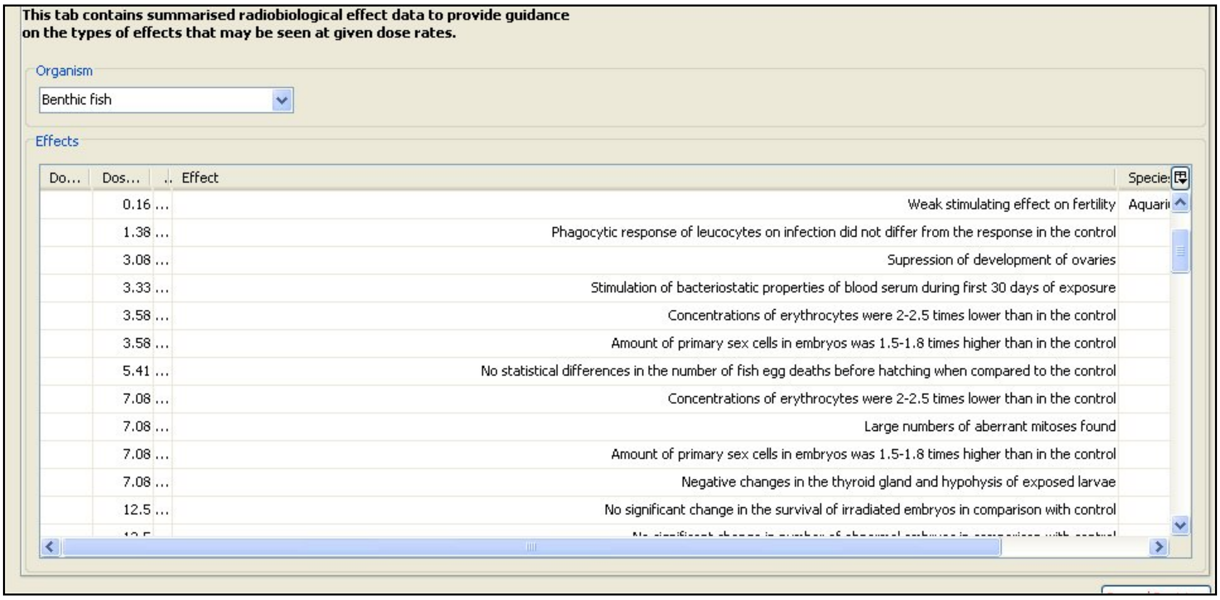


Figure 4.10 Example of effects summaries provided at Tier 2 for benthic fish, dose rate range 0-50 µGy h^{-1}.

For phytoplankton, the effects look-up table shows only one entry, for a dose rate substantially lower than that which is of interest, and implying there are beneficial effects at this low dose rate (Figure 4.11).

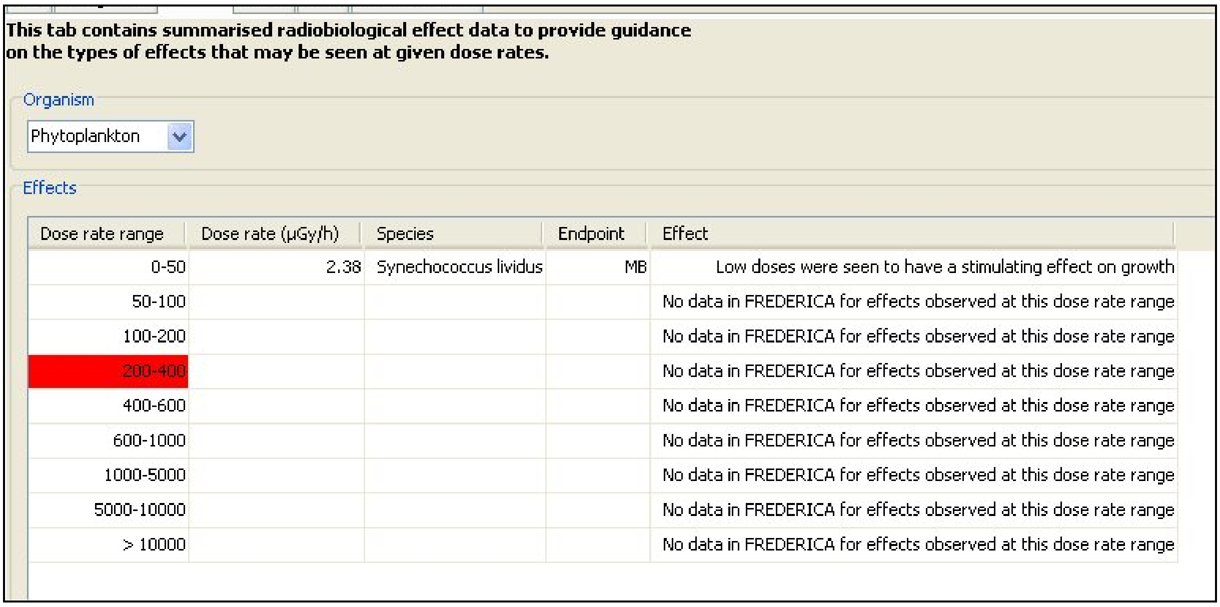


Figure 4.11 Example of effects summaries provided at Tier 2 for phytoplankton, dose rate range 200-400 µGy h^{-1}.

Proceeding to Tier 3 does not assist further. The effect (using uncertainties on CRs only) is to somewhat reduce the 95th percentiles of organism doses and on that basis some may be judged to have a conservative (compared to Tier 2 definition) risk quotient less than one, but not in sufficient numbers to change the overall impression. Looking at effects, by accessing the FREDERICA database, now produces, for example, summaries of 51 relevant effects studies for benthic fish (Figure 4.12). Clearly, considerable time and expertise would be needed to review all of these and it is uncertain whether any clear or convincing picture of the expected effects, at the dose rates of interest, would emerge.

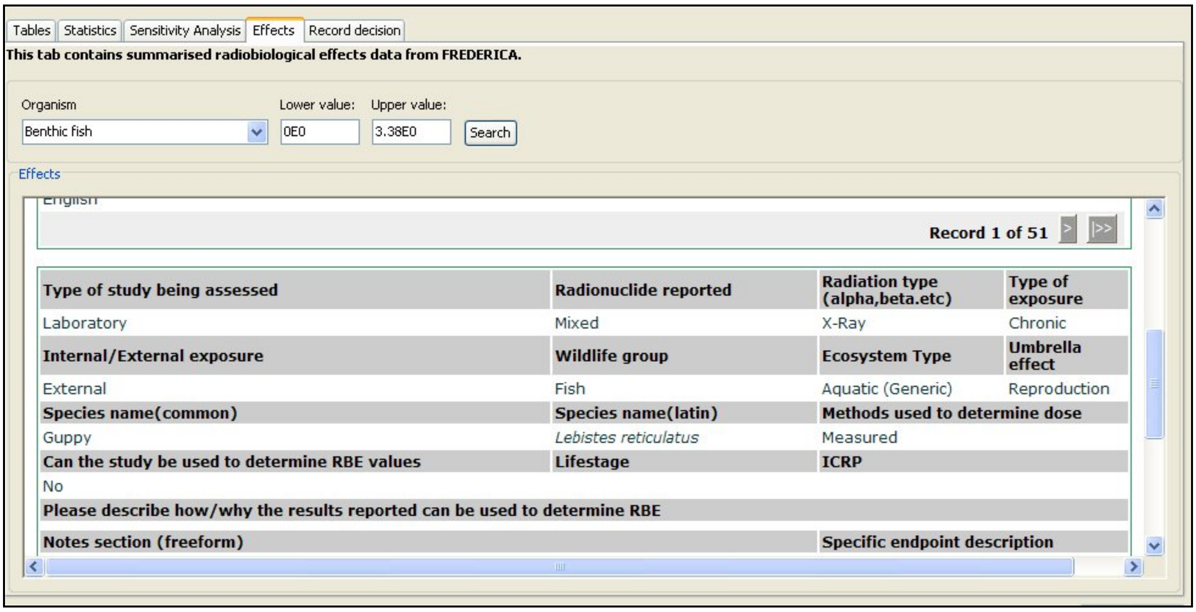


Figure 4.12 Example of effects summary provided at Tier 3 for benthic fish, 1980 scenario with calibrated CRs.

If the assessment is re-run with optional screening dose rates of 40 µGy h^{-1} for terrestrial animals and 400 µGy h^{-1} for terrestrial plants and all aquatic species provided within the ERICA Tool, all risk quotients would be less than one and (on that basis) it could be concluded that adverse effects on populations were 'unlikely'. These alternative screening values are chronic exposure dose rates below which it has previously been suggested that no measurable population effects would occur (IAEA, 1992; UNSCEAR, 1996). This was, of course, the only advice that would have been available prior to the current screening values recommended in the ERICA approach.

In our opinion, a regulator (or indeed a site operator) presented with all this information would find it hard to justify discharges at the 1980 levels in the sense that it would not be possible to rule out adverse effects on biota. Although the situation might be judged acceptable in terms of earlier suggestions (*i.e.* IAEA, 1992), it would not be possible to 'close out' the assessment using the ERICA methodology and screening dose rate without, at least, very substantial additional assessment and, quite possibly, following that, a substantial amount of research.

In 1980, radiation doses to the critical group of humans (a small group of high rate consumers fish and shellfish from Cumbrian coastal waters) were estimated to be about 1.2 mSv a^{-1} (FRL, 1980) using ICRP recommendations current at that time. The ICRP's recommended dose limit for the public at that time was 5 mSv a^{-1}, compared to 1 mSv a^{-1} currently. The discharges could therefore be deemed as being acceptable from the perspective of human radiological protection at that time.

4.10.2 2005 concentrations

Here, the results present less difficulty in interpretation. Only phytoplankton show a risk quotient in excess of one, and discharges at the 2005 rate (as opposed to the residual effects of past discharges) are only contributing about 10 % of the dose. Two further considerations apply to the results for phytoplankton.

First, the dose results almost entirely from the alpha emitting radionuclides ^{238}Pu, $^{239+240}$Pu, and ^{241}Am. The assumption is made (as for all other organisms) that all of the alpha decay energy is absorbed within the organism. However, phytoplankton are small single celled organism with dimensions comparable to the range of alpha particles in tissue; in practice, only a fraction of the alpha energy would be absorbed and the *average* doses would be less than that calculated by the ERICA methodology.

More importantly, at the average concentration of about 330 Bq kg^{-1} for these alpha emitters combined, taking account of the extremely small mass (and hence total activity content) of an individual organism, the mean time between alpha disintegrations in an individual phytoplankton cell would be in excess of 500 days. The life cycle of phytoplankton is short, the life span of individual cells being of the order of days. Therefore, only a small proportion of the phytoplankton population would receive any dose at all. The very small proportion that did would receive a high dose and may well be killed, although this would be unlikely to affect the population.

Comment: This emphasises that dose rates to single celled organisms from alpha emitters must be interpreted with care – whether in assessments or effects studies.

It is likely that, given a little more detailed consideration and assessment, including consideration of the radiation sensitivity of algae more generally, that discharges at the 2005 levels would be interpreted as having no adverse effects on the local ecosystem.

4.11 Testing of Tier 3 probabilistic calculations

As described above (Section 4.7), the advice provided in the Tool would lead to the implementation of Tier 3 in the case of the WC area in 1980 as numerous Risk Quotients exceeded unity. As the options are many in Tier 3 and the management of the output can rapidly become difficult, a decision was made to limit the following analysis within this tier to the reference organism “benthic mollusc” and ^{137}Cs (this combination was chosen because of the relatively high concomitant RQ observed at Tier 2 and empirical data availability). This allowed a straight-forward, direct comparison of risk for this particular organism at Tiers 2 and 3. Although risk quotients are not used at Tier 3, it is possible to derive an equivalent risk quotient by selecting the appropriate dose rate percentile (in agreement with the Tier 2 default, this equals the 95th percentile) and dividing by the ERICA screening dose rate used at Tier 2. In this way, the intended “conservatism” with respect to RQs in the Tier 2 approach can be considered. The empirical data which have been used in the Tier 3 assessment runs are presented in Table 4.12.

Table 4.12 Observed data for ^{137}Cs in the eastern Irish Sea as used in the Tier 3 test.

Media and Biota	Mean	SD	Min.	Max.
Water (Bq l^{-1})	4		1	20
Sediment (Bq l^{-1})	2800	2670	90	7700
Benthic mollusc (Bq kg^{-1})	217	187	14	470

Different probability distribution functions (PDFs) for water concentrations based on the information given in Table 4.12 were used initially. PDFs were chosen in the form of uniform, triangular and log-normal distributions all of which can be conceivably utilised based on the available information. This input is used by the Tool to define output PDFs for activity concentration in mollusc and in sediment using default distributions for CRs (log-normal) and K_ds (exponential), respectively. In this way, the differences resulting from the application of different distributions in input data could be explored. Furthermore, the robustness of Tool prediction using different underlying assumptions in the distribution of input data could be considered through comparison with measured data for the corresponding location and time (Table 4.13).

In all input data cases, the prediction made by the Tool for mollusc and sediment are substantially higher than measured data. The selection of a uniform distribution for the activity concentration in water results in the greatest over-estimation whereas the choice of an exponential distribution results in the closest agreement between predicted and measured activity concentrations. This is contrary to expectation in the sense that the provision of more information regarding the underlying distribution might be anticipated to result in more realistic prognoses. The fact that such large boundaries have been placed on the minimum and maximum values used in the uniform and triangular distribution configuration, reflecting the uncertainty in deriving water concentrations from contour maps, might largely account for this discrepancy. The corresponding total dose rates and RQs are presented in Table 4.14.

Tables 4.13 Outputs (in the form of PDFs) from Tier 3 of the Tool and observed data for comparison for a) ^{137}Cs activity concentration in mollusc, (b) ^{137}Cs activity concentration in sediment.

a)

PDFs used for activity concentration in water*	Cs-137 Activity concentration in mollusc (Bq kg^{-1}) Calculated in Tier 3 Mean	95th percentile	Observed Mean	Max.
Uniform (1,20)	702	2150	217	470
Triangular (1,20,4)	557	1670	217	470
Exponential (4,1,infinity)	335	1120	217	470

b)

PDFs used for activity concentration in water	Cs-137 Activity concentration in sediment (Bq kg^{-1}) Calculated in Tier 3 Mean	95th percentile	Observed Mean	Max.
Uniform (1,20)	41900	141000	2800	7700
Triangular (1,20,4)	33200	109000	2800	7700
Exponential (4,1,infinity)	19800	71500	2800	7700

*values in parenthesis relate to the PDF parameters as required by the Tool.

Table 4.14 Total dose rates and equivalent RQs for mollusc calculated at Tier 3 and Tier 2 using water concentrations (PDF) as input, CR and K_d default distributions.

PDFs used for activity concentration in water*	Total dose rate (µGy h^{-1}) and Risk Quotient: Calculated in Tier 3 Mean	95th percentile	RQ+	Calculated in Tier 2 Mean	RQ (conservative)
Uniform (1,20)	6.8	22.7	2.3	2.6	0.78
Triangular (1,20,4)	5.39	17.6	1.8	2.6	0.78
Exponential (4,1,infinity)	3.22	11.5	1.2	2.6	0.78

*values in parenthesis relate to the distribution parameterisation; +where 'RQ' is estimated as the 95th percentile dose rate divided by the ERICA Integrated Approach recommended screening dose rate of 10 µGy h^{-1}.

The results from the comparison of conservative RQ from Tier 2 (corresponding to a 95th percentile) with an equivalent RQ at Tier 3 (derived as described above) shows that the degree of conservatism employed at Tier 2 may be less than expected. The implementation of an uncertainty factor at Tier 2 based on the assumption that the underlying distribution is exponential was designed to provide a conservative estimate of the risk that might encompass the type II uncertainty (i.e. variability, inherent in the inputs and parameters used in the calculation). This exercise, using appropriately selected distributions for these components of the calculation, shows that this intention does not necessarily hold under all conditions.

To explore further whether the uncertainty associated with the use of ^{137}Cs activity in seawater contour maps might be leading to the discrepancy identified above, a second study was conducted using different PDFs for sediment concentrations, instead of water, as input data. The sediment data are more precisely defined (i.e. individual values are reported for specific locations). In this case, the sediment input data are used by the Tool to first define a PDF for activity concentration in water, which then makes use of the default CRs to calculate a PDF for activity concentrations in the organism and thereafter internal dose rates. The input PDF is also used for the calculation of external dose rate from sediment allowing total dose rates in the form of a PDF to be derived. The total dose rates and RQs for mollusc derived at Tiers 2 and 3 are presented in Table 4.15.

Table 4.15 Total dose rates and equivalent RQs for mollusc calculated at Tier 3 and Tier 2 using sediment concentrations (PDF) as input, CR and K_d default distributions.

PDFs used for activity concentration in sediment*	Total dose rate (µGy h^{-1}) and Risk Quotient Calculated in Tier 3 Mean	 95th percentile	 RQs	 Calculated in Tier 2 Mean	 RQ (conservative)
Uniform (90,7700)	0.72	1.24	0.12	0.46	0.14
Triangular (90,7700,2800)	0.66	1.1	0.11	0.46	0.14
Exponential (2800, 90, infinity)	0.53	1.46	0.15	0.46	0.14

*values in parenthesis relate to the distribution parameterisation

In this case, the RQs derived manually at Tier 3 compare closely with those reported at Tier 2. The RQs derived using a uniform or triangular underlying distribution for the input data are lower than the conservative RQs calculated at Tier 2 whereas the RQ derived using an exponential distribution for the input data at Tier 3 is slightly higher than the equivalent Tier 2 value. Therefore, in this case, the intended level of conservatism implemented at Tier 2 holds with the expectation (*i.e.* encompasses the type II uncertainty associated with underlying input data and parameters in a convincing manner).

It is clear from this exercise that the assessor should be careful in the blind acceptance of uncertainty factors applied at Tier 2 in the process of deriving a conservative risk quotient. If the uncertainties associated with underlying datasets, parameters and input data, are large then the higher percentiles for the output, generated through probabilistic analyses, may be substantially higher than those approximated at Tier 2.

4.12 Conclusions

Application of SRS-19 to distant locations from the source should be viewed with some caution owing to problems in defining distance in terms of the direction of down-stream longitudinal current. Advice concerning the limitations in the default model should be included in the Help file. This advice could be expanded upon by including additional warning with respect to the application of high K_d radionuclides where hyper-conservatism may be observed in some cases, especially those where source to target distances are great.

Tool predictions for activity concentrations in reference biota, calculated by selecting input data from water activity concentrations determined in 1980, were compared with corresponding values for biota sampled in the same year. Generally, the predictions made by the Tool using the default CRs are reasonable (often within one order of magnitude) and where there are large differences these can often be explained. A notable exception to this was observed for seabird where differences between predicted and measured values exceeded two orders of magnitude. Similar comparisons to test water activity concentration predictions made by the Tool show that calculated and measurement values lie within one order of magnitude for locations relatively near to the discharge point along the Cumbrian coast.

Although there were some minor difficulties in applying results from the *Add organism* module (not in relation to the functionality itself, more in relation to the fact that in the Tool there was no easy way of entering the level of detail on occupancy within different parts of the selected organism's habitat in correspondence with the original work), the Tool provided sensible results that were in reasonable agreement with a study of black-headed gulls in an estuarine environment on the Irish Sea coast (Woodhead, 1986).

The example application of the ERICA methodology in a role-playing regulatory context suggests that, in a complex assessment with many radionuclides and many reference organisms involved, if the assessment results at Tier 2 show a substantial number of risk quotients exceeding 1 then it is unlikely that consideration of potential effects at either Tier 2 or at Tier 3 will result in any clear resolution of the situation. In other words, one will be left in a position whereby the possibility of effects cannot be excluded, but it is not possible to determine what (if any) effects may be expected. This arises in some cases from the general lack of effects data for some organism groups, and in the case of organism groups for which there are many (and potentially conflicting) relevant effects studies, from the lack at the present time of a systematic evaluation by organism type of the available effects data.

It should be remembered that the intention for ERICA Tier 3, in terms of the evaluation of dose rates derived, was that it would be "open ended". The screening dose rates, as applied at earlier tiers become redundant and the onus to establish whether environmental dose rates are acceptable or not falls on the group of stakeholders responsible for the assessment. Such a group might normally comprise of regulators and representatives from industry and the process of establishing an acceptable level of protection might be performed through synthesis and discussion of effects data with a subsequent agreement regarding case-relevant dose rate criteria. The ERICA Tool (mainly the FREDERICA effects database) would then be used as a decision support system. The ERICA methodology, therefore, gives no definitive answer to the question of what constitutes an acceptable degree of environmental protection once risk quotients above 1 have been documented at the screening tiers. In light of the considerations given above, this intended application may be more difficult than envisaged and there is a danger that, contrary to the intentions of the ERICA Integrated Approach, compliance with the screening dose rate and risk quotients at Tier 2 could be used effectively as a 'pass / fail' criterion for assessment and nothing more.

The 1980 Cumbrian coast scenario that we have assessed suggests that application of the ERICA screening dose rate would lead to compliance criteria that are equally, if not more, restrictive than those based on human radiological protection. Having said this, the screening dose rate is arguably placed at a level that might be acceptable from the perspective of modern-day expectations for environmental protection in the sense that an area close to the Sellafield plant during a period when radionuclide discharges were near their peak, is flagged as being a situation for which more detailed assessment, beyond a simple calculation of risk quotients, is required. However, this case study indicates that currently available effects data, and the way in which that data are organised, makes carrying out such a detailed assessment very difficult.

Probabilistic model runs have been made at Tier 3 using data that might be typical of those available to an assessor. Although the Tier 2 uncertainty factor is expected to generate conservative RQs (approximating to the 95th percentile), this will not always be the case as a comparison of the risk quotients generated in the exercise presented demonstrates. If the assessor questions the validity of applying default UFs at Tier 2, they should utilise the functionality of Tier 3 as an appropriate alternative to deriving dose rates. However, we suggest that probabilistic analyses are only considered in cases where the assessor has sound data from which reasonable PDFs can be defined.

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5 Komi Republic, Russia

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5.1 Introduction

Areas in the Komi Republic (Russia) have enhanced levels of naturally occurring radioactivity and provide a useful case study focusing on natural decay series radionuclides in terrestrial ecosystems. The authors of this chapter play the role of an assessor with a view to testing the robustness of the Tool outputs and guidance in a decision-making context. In playing this role it was also considered necessary to test default parameters and databases used by the Tool (*e.g.* transfer parameters, effects summary tables) through comparison with field data. Highly contaminated sites at Vodnyi (Komi Republic) provide an ideal test site as environmental effects have been studied there for several decades (Taskaev et al. 2003).

The assessment procedure as prescribed by the dialogue screens of the ERICA Tool has been followed as strictly as possible and this is reflected in the subsequent section headings.

5.1.1 Stakeholder involvement

Stakeholder involvement has not been considered explicitly in this hypothetical assessment. In a "real" situation it would have been appropriate to involve research institutes actively involved in the study area and local people living in proximity to the contaminated sites.

5.1.2 Areas of consideration

For the assessment three areas in different parts of the Komi Republic are considered: Two areas with enhanced natural levels (Middle Timan, North Urals) and one area contaminated as a result of industrial activities in the past (Vodnyi) (see Figure 5.1). More thorough site descriptions are given in subsequent sections.

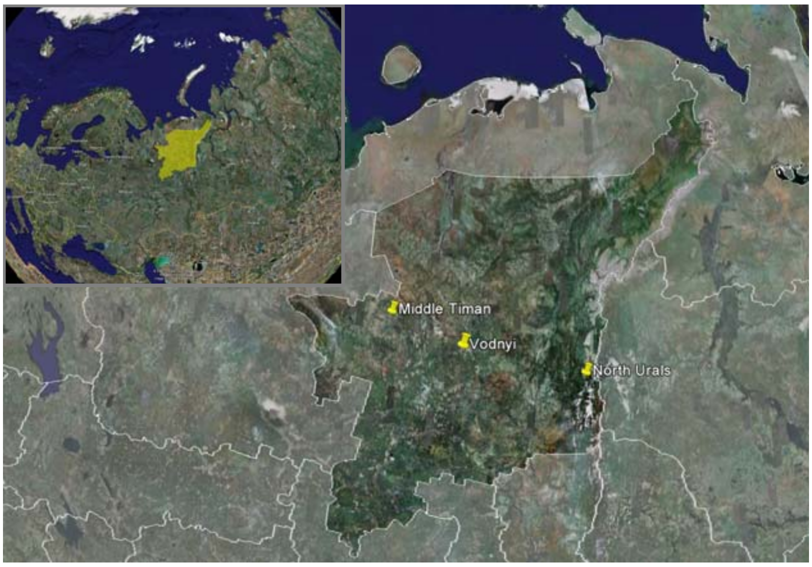

Figure 5.1 Location map of the three assessment areas in the Komi Republic, Russia.

5.1.3 *Radionuclides of relevance*

Radionuclides important for the study sites are summarised in Table 5.1. These nuclides form components of the ^{238}U and ^{232}Th decay series. In ERICA, radionuclides with physical half lives of 10 days or less have been included in the dose conversion coefficient of their parent radionuclide (as shown in Table 5.1).

Table 5.1 Radionuclides (and their progeny) considered in this assessment.

Nuclide	Progeny in assumed equilibrium
U-238	-
Th-234	-
U-234	-
Th-230	-
Ra-226	Rn-222, Po-218, Pb-214, Bi-214, Po-214
Pb-210	Bi-210
Po-210	-
Th-232	-
Ra-228	Ac-228
Th-228	Ra-224, Rn-220, Po-216, Pb-212, Bi-212, Po-212, Tl-208

5.2 Assigning assessment details – general points

5.2.1 *Environmental Media Concentration Limit (Tier 1)*

At the time of testing of the Tool, the Environmental Media Concentration Limit (EMCL) values had not been updated using the most recent version of the terrestrial Concentration Ratio (CR) database which contained revised values for ^{226}Ra and ^{210}Po. For this reason, the revised EMCLs for ^{226}Ra and ^{210}Po were derived using the methodology outlined in the Help function for use in this assessment (and to maintain comparability between Tiers 1 and 2). "F" values were derived using the equations provided in Table H24 of the (draft) Help and the Tool database values for CRs and dose conversion coefficients (DCCs). Default radiation weighting factors of 10 for alpha particles were used. The simulations were run probabilistically using appropriate software (Ekström and Broed, 2005). The 95th percentile of the F value was selected in deriving the EMCL for each reference organism (RO) as:

$$\mathrm{EMCL} = \mathrm{PNEDR} / \mathrm{F}$$

Where:

PNEDR is the predicted no effects dose rate or screening dose-rate (i.e. 10 μGy h^{-1});

F is the dose rate that an organism will receive for the case of a unit concentration in soil (μGy h^{-1} per Bq kg^{-1} of soil).

The lowest EMCL values derived for ^{226}Ra and ^{210}Po were 230 Bq kg^{-1} (DW) and 25 Bq kg^{-1} (DW). The limiting reference organism were found to be "Lichen and Bryophyte"in both cases.

5.2.2 *Ecological parameters (Tiers 2 and 3)*

It was noted that there was particularly poor concentration ratio data coverage within the Tools default database for Amphibian, Bird egg, Flying insect, Gastropod and Soil invertebrate concerning the natural nuclides shown in Table 5.1. In these cases significant recourse to derived values was required.

The derived methods used included: similar taxonomy (Code 1); similar reference organism (Code 2); similar biogeochemistry and taxonomy (Code 6); allometric and/or other modelling methods (Code 8) "highest available value" (Code 9). Such assumptions will, of course, affect the reliability of internal dose-rate predictions for these reference organisms.

Default occupancy factors and radiation weighting factors were used unless specified otherwise.

5.2.3 Input data

For the screening Tiers (1 and 2) – for the sake of simplicity and transparency – the only input parameter considered was soil activity concentrations. Site specific plant and animal data were included at Tier 3.

In connection with external dose-rate predictions at Tiers 2 and 3, better estimates may be derived by making assumptions about the *in situ* fresh weight density of soil. A dry weight (DW) / fresh weight (FW) ratio of 0.5 was assumed, which was considered a reasonable approximation for soils in the assessment areas. In practice, the uncorrected external absorbed dose-rate will be reduced by a factor of two by applying this correction factor.

5.3 Middle Timan

Middle Timan is located in the northwestern part of Komi Republic (N 64°5' E 50°9') in the watershed of Vorikva and Mezen rivers (Figure 5.1). Enhanced levels of natural radionuclides in this area are due to rocks containing thorium-rare-earth mineral impregnations. Soils are skeletic or haplic podzols and stagnic albeluvisols. Seventy percent of the area is covered with fir and fir-birch forest and the rest bog. Among trees and bushes: *Picea obovata*, *Betula pendula*, and to lesser extent *Larix sybirica* and *Sorbus aucuparea* are common. *Vaccinium vitis-idea*, *Vaccinium myrtillus*, *Vaccinium uliginosum* and *Empetrum nigrum* are the dominating low growing shrubs (Vinogradov, 1957).

5.3.1 Soil input data

Soil data used for this assessment were from 1977. The data are taken from Titaeva and Taskaev (1983) and Shmuktomova (1986). Generally, data are for horizons in the upper 25 cm of topsoil. A summary of activity concentrations of ^{238}U, ^{226}Ra and various thorium isotopes are shown in Table 5.2.

Table 5.2 Radionuclide activity concentration in soil from Middle Timan (Bq kg^{-1} ash weight).

		Activity concentration (Bq kg^{-1} AW)		
Radionuclide	**n**	**Mean (Median)**	**SD**	**Min.-Max.**
Uranium-238	21	20 (16)	12	7-50
Thorium-230	16	24 (19)	15	3-64
Radium-226	21	46 (30)	37	15-150
Thorium-232	21	200 (98)	360	20-1700
Thorium-228	16	83 (38)	110	10-400

The soil data are presented in terms of activity concentration per unit ash weight (AW) reflecting the reporting format of the original data archives. No information concerning the ash to dry weight ratios is available for the study area. In view of the fact that the percentage of combustible material is likely to vary substantially between organic rich surface soils and underlying mineral soils, the application of a generic correction factor was not deemed suitable. Levels of organic matter (OM) lie approximately within a range defined by mineral soils (typically 1-6 % OM) and organic soils (typically > 20 % OM) (Brady, 1990). The lack of site-specific information resulted in our choice not to introduce a correction for this thus avoiding the possibility of entering erroneous information for a parameter that is likely to make little difference to overall uncertainty. This decision, however, will lead to an over-estimation of biota activity concentrations and resultant internal dose rates, most especially in cases where the organic matter content in soil is high.

5.3.2 Tier 1

Using maximum soil concentrations of all nuclides as shown in Table 5.2 the risk quotient (RQ) at Middle Timan exceeds unity (RQ=3.4). The risk from this screening Tier analysis is therefore deemed unacceptable and further investigation at Tier 2 would be warranted.

5.3.3 Tier 2

The assessment methodology requires the entry of expected values at Tier 2 for all parameters and input values. Mean values from Table 5.2 were used as input data in Tier 2. A default Uncertainty Factor (UF) of 3 was chosen for the calculations to ensure conservative estimates of risk. It was considered appropriate to test for the 5 % probability of exceeding the screening dose rate. The screening dose rate was selected to be 10 $\mu Gy\ h^{-1}$ (i.e. the ERICA Integrated Approach recommended value) and all 14 reference organisms were selected for further consideration. In the absence of information showing that a particular reference organism type was not present at the site, it would be likely that an assessor would include all reference organisms in the assessment.

Total dose rates estimated are in Figure 5.2. The "conservative" risk quotient was slightly above screening levels for the reference organism "Lichen and Bryophytes" (RQ=1.1). For other reference organisms, the conservative RQs were lower (0.014-0.35). However, "expected" risk quotients are below one for all reference organisms and more detailed analyses at Tier 2, either through the derivation of site-specific uncertainty factors or justified selection of a less conservative uncertainty factor, might have resulted in a screening analyses that reported the probability of harm was acceptably low (this could alternatively be achieved by running the assessment probabilistically at Tier 3 and comparing the 95 % dose rate with the 10$\mu Gy\ h^{-1}$ benchmark). For cases where the RQ (expected) is below one, but the RQ (conservative) is greater than one advice that allows the assessor a certain degree of discretion and thereby flexibility would be preferable. Using the Tool advice (in Help and D-ERICA) as it stood at the time of testing, an assessor would be required to access Tier 3 regardless of the fact that the conservative RQ was only marginally greater than unity and that consideration of other evidence (e.g. background and summary effects data) suggests that significant environmental impact is unlikely. Nonetheless, for the sake of consistency and moreover to allow comparison between site specific and predicted data the assessment was taken to Tier 3 for this site.

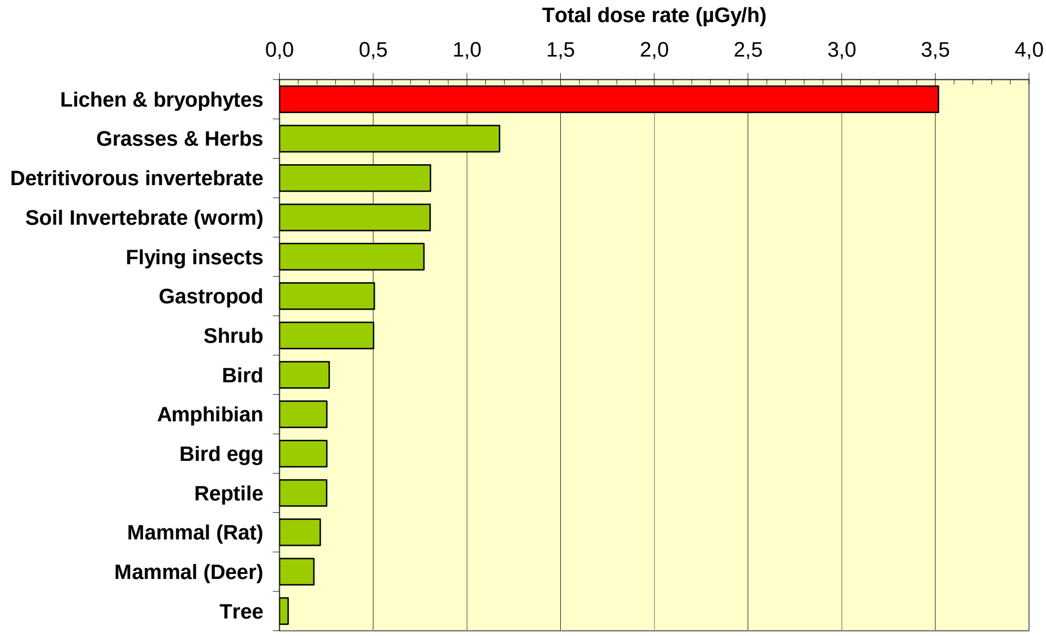

Figure 5.2 Total (expected) dose-rates for reference organisms at Middle Timan. Bars in red exceeded the Tier 2 conservative RQ value.

An assessor might use output information from Tier 2 to identify which radionuclides are contributing most to the dose and which component (internal or external) dominates. For this particular case, data provided by the Tool show that the total dose rates are mainly attributable to the internal component of dose-rate (66-98 %). Ra-226 and ^{228}Th are the nuclides that dominate in terms of their contribution to total dose-rates. Furthermore, the external dose rate (on soil) for "Lichen and Bryophytes" is low compared to other reference organism categories (β,γ–DCCs on soil for this reference organism appear to be much lower than for other reference organism categories). These particular DCCs should therefore be checked. For all other ROs predicted external dose rates are in the range of 0.011-0.054 µGy h^{-1}. The internal dose rate for "Tree" is considerably lower than for other ROs (0.030 µGy h^{-1}). This is attributed to a low CR compared with other ROs, especially for radium. This issue will be considered further in the next section.

5.3.4 Tier 3

The main emphasis on the next stage of the study was to test the default CRs used by the ERICA assessment Tool against site-specific data from Middle Timan as decision-making considerations have been dealt with as far as was possible above. Unfortunately, there are only plant data available for the radioisotopes ^{238}U, ^{226}Ra and ^{232}Th, the data are from same time and sites as the soil input data given in Figure 5.2 (Titaeva and Taskaev, 1983). Another drawback is that all data are originally given in ash weight. To mitigate this, correction factors from the Help functions have been used to derive fresh weight activity concentrations. Most of the data fall under the categories "Grasses and Herbs" and "Shrub" and a very limited data set exists for "Lichen and Bryophytes". Using statistical information for soil activity concentrations from Table 5.2, predictions were made for plant categories assuming lognormal distributions. No pre-analyses of the available empirical data were conducted in line with the knowledge that lognormal distributions are often applied to environment data sets (Blackwood, 1992). Predictions are compared with the site-specific data in Table 5.3.

Table 5.3 Statistical information for predicted and measured activity concentrations in different reference organisms at Middle Timan.

	Activity concentration (Bq kg^{-1} FW)						
	Predicted			Measured			
	Mean	5^{th} percentile	95^{th} percentile	n	Mean	SD	Range
Radium-226							
Grasses and Herbs	2.0	0.15	6.8	17	0.72	0.40	0.19-1.7
Shrub	1.2	0.32	3.1	14	2.0	2.1	0.11-8.2
Tree	0.034	0.0032	0.12	9	8.5	6.7	0.46-18
Lichen and Bryophytes	11	3.2	27		-	-	-
Thorium-232							
Grasses and Herbs	9.4	0.22	36	17	0.95	0.78	0.16-3.4
Shrub	3.5	0.063	14	14	4.1	11	0.15-44
Tree	0.24	0.011	0.87	9	2.4	3.6	0.019-11
Lichen and Bryophytes	23	1.7	80	1	15	-	-
Uranium-238							
Grasses and Herbs	0.31	0.0059	1.2	18	0.13	0.072	0.042-0.25
Shrub	0.15	0.0061	0.57	11	0.11	0.12	0.015-0.36
Tree	0.14	0.0055	0.55	7	0.25	0.34	0.015-0.98
Lichen and Bryophytes	1.5	0.061	5.1	1	2.6	-	-

For all radionuclides, the Tool predicted activity concentrations seem to be in reasonable agreement with the site specific data for “Grasses and Herbs” and “Shrub”, whereas for “Tree” there seem to be an under prediction of several orders of magnitude for ^{226}Ra and ^{232}Th. However, transfer of radionuclides from soil to trees differs largely depending on the part of the tree that has been sampled. The available data from Middle Timan (when specified) comprises of leaves, stem, branches and above-ground parts of plants, with leaves having the lowest concentrations. The underlying datasets on ^{226}Ra and ^{232}Th CRs for “Trees” are based on relatively large empirical datasets and therefore the discrepancy cannot be assigned to the fact that derived values have been used to characterise transfer. In the absence of further information on the environmental conditions under which CRs have been derived for the studies constituting the default values compared to those in Middle Timan, it is not possible to comment further on this anomaly.

For comparison purposes (with Tier 2), total dose rates to all four “plant” reference organisms were predicted using:

- Soil input concentrations - ^{238}U, ^{226}Ra, ^{232}Th, ^{230}Th, ^{228}Th (Table 5.2) with Lognormal distributions assumed for all radionuclides.
- Site specific plant concentrations - ^{238}U, ^{226}Ra, ^{232}Th (Table 5.3). Lognormal distributions assumed for “Grasses and Herbs”, “Shrub” and “Tree”, whereas exponential distributions were assumed for “Lichen and Bryophytes” reflecting the limited amount of data associated with the latter.
- Default distributions of concentration ratio parameters from the assessment Tool.

Selected output statistics for Total dose rates at Tier 3 are shown in Table 5.4.

Table 5.4 Statistical information for Tier 3 predicted Total dose rates (µGy h^{-1}) for Middle Timan.

Reference organism	Mean (*)	Total Dose rate (µGy h^{-1}) 5th Percentile	Median	95th Percentile
Grasses and Herbs	0.89 (1.2)	0.14	0.39	2.9
Shrub	0.67 (0.50)	0.16	0.49	1.8
Tree	1.3 (0.046)	0.38	1.0	3.0
Lichen and Bryophytes	3.6 (3.5)	1.2	2.8	8.5

*Total dose rate outputs from Tier 2

The mean values reported at Tier 3 roughly correspond to the values reported in Tier 2 with the exception of the category “Tree”. This is because at Tier 3 measured activity concentrations of radionuclides in plants were entered which, in the case of ^{226}Ra and ^{232}Th in trees, were considerably higher than those predicted through the use of a CR (see Table 5.3). For other vegetation, the CRs represent the site well leading to a close similarity in predicted dose rates between the two Tiers. This highlights a possible pitfall in conducting an assessment and it would seem pertinent whenever possible to use all available data (including activity concentration data on plants and animals) at Tier 2 to avoid the possibility of underestimating (or for that matter over-estimating) the dose rates at this stage.

5.3.5 *Effects*

Radiation induced effects are not available for Middle Timan and therefore comparison between Tool predicted and observed data is not possible. However, the predictions suggest that even conservative estimates of dose rate (*i.e.* 95th percentile) fall below the ERICA screening dose rate and are generally at a level where statistically significant effects would not be expected for most biota groups (this can be inferred from consideration of the effects summary tables included in the Tool). Consideration of “normal” background dose rate data also shows that the predicted dose rates fall within one order of magnitude of the upper range values documented in the Tool and as such add weight to the view that the prevailing exposure levels do not pose a serious environmental hazard. In view of the fact that this is an area of naturally enriched radioactivity where plants and animals have adapted to thrive it is unlikely that decision maker would pursue this assessment further.

5.4 North Urals

The second area constituting this case study is located in northeastern Komi Republic (N 63° E 58°47') (Figure 5.1). Enhanced levels of natural radioactivity in this area are due to the presence of rocks containing rare-earth mineral impregnations (Th and U). The area belongs to the mountain part of North Taiga subzone. Studies were carried out in mountain-wood and mountain-tundra zone. Soils are described as skeletic or haplic podzols. The mountain-wood zone is covered with fir and fir-birch forests. Common trees and bushes are *Picea obovata*, *Abies sybirica*, *Betula tortuosa, Pinus sibirica* and *Sorbus aucuparea.* Among the low growing shrubs and herbaceous species *Vaccinium myrtillus*, *Veratrum lobelianum and Solidago virgaurea* dominate. In the mountain-tundra zone *Betula nana* and *Salix laplandica* are the predominant shrubs. Among low growing shrubs and herbaceous species *Vaccinium vitis-idea*, *Vaccinium uliginosum, Arctostaphulos uva-ursi* and *Empetrum nigrum* are common. *Pleurosium schreiberi* is the predominant moss species.

5.4.1 *Soil input data*

Soil profiles were sampled in 1979 (Titaeva and Taskaev, 1983). A summary of soil activity concentrations of ^{238}U, ^{226}Ra and ^{232}Th in the upper 25 cm are shown in Table 5.5.

Table 5.5 Radionuclide activity concentration in soil from the North Urals site (Bq kg^{-1} AW).

Radionuclide	n	Activity concentration (Bq kg^{-1} FW) Mean (Median)	SD	Min –Max.
Uranium-238	34	57 (41)	69	17-420
Radium-226	33	310 (67)	1000	18-5800
Thorium-232	34	70 (60)	34	18-160

As for Middle Timan, the soil data are presented in terms of activity concentration per unit ash weight. Unfortunately, no information concerning the ash to dry weight ratios is available for the study sites. For reasons described in Section 5.3.1 it was decided not to introduce a correction ash weight to dry weight.

5.4.2 Tier 1

Using maximum soil concentrations of all nuclides shown in Table 5.5 the risk quotient at Middle Timan exceed unity (RQ=26). A decision was therefore made to move to the next Tier.

5.4.3 Tier 2

Mean values from Table 5.5 were used as input data in Tier 2. As for Middle Timan, a default Uncertainty Factor of 3, in connection with risk quotient calculations, and a screening dose rate of 10 µGy h^{-1} were used. All 14 terrestrial reference organisms were considered in the assessment.

The "conservative" risk quotient was above unity for four reference organisms with "Lichen and Bryophytes" constituting the highest value (RQ=2.8); the other RQs ranged from 0.025-1.2. RQs for different reference organisms correspond to Total dose rates given by Figure 5.3. As was the case for Middle Timan, "expected" RQs were all below one.

As shown in Figure 5.3, "Lichen and Bryophytes" receive the highest total dose-rate (9.3 µGy h^{-1}), while "Tree" receive the lowest (0.082 µGy h^{-1}). Ra-226 is the most important contributor to both internal and external dose. The internal component of dose rate dominates. Excluding "Lichen and Bryophytes", external dose rates to reference organisms ranged from 0.028 to 0.14 µGy h^{-1}. The reference organism "Mammal (deer)" received the lowest external dose whereas detritivorous invertebrates were exposed to the highest levels.

Since data for only three radionuclides (^{238}U, ^{226}Ra, ^{232}Th) were available for the North Urals, it was decided to run a simple test to check the importance of other relevant radionuclides from Table 5.1 to the Total dose rates to reference organisms. Assuming secular equilibrium, the ^{226}Ra progeny ^{210}Po had a large impact on the dose rate to "Lichen and Bryophytes" increasing it from 9.3 to 72 µGy h^{-1}. The importance of ^{210}Po was also evident for other plant reference organisms , for example the estimated Total dose rate for "Grasses and Herbs" doubled. For all animal reference organisms ^{210}Po seems to be of little importance.

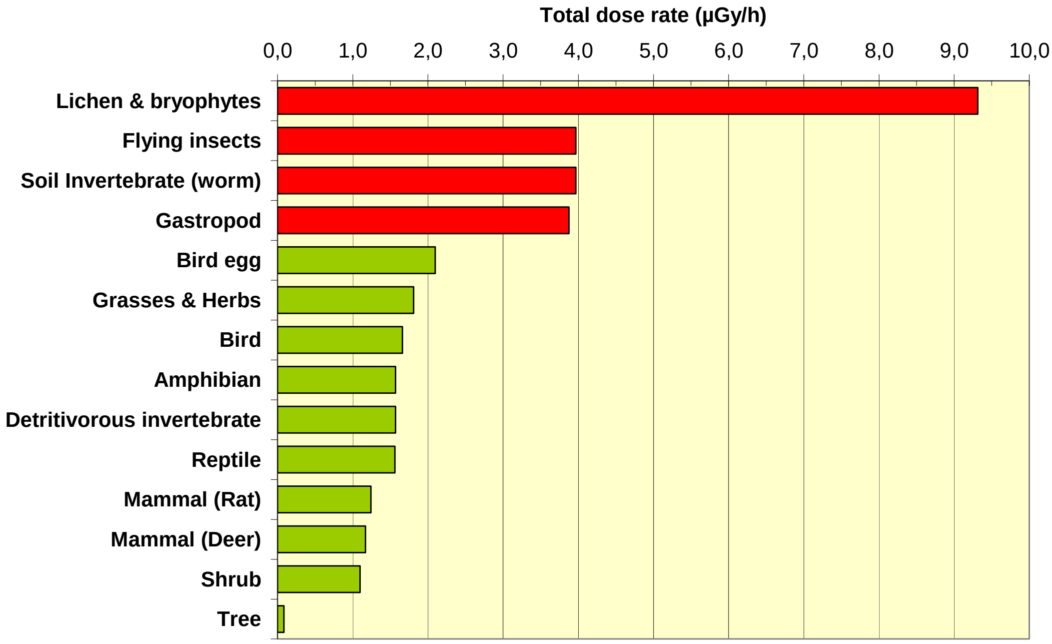

Figure 5.3 Total (expected) dose rates for reference organisms at the North Ural site. Bars in red exceeded the Tier 2 conservative RQ value.

5.4.4 Tier 3

As was the case for Middle Timan, the main emphasis for this case was to test the default CR values used by the ERICA assessment Tool against site specific data. The site-specific plant data are from 1979 and from the same sites as the soil input data given in Table 5.5. As for the previous case, all data are originally given in ash weight and correction factors from the Help file has been applied to derive fresh weight activity concentrations. Most of the data fall under the categories "Grasses and Herbs" and "Shrub".

Using statistical information for soil from Table 5.5, predictions were made for plant categories assuming lognormal distributions. These are presented together with the site-specific data in Table 5.6.

As for the Middle Timan area, measured activity concentrations for "Grasses and Herbs" and "Shrub seem to be within the range of those predicted for ^{226}Ra. However, activity concentrations of ^{226}Ra in "Tree" seem to be under-predicted by the ERICA assessment Tool. As was the case for Middle Timan, tree parts sampled may explain some of this discrepancy but it seems clear that the default CRs for ^{226}Ra are not representative for this particular study. In contrast to the Middle Timan area, site specific data are available for "Lichen and Bryophytes", seem to be in the lower range of the predicted concentrations, but no conclusions may be drawn since the amount of data is limited and the correction from ash weight to fresh weight is a source of uncertainty. There seems to be a general over-prediction of ^{238}U and ^{232}Th in most cases.

Table 5.6 Statistical information for predicted and measured activity concentrations in different reference organisms at the North Ural site

	Activity concentration (Bq kg^{-1} FW)						
	Predicted			Measured			
	Mean	5th percentile	95th percentile	n	Mean	SD	Range
Radium-226							
Grasses and Herbs	14	0.26	57	24	1.8	1.5	0.39-6.0
Shrub	8.8	0.48	33	18	3.5	3.8	0.092-12
Tree	0.25	0.0056	0.93	10	3.0	3.2	0.061-7.8
Lichen and Bryophytes	78	4.5	290	2	5.8	0.79	5.2-6.4
Thorium-232							
Grasses and Herbs	3.1	0.18	12	26	0.037	0.073	0.0030-0.38
Shrub	1.1	0.047	3.7	19	0.077	0.074	0.0042-0.29
Tree*	0.077	0.0096	0.24	17	0.051	0.056	0.0012-0.051
Lichen and Bryophytes	7.2	1.6	19	2	1.1	1.1	0.29-1.8
Uranium-238							
Grasses and Herbs	1.0	0.016	3.8	28	0.021	0.033	0.0010-0.13
Shrub	0.49	0.015	1.9	20	0.034	0.033	0.0033-0.14
Tree	0.45	0.014	1.8	18	0.026	0.019	0.0003-0.068
Lichen and Bryophytes	5.2	0.16	18	3	0.33	0.018	0.32-0.35

*high result 7.0 Bq kg^{-1} removed from the measured data

Regarding Total dose rate predictions, two cases were considered:

Case 1: Weighted mean data (all available nuclides) in soil. Soil concentrations assumed to be lognormally distributed. Default distributions of concentration ratio parameters from the assessment Tool are implemented.

Case 2: Weighted mean data (all available nuclides) in soil + site specific data for plants (all available nuclides). Lognormal distributions assumed for “Grasses and Herbs”, “Shrub” and “Tree”, whereas exponential distributions were assumed for “Lichen and Bryophytes” due to the limited amount of data.

The resulting output total dose rates for both cases are shown in Figure 5.4.

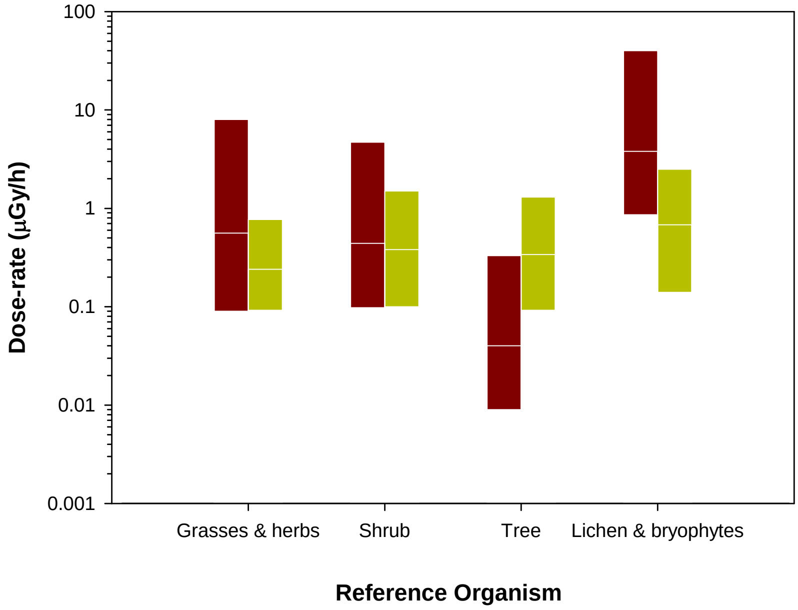

Figure 5.4 Total dose rates for reference organisms (µGy h^{-1}). The boxes indicate 5th percentile, median and 95th percentile. Brown=Case 1 and green=Case 2.

Virtually the entire estimated Total dose rate arises from exposure to ^{226}Ra. An assessor/decision maker considering these results might reflect, as for the case of Middle Timan, that this is an area of naturally enhanced levels of background radiation and that plants and animals have adapted to live in this environment. Furthermore, the predicted dose rate ranges do not generally exceed the 10 µGy h^{-1} screening level used in earlier Tiers. Only in the case of "Lichen and Bryophytes" is the screening dose rate exceeded substantially and this for conservative dose rate estimates (i.e. using UF=3). In view of the fact that Lichen and Bryophytes constitutes a relatively radio resistant organism group, at least for acute radiation exposures (UNSCEAR, 1996), a decision maker might reduce the emphasis placed on this particular result.

5.5 Vodnyi settlement area

As a result of 'radium-from-water' operations (ca. 1931-50), and uranium extraction from ores (1947-56), the Vodnyi area comprises of a number of small contaminated sites (Table 5.7). The radioecology of the sites and the effects of elevated concentrations on indigenous flora and fauna have been studied for several decades under the auspices of the Russian Academy of Sciences. Studies have indicated biological effects in some of the most contaminated areas. For a more comprehensive overview see Taskaev et al. (2003).

Table 5.7 Overview of contaminated sites in Vodnyi.

Site	Description of contamination
Krokhal (Ra site 1)	Contaminated by discharge of radium-rich ground water
Factory 10 (Ra site 2)	Contaminated by discharge of radium-rich ground water
Obzhig (U-Ra site 1)	Site contains residues similar to Otvally with the additional presence of large quantities of semi-decomposed woody residues in the top soil.
Otvally (U-Ra site 2)	Contaminated by tailings from residues of radium extraction from groundwater and from the processing of uranium ores

Figure 5.5 Map showing the approximate locations of the Vodnyi study sites: Obzhig (U-Ra site 1), Krokhal (Ra site 1), Factory 10 (Ra site 2) and Otvally (U-Ra site 2). The picture was taken outside the Krokhal site in May 2006.

5.5.1 Soil input data

Radionuclide activity concentrations in soils from the sites shown in Figure 5.5 have been measured since the late 1950s. The latest published results coming from 2001 (Taskaev et al., 2003). The early data, however, are rather scarce. The most comprehensive measurements are from the early 1970s and it was decided to use these data in the following case study runs. As was the case for the other two areas, sampling depths of <25 cm were considered when deriving data. In Table 5.8 weighted mean concentrations from Krokhal (Ra site 1), Obzhig (U-Ra site 1) and Otvally (U-Ra site 2) are shown (source: Titaeva et al. 1984). Such information is not available for Factory 10 (Ra site 2). The ranges of most radionuclides are large and the data are poorly defined in many cases. Consequently, statistical parameters of the soil data similar to those in Tables 5.2 and 5.5 are not available. However, maximum measured values of all radionuclides, based on available soil data, for the sites from 1972-73 are shown in brackets (sources: Moiseev et al., 1973; Maslov, 1980).

Table 5.8 Mean (highest measured) activity concentrations of radionuclides in soil from Vodnyi sites (Bq kg^{-1} AW).

Nuclide	Krokhal (Ra site 1)	Factory 10 (Ra site 2)*	Obzhig (U-Ra site 1)	Otvally (U-Ra site 2)
U-238	48 (140)	(32)	2300 (-)	700 (-)
U-234	44 (120)	(77)	1200 (2500)	860 (-)
Th-230	1000 (4800)	(410)	240000 (-)	61000 (-)
Ra-226	10000 (95000)	(24000)	14000 (160000)	35000 (-)
Po-210	7900 (-)	(-)	10000 (-)	41000 (-)
Th-232	27 (94)	(33)	22 (150)	24 (-)
Th-228	1500 (6400)	(2000)	10000 (-)	3900 (-)

- Not available *Highest measured activity concentrations only

Other radionuclides in the ^{238}U and ^{232}Th decay series, such as ^{234}Th and ^{210}Pb, will contribute to the dose rates in these environments, although these might be shown to be of low importance to the overall dose-rates for plants and animals. A decision was made, therefore, to only include those radionuclides for which empirical data were reported. This avoided the necessity of introducing any assumptions concerning equilibrium at this stage.

5.5.2 Tier 1

Since the soil activity concentrations at Vodnyi sites are very high, it was decided to use maximum ^{226}Ra soil activity concentrations only in the “screening” process. Using this information, risk quotients for ^{226}Ra exceed screening criteria for all four Vodnyi sites (RQs: 100-700). The highest RQ was observed for Obzhig (U-Ra site 1) and the lowest for Factory 10 (Ra site 2). Although no maximum data were available for Otvally (U-Ra site 2), even the mean ^{226}Ra activity concentration for the site yields an RQ much in excess of unity.

Chemical toxicity of ^{238}U (or ^{232}Th) is not considered within the ERICA Tool. This may lead to an underestimation of risk at the U-Ra contaminated sites Obzhig and Otvally.

5.5.3 Tier 2

As for case studies in Middle Timan and North Urals, a default Uncertainty Factor of three and a screening dose rate of 10 μGy h^{-1} was selected for the calculations. All reference organisms were selected for further consideration. Since weighted mean is not available for the Factory 10 site it was decided to leave it out in the following assessment. All available activity concentrations were used for this Tier.

The expected value RQs are above one for all reference organisms at all three sites based on the ERICA screening criteria the level of risk is considered unacceptable at Krokhal, Obzhig and Otvally. Total dose rates to all reference organisms are shown in Figure 5.6.

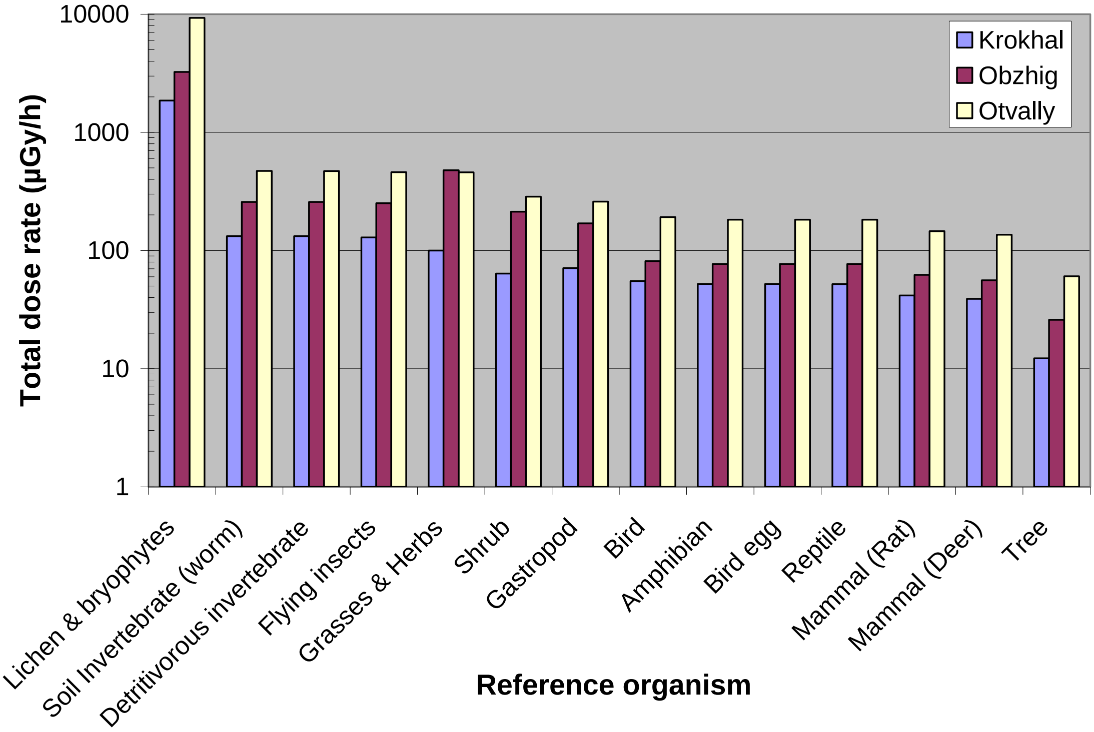

Figure 5.6 Total (expected) dose rates for reference organisms at different Vodnyi sites.

According to Tier 2 outputs, the internal component of dose dominates completely over the external component. For most reference organisms (at least animal reference organisms), ^{226}Ra is the main contributor to the overall dose rate. The exception is the group "Lichen and Bryophytes" where the internal component of ^{210}Po exposure dominates the total dose rate (this also explains the relatively higher Total dose rates for this reference organism). Clearly this would suggest further study, field work to sample this organism category. It might also be useful to consider the underlying datasets. The source of ^{210}Po deposition is usually from the atmosphere via in-growth from ^{210}Pb deposition. The case in Krokhal is different because the source of ^{210}Po is contaminated soil. The applicability of the default CR datasets may be questioned for this reference organism.

Comparison of predicted external dose-rates versus measured GAKRs

Predictions of (total) external dose-rates for on-soil organisms can be compared with gamma air kerma rates (GAKRs) measured in the field with some caveats. A factor is required to convert the absorbed dose rate in air to the absorbed dose in a tissue equivalent ellipsoid representing a reference organism. This calculation has not been performed explicitly in this study as this factor will not deviate substantially from unity for most of the organisms we are interested in (see Table 7.1). For example, in the case of a 15 cm radius tissue equivalent sphere the absorbed dose rate in tissue is approximately only 20 % below the air kerma in the energy range 0.1 to 1 MeV (see Golikov and Brown, 2003). Furthermore, GAKRs tend to be measured at 1 m above ground whereas the organisms of interest are normally located at ground level. However, according to ongoing work (ICRP, in press) the dose rate does not decrease greatly within the first few metres above ground especially for gamma photons with energies exceeding 50 keV. Therefore, it is likely that GAKR will represent a good approximation of the external dose rate.

The (Total) predicted external dose rates for on-soil organisms at Krokhal vary from approximately 1 to 2 µGy h^{-1} (assuming a ratio DW/FW of 0.5 for soils). Predictions are compared to measured GAKRs at the Vodnyi sites in Table 5.9.

Table 5.9 Predicted external dose rates (µGy h^{-1}) to reference organisms at Vodnyi sites compared with Gamma air kerma rates (µGy h^{-1}) across all three sites.

Site	**Total dose rate** (µGy h^{-1}) **Predicted range***	**Gamma air kerma rate** (µGy h^{-1}) **Weighted Mean**	**Min. – Max.**	**Comments**
Krokhal	1.0-2.0	0.65	0.2-18	Typical range: 0.5-2.5
Obzhig	2.0-3.6	2.6	0.2-30	
Otvally	3.5-6.5	-	0.7-35	

- Not available *Corrected assuming a DW/FW ratio of 0.5

In view of the limitations associated with such comparisons, as described above, the predicted external dose rates for the ERICA suite of on-soil reference organisms at Krokhal, Obzhig and Otvally compare favourably with the expected absorbed dose rates to organisms in the field as inferred from air measurements (GAKR). Ra-226 is the main contributor to the predicted external dose rates at all three sites (*ca*. 90 % at Krokhal and Otvally, but only 60 % at Obzhig where the contribution from ^{228}Th is higher).

Predicted vs. observed effects at Krokhal (Ra site 1)

A summary of the reported effects from the ERICA Tool for the predicted dose ranges in Krokhal are summarised in Table 5.10.

Comparison is only possible between the Mammal (rat) reference organism predictions and the *in situ* data for small rodents. The input (i.e. soil) data used for this scenario were sampled in 1973. This approximately coincides with effects studies that were conducted on Tundra voles (*Microtus oeconomus*) in 1975. In this period (minor) chromosome aberrations were detected in some biological samples. Furthermore, the number of anomalous erythrocytes in some rodents appeared to be elevated in contaminated areas (Materyi et al., 2003).

Consideration of the summary table, in contrast, informs that no experiments have established negative effects on mortality or morbidity for the predicted dose rate range for mammals. The discrepancy between predicted and observed effects may arise for several reasons, for instance dose estimations may be erroneous. Furthermore, the lack of underlying data in the effects database (no studies are recorded wherein cytogenetic effects were reported for the dose rate band of interest) limits rigorous comparison.

A comparison with background dose rates shows that the predicted values are orders of magnitude above the natural ranges.

Predicted vs. observed effects at Obzhig (U-Ra site 1)

Biological effects in Tundra voles were studied in 1975, a period that approximately coincides with the year for which the assessment was conducted. At this site chromosome aberrations were documented including the elevated appearance of “bridges” and “fragments” (Materyi et al., 2003). The predicted dose rate for Mammal (Rat) representing the most suitable reference organism for analyses corresponds to 72 µGy h^{-1}, an exposure level that is placed within the Tools Tier 2 effects table band in the category 50-100 µGy h^{-1}. This is notably one band above the predicted effects range predicted at Krokhal. In this dose rate range effects on the reproductive capacity of small mammals, notably mice, have been observed in the form of reduced numbers of offspring sired and weaned, and in a moderate increase in the sterility of pairs. The predicted and observed data sets are not in contradiction but neither does the prediction, or appearance, of the one assure the presence of the other, i.e. the link between the effects on these two quite different end-points is not immediately apparent. If an assessor is provided with a summary table predicting moderate impacts on selected endpoints relating to reproductive capacity it is not possible to directly infer that effects on an arbitrary suite of mutation endpoints will be manifested (or *vice versa*). A validation of the predictions concerning effects is therefore not possible in the strict sense originally anticipated.

Table 5.10 Summary table of effects from the ERICA Tool for dose rate ranges estimated for Krokhal.

Organism	Type of effect	Comment
Amphibian		No data
Bird and bird egg	MB, RC	Increase of infestation of parasites, no effects on reproduction endpoints
Detrivorous insect		No data
Flying insect	MUT	Numerous cytogenetic studies: no effects observed
Gastropod		No data
Grasses and Herbs*	MUT, MT, MB, RC	Numerous data entries, cytogenetic effects observed in numerous plant species, effects on survival of some species noted
Lichens and Bryophyte		No data
Mammal (rat and deer)	MB, MT	No effect on body weight, increase in lifespan
Reptile		No data
Shrub		As for Grasses and Herbs
Soil invertebrate		No data

MB- morbidity; RC- reproduction; MUT- mutation; MT- mortality

Predicted vs. observed effects at Otvally (U-Ra site 2)

The Otvally site is of particular interest because data on radiation effects on plants have been collated for this site. Effects on bird vetch (*Vicia cracca*) were studied in the period 1980-81 (Popova et al., 1984, 1985; Bondar and Popova, 1989). Reported effects in these studies comprised various cytogenetic disturbances and lower seed weights in exposed populations (compared to controls). Although performed slightly later than the period for which our case study was conducted, the site specific soil activity concentration of ^{226}Ra reported for the 1980-81 studies were 3700 Bq kg^{-1}, approximately 10% of the weighted mean used as the input for the present assessment (see Table 5.8). Due to this discrepancy, predicted and observed effects may not be directly compared. The following text will therefore deal with predicted effects using soil activity concentrations shown in Table 5.8

The predicted weighted total dose rate to "Grasses and Herbs", the reference organism that most closely represents the studied flora *Vicia cracca*, lies at circa 460 µGy h^{-1} (as shown in Figure 5.6). Accessing the appropriate dose rate range in the Tools summarised effect table, *i.e.* 400-600 µGy h^{-1}, provides the assessor with a comprehensive summary of data for various plant types. At this dose-rate level one might infer that significant effects on mutation and morbidity endpoints might be expected. Effects on reproductive capacity are also reported in the summary tables for dose rates below 400 µGy^{-1} but these are either relate to observations of no effects or to effects that might not be considered deleterious, e.g. at 45 µGy h^{-1} a slight increase in yield was observed for potato in one reported study. The summary tables for plants are fairly detailed and provide a slightly confusing view of effects that may be expected – in some cases evidence appears to be contradictory, for instance, compare, no effects of frequency of pink mutant cells in stamens of spiderwort at 45 µGy h^{-1} with the presence of various mutation effects for other endpoints and species at lower dose rates. The summary tables reflect the true nature of the available effects data, i.e. *ad hoc* studies that consider different endpoints and species which often exhibit quite different radio-sensitivities. However, providing a synthesised, holistic view on the implication for these experimental observations on predictions for *in situ* assessments is far from straightforward.

5.5.4 Tier 3

A decision was made to limit the assessments at Tier 3 to one of the contaminated sites, Krokhal, and to one reference organism; "Mammal (Rat)". Mouse-like rodents are the most common animals at Krokhal, and measurements of activity concentrations of ^{238}U, ^{232}Th, ^{226}Ra in Tundra voles are available (Materyi et al., 2003). Approximately 80 % of the small rodents at Krokhal are Tundra Voles.

Soil concentrations

Due to the lack of statistical data, such as standard deviations for soil activity concentrations at the site, exponential distributions were assumed for all radionuclides. Weighted mean values from Table 5.8 were used as input values. Results are shown in Table 5.11. The predicted 95th percentile seems to agree fairly well with the maximum soil activity concentrations measured at Krokhal (see Table 5.8).

Table 5.11 Predicted statistical information for the activity concentration of soil at Krokhal assuming exponential distributions.

	Predicted activity concentrations in soil (Bq kg^{-1} AW)			
Nuclide	**Mean**	**5th percentile**	**Median**	**95th percentile**
U-238	48	2.6	33	140
U-234	44	2.4	30	130
Th-230	100	49	690	3100
Ra-226	10000	510	7000	31000
Po-210	8000	400	5600	24000
Th-232	26	1.3	18	79
Th-228	1500	76	1000	4400

Comparison of predicted external dose-rates versus measured GAKRs

It is not possible to derive statistical information for total external dose-rates at Tier 3. However, according to Tool outputs at Tier 2, external dose-rates at Krokhal are dominated by contributions from ^{226}Ra (about 90 %). The distribution of Total external dose rates at Krokhal will, correspondingly, be close to external dose rates estimated from ^{226}Ra activity concentrations in soil. At Tier 2, expected external dose rate for on-soil reference organisms were 1-2 µGy h^{-1}. Assuming 100 % on-soil occupancy for "Mammal (Rat)" (the Tool default for this reference organism being 100 % in-soil) the following statistics were obtained: Mean: 1.7 µGy h^{-1}; Median: 1.2 µGy h^{-1}; 5th percentile: 0.086 µGy h^{-1}; 95th percentile: 5.2 µGy h^{-1}.

For comparison purposes, a gamma irradiation scheme for the entire Krokhal site is shown in Figure 5.7.

Assuming default occupancy factors (i.e. 100 % in-soil) external dose rate to "Mammal (Rat)" increases by a factor of 2.5.

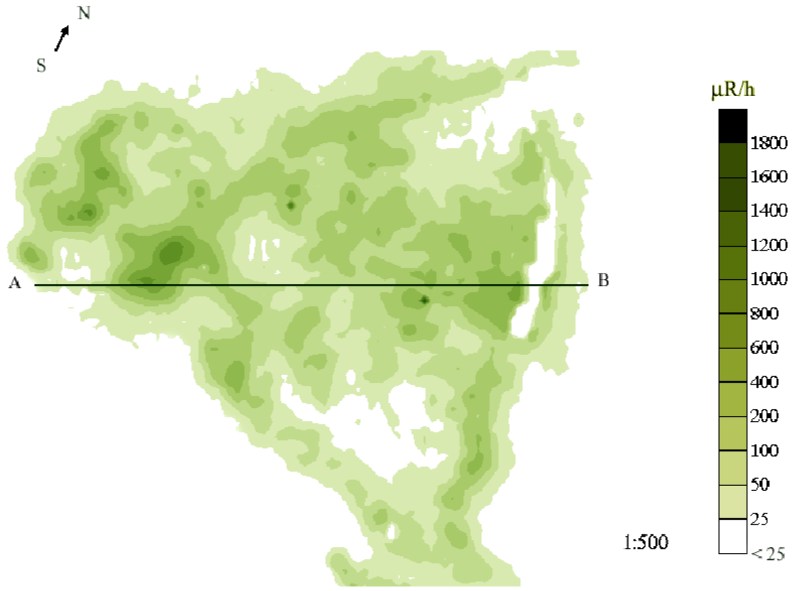

Figure 5.7 Absorbed dose-rates to air (µR h^{-1}) for the contaminated site of Krokhal (early 1970s) (Source: IOB). 1 µR h^{-1} ≈ 0.01 µGy h^{-1} (see Section 5.5.4).

Testing transfer parameters

Unfortunately, only limited data are available for activity concentrations of natural radionuclides in small mammals at the Krokhal site (or any of the other contaminated sites at Vodnyi). As shown in Table 5.12 only (assumed) mean values of (assumed) whole-body concentrations of ^{226}Ra, ^{238}U and ^{232}Th are available for Tundra voles (Materyi et al., 2003). As evident from Table 5.12, ^{226}Ra activity concentrations seem to be over-predicted by the assessment Tool (based on the soil data). However, the measured activity concentrations are within the range of the 5th - 95th percentiles. The opposite is observed for ^{238}U and ^{232}Th, where there seems to be an under-prediction by a factor of 30 (^{238}U) and 300 (^{232}Th). Both observed means are well in excess of the 95th percentile for the predicted activity concentrations.

The explanation for this discrepancy must be speculative owing to lack of knowledge about the derivation of certain values. In particular, there is no description relating to whether the data represent whole-body measurements. This has implications as the CR default data derived in ERICA are often related to whole-body as oppose to organ or muscle specific values. On the other hand, contributions of ^{238}U and ^{232}Th to total dose are insignificant and thus these discrepancies in predicted and measured transfer will not influence the total dose rate to any substantial degree. In contrast, activity concentrations of ^{226}Ra will have a significant effect on the dose rates received.

Table 5.12 Statistical information for predicted and measured activity concentrations (Bq kg^{-1} FW) in Tundra voles.

	Activity concentration Bq kg^{-1} FW						
	Predicted			Measured			
Nuclide	Mean (Median)	5th percentile	95th percentile	n	Mean	SD	Range
U-238	5.0×10^{-3} (2.1×10^{-3})	1.2×10^{-4}	1.9×10^{-2}	-	0.16	-	-
U-234	4.6×10^{-3} (1.9×10^{-3})	1.1x10-4	0.017	-	-	-	-
Th-230	0.12 (0.044)	2.1×10^{-3}	0.49	-	-	-	-
Ra-226	270 (100)	5.4	1040	-	36	-	-
Po-210	22 (13)	0.88	74	-	-	-	-
Th-232	3.2×10^{-3} (1.1×10^{-3})	5.4×10^{-5}	0.013	-	0.99	-	-
Th-228	0.18 (0.066)	3.2×10^{-3}	0.73	-	-	-	-

- Data not available

Total dose rates

Based on results at Tier 2, ^{226}Ra contributes 97 % of the total dose to the Mammal (Rat) reference organism at Krokhal. Therefore, it is not necessary to include any other radionuclides in the process of deriving a reasonable dose estimate.

Two cases we considered:

Case 1: Weighted mean data for ^{226}Ra in soil. Assume that soil concentrations are exponential distributed (insufficient statistical data for soil concentrations). Default distributions of concentration ratio parameters from the assessment Tool are implemented.

Case 2: Weighted mean data for ^{226}Ra in soil + site specific data for Tundra voles (^{226}Ra). Since also Tundra vole data are limited an exponential distribution is assumed.

The results of the two cases are shown in Figure 5.8. The predicted Total dose rate is somewhat lower for Case 2, which should be evident from the data in Table 5.12 (lower mean activity concentration in Tundra vole). The 5th and 95th percentile range are within the corresponding range for Case 1. Large variance for Case 1 is due to the large range in soil activity concentrations at Krokhal in combination with a large variance for default CRs used for this case.

Evaluating effects using FREDERICA

Using the 95th percentile of Figure 5.8 as an upper value, 75 references were found for mammals using the FREDERICA database. Most references seem to concern mice or rats, a small fraction dealing with larger mammals. Two errors were noticed, these were reference ID 319 (for molluscs) and ID 377 (for plants).

The analyses of data from the FREDERICA database did not aid the assessment process to a level commensurate with expectation. The data in the format provided are not easily synthesised for a non-specialist and the assessor is left in a position that is arguably less clear than when the process began. The FREDERICA database is undoubtedly a well constructed and useful research tool, but its efficacy as a tool to aid decision making, at a more practical level, is less convincing.

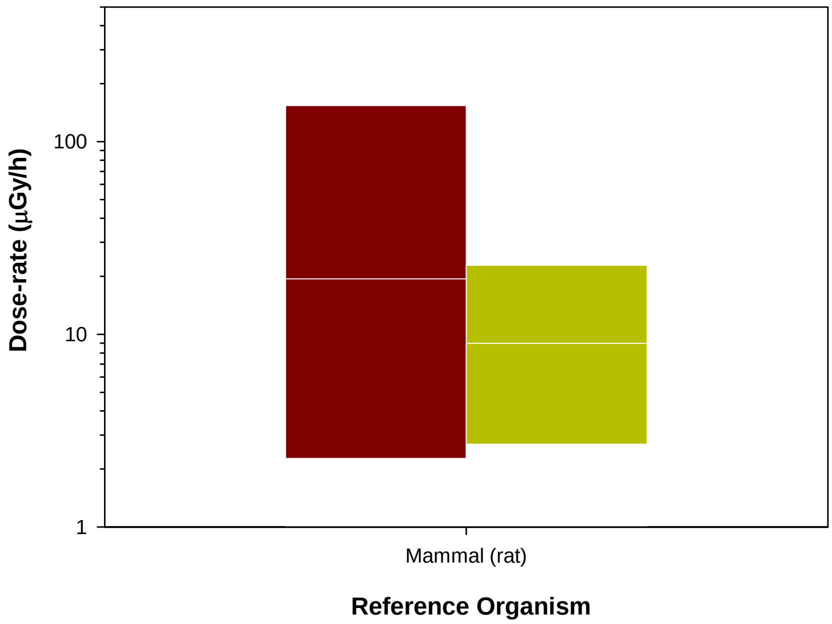

Figure 5.8 Total dose rates (µGy h^{-1}) for Mammal (Rat) reference organisms. The boxes indicate 5^{th} percentile, median and 95^{th} percentile. Brown=Case 1; green=Case 2.

5.6 General conclusions and comments

For the contaminated areas in Vodnyi, the Tool appears to be giving sensible results in the sense that the expected and conservative value risk quotients are >> 1 in all cases, indicating that the levels of contamination may be having a detrimental impact on the environment and that further analyses is recommended. In view of the fact that effects have been observed in the Vodnyi environment for the period from which input data were generated, an appropriately "calibrated" calculation methodology would be expected to flag this area as one that exceeds screening dose-rates. The other sites in the case study, Middle Timan and North Urals, also yield RQs > 1 at both Tier 1 and 2. These are both areas of naturally enhanced radiation background and as such might be treated differently by an assessor. The facts that the predicted dose-rate ranges are: (i) generally within an order of magnitude of the upper range values documented in the Tool for generic background (where this is not the case it can be argued that the reference organism is relatively radio-resistant); (ii) only exceed the ERICA dose rate screening level for high percentile dose-rate predictions; (iii) plants and animals have adapted to live in these high radiation areas, suggest that an assessor might not evaluate the exposure levels to be a cause of substantial environmental detriment at these sites. If the assessment is to estimate a dose rate excess (i.e. dose rate arising above that occurring naturally), it is clearly important to characterise local background radiation levels.

Chemical toxicity of ^{238}U and ^{232}Th are not considered in the ERICA methodology. This may lead to an underestimation of risk (*e.g.* at Komi-sites with U-Ra contamination). The simplest solution to this is probably to add a few lines of text in the Help flagging the limitation in applying EMCLs for long-lived radionuclides from a radiotoxicity perspective only. Application of the methods presented in EC (2003) may allow predicted no effects concentrations to be derived explicitly for radionuclides like ^{238}U and ^{232}Th. These values might be used to replace the EMCL if this was deemed appropriate.

Measured activity concentrations seem to be within the range of the predicted concentrations of ^{226}Ra in "Grasses and Herbs" and "Shrub" for Middle Timan and North Urals using default CRs. However, "Tree" concentrations of ^{226}Ra seem to be under-predicted by the ERICA assessment Tool and there seems to be a general over-prediction of ^{238}U and ^{232}Th activity concentrations using default CRs in most cases. The default transfer databases used by the Tool could be updated using the substantial empirical data bases compiled for this case study at a later date. Predictions of external dose rate to reference organisms made using the Tool compare favourably with data from direct measurement of gamma-air kerma rates making allowances for the fact that the two measurement endpoints are different.

In view of the limitations in relating effects observed for one set of endpoints to those observed for another set, the analyses conducted in the case study cannot provide any definitive validation of the effects prediction provided by the Tool beyond the observation that, for this particular case, the predictions made are not in contradiction to the observed effects in the field. The effects summary tables at Tier 2 are not as informative as expected: this undoubtedly reflects the paucity of data for some organism groups within some dose rate bands. In many cases, there is no information at all (*e.g.* "Lichen and Bryophytes" and "Reptile"). For "Grasses and Herbs", there was some difficulty in extracting useful information: the summary table for the dose rate range of interest consists of numerous studies looking at a variety of various endpoints, predominantly cytogenetic in character, for various species. This results in confusion more than clarification. Future versions of the Tool might be best supported by adopting a simpler, synthesised view of effects data.

Considering the high concentrations of ^{226}Ra in these contaminated sites and the habitual utilisation of burrows by organisms such as Tundra voles, the dose rate attributable to the inhalation of ^{222}Rn might be considerable. The current ERICA methodology provides no way of assessing the importance of this particular exposure pathway. Furthermore, in many studies involving the assessment of impacts of radionuclides on man and the environment GAKR data will be readily available. The detail of the information presented in this case study, based on real data sets available for the sites of interest are a "case in point". At present there are no means of utilising such information in the ERICA Tool.

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6 Chernobyl Exclusion Zone

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6.1 Introduction

In our earlier assessment of the application of the FASSET framework (Beresford and Howard, 2005) we highlighted the inability to compare predictions of absorbed dose with measured data. To enable a more complete assessment of the ERICA Tool, a study was performed during the summer of 2005 to estimate the external gamma dose rates received by various small mammal species within the Chernobyl exclusion zone using attached thermo-luminescent dosimeters. Here, we concentrate on testing the ERICA Tool against these data.

The database compiled for the Chernobyl case study application of the FASSET methodology (Beresford and Howard, 2005; Beresford et al*.*, 2005) has subsequently been adapted for use as an inter-comparison exercise of the IAEA's EMRAS programme Biota Working Group (BWG) (see Beresford et al., 2006). Predictions of the ERICA Tool from this exercise are also discussed.

6.2 Estimating the exposure of small mammals

6.2.1 Study sites and determination of soil activity concentrations

Three forest sites anticipated to have differing soil radionuclide activity concentrations were selected in the Chernobyl exclusion zone. The sites will be referred to throughout as *Low*, *Medium* and *High* on the basis of their anticipated soil activity concentrations. A 100 m x 100 m study area was marked out at each site.

The *Low* site was located approximately 8.5 km south-east of the Chernobyl reactor number 4 (Figure 6.1). The dominant tree species at this site was *Pinus sylvestris* (Scots pine) with very few deciduous trees. Most of the 10 000 m^2 study area had sparse understorey vegetation although towards the east side there was complete ground cover dominated by graminaceous species. The *High* and *Medium* sites were approximately 5 km and 8 km respectively to the west of the Chernobyl power plant complex (Figure 6.1). The *Medium* site consisted of mainly *P. sylvestris* and *Querqus robur* (Oak) with some *Sorbus aucuparia* (Rowan) and *Tilia platyphyllos* (Large leaved lime), the sparse understorey vegetation included *Pteridium aquilinum* (bracken). The *High* site was dominated by *Betula* spp., with no coniferous species present. This site had a ground cover consisting predominantly of graminaceous species throughout, although there was also some *Calluna vulgaris* (heather).

Over each 10 000 m^2 study area, a grid, was marked out using posts at 10 m intervals. These were subsequently used as the location of a small animal traps. Gamma kerma rates were determined at 5 cm above ground surface at each trapping site using a MKS–01R–01 dose rate meter with a BDKB-01R detector. A stand was used to achieve the same height at each location and the mean of three 10 second measurements was recorded. The location of each trapping site was determined using a handheld GPS.

Using a random sampling scheme, twenty-three soil samples were collected from each site. Samples were taken to a depth of 10 cm and sampling locations recorded using a handheld GPS. The soil sampling area was extended to 50 m beyond the trapping area to encompass the likely home ranges of the species being trapped (see The Mammal Society, 2006). Samples were subsequently, dried and homogenized. Sub-samples were analysed on hyper-pure (Canberra-Packard) germanium detectors to determine the activity concentration of gamma-emitting radionuclides, spectra were analysed using the Canberra-Packard Genie-2000 software package. Count times were such that an error of < 20 % on the ^{40}K determination was achieved. For samples from the *Medium* and *Low* sites the mass analysed was approximately 750 g dry weight (DW) per sample and that for the *High* site was approximately 130 g (DW) per sample.

Figure 6.1 Location of the study sites relative to the Chernobyl nuclear power plant (NPP). Photo adapted by Simon Wright (CEH) from the original with the kind permission of Valery Kashparov of the Ukranian Institute of Agricultural Radiology (UIAR, 2001).

The ^{238,239,240}Pu activity concentrations soils were determined in 10 g (DW) sub-samples using the method described by Bondarkov et al. (2002a). Strontium-90 activity concentrations in soils were determined in 10 g (DW) sub-samples by the method described by Bondarkov et al. (2002b; 2002c).

6.2.2 Small mammal trapping, whole-body counting and TLDs

One hundred Sherman humane traps were placed over each sampling area (at the marker posts as described above) and baited with rolled oats overnight. Only *Apodemus flavicollis* (Yellow-necked mouse), *Clethrionomys glareolus* (Bank vole) and *Microtus spp*. (Vole species) were processed for this study. Other trapped species were released. Animals to be processed were transported to a laboratory in the town of Chernobyl. Trapping occurred on 14 occasions over the period from early July to mid-August 2005. The trapping location of each animal was recorded.

The first time an animal was caught it was fitted with a numbered collar to which a LiF-100 TLD (Global Dosimetry Solutions Inc., California) had been attached. The collar comprised a 4 mm wide cable-tie; the TLD was attached to this using electricians tape having first been covered in a single layer of polythene (cut from the corner of a 200 gauge polythene bag). The live-weight of the animal was recorded and its whole-body ^{137}Cs and ^{90}Sr content then determined using a method based upon that described by Bondarkov et al. (2002b; 2002d). The animals were placed in a small, disposable, cardboard box (70x40x40 mm) the upper side of which was made from <0.1 mm thick polyethylene prior to whole-body counting. The box was then placed inside a lead shielded counting container. The detectors comprised a hyper-pure germanium detector and thin-film (1 mm) NaI scintillation detector to measure ^{137}Cs and ^{90}Sr respectively. The ^{137}Cs spectra was analysed using the Genie-2000 software package. The activity concentration of ^{90}Sr was determined from that of its daughter nuclide, ^{90}Y. The method has been calibrated against phantoms containing ^{137}Cs and ^{90}Sr, and ^{90}Sr results validated against traditional radiochemical extraction and analyses (Bondarkov et al., 2002b). The duration of count times varied from 150 to 1200 seconds depending upon the activity content of the animal.

Following live-monitoring, the animals were each returned to the individual trapping location from which they were caught and released. If an animal was recaptured more than 14 d after being fitted with a TLD-collar the TLD was removed, the animal reweighed and its whole-body ^{90}Sr and ^{137}Cs activity concentrations measured again. If it was recaptured less than 14 d after having the collar fitted, the trapping location was recorded and it was released (on some instances additional whole-body Cs measurements were also made). In the last two weeks of the study TLDs were removed if an animal was recaptured within 6 days of the TLD having been fitted. A total of 230 TLD-collars were fitted to animals of which 85 were recovered; the time recovered TLDs had been on the animals ranged from 6 to 36 days. Seven TLDs mounted on collars were transported to the Chernobyl laboratory and left there for the duration of the study as controls.

At four randomly selected trapping locations within each sampling site TLDs were placed 5 cm above ground level, at ground level and 10 cm deep within the soil. In each position, one TLD was prepared in the manner of those attached to the collars and a second was additionally encapsulated within a 2x2x2 cm cube of Perspex. These were left at the study sites for the duration of the experiment (N.B. only 25 of the possible 36 paired TLDs were recovered).

The TLDs recovered from small mammals and the study sites were returned to the supplier for analyses together with the controls.

6.2.3 Results

Soil activity concentrations of ^{90}Sr, ^{238,239,240}Pu, and those gamma emitting radionuclides which were detectable, are summarised in Table 6.1. Where activity concentrations below the detection limit are shown within the table, a value of half the detection limit has been used to derive the mean estimate. The mean percentage dry matter contents of the soils were 97 %, 88 % and 87 % at the *Low*, *Medium* and *High* sites respectively.

The trapping success and animals species trapped varied across the three sites. Table 6.2 presents the numbers and species of animals from which TLDs were recovered at each of the sites; the predominant species differed at each site. Trapping success at the *Low* site was reduced because of continued interference with the traps by wild boar (*Sus scofa*).

Whole-body ^{137}Cs and ^{90}Sr activity concentrations determined in animals from which TLDs were recovered are summarised by species in Table 6.2; the mean of measurements made for individual animals (each animal being live-monitored at least twice) were used to estimate the summary values.

Table 6.1 Activity concentrations determined in soil samples collected over each sampling area (n=23 from each sampling site).

	Soil activity concentrations (kBq kg^{-1} DW)								
Site	^{134}Cs	^{137}Cs	^{90}Sr	^{40}K	^{60}Co	^{241}Am	^{154}Eu	^{155}Eu	$^{238,239,240}Pu$
Low									
Mean	0.007	7.37	2.20	0.19	<0.004	0.21	0.04	0.02	0.13
SD	0.005	4.21	1.10	0.05		0.15	0.02	0.005	0.14
Min.	<0.004	1.70	0.85	0.14		0.04	<0.05	<0.025	<0.02
Max.	0.02	23.72	5.99	0.33		0.65	0.12	0.03	0.68
Medium									
Mean	0.09	43.29	18.55	0.09	0.02	1.47	0.19	0.06	0.83
SD	0.21	25.72	14.90	0.05	0.02	2.48	0.16	0.07	1.49
Min.	0.0005	12.61	1.84	0.00	<0.004	0.01	<0.05	<0.025	<0.02
Max.	1.05	115.34	61.10	0.20	0.09	11.65	0.56	0.26	7.35
High									
Mean	0.10	97.74	56.53	0.08	0.07	3.20	0.52	0.20	1.47
SD	0.05	41.81	38.96	0.02	0.08	4.59	0.63	0.24	2.02
Min.	0.001	27.52	7.43	0.04	<0.004	0.03	<0.05	<0.025	0.08
Max.	0.22	208.36	165.14	0.15	0.39	19.21	3.20	1.17	9.79

Dose rates as determined from measurements of the TLDs recovered from trapped animals are presented in Table 6.3 together with the gamma kerma rates determined for each site. No measurable dose rates were recorded on any of the control TLDs. The mean ratio of dose rates between the TLDs without and with a 2 cm Perspex covering placed at various heights above, and depths below, the soil surface at 16 sampling locations was 1.95±0.75 (n=25). There was no trend in this ratio either between study areas or with position above or below the soil surface.

6.2.4 ERICA predictions

Organisms to represent each of the three study species was generated within the ERICA Tool. The ERICA terrestrial mammal concentration ratios (CRs) for ^{90}Sr and ^{137}Cs of 1.74±2.35 and 2.88±4.25 (mean±SD) respectively were assumed to be applicable to each of the species. The live-weights of the three species, as determined during the study, were similar. Therefore, the average live-weight of 30 g was used for all species together with dimensions, based on measurements made from animals trapped in the area previously (Gaschak *pers comm.*), of 8 cm long, 3.5 cm in height and 3.5 cm in width to create a small rodent geometry. It was assumed that both *C. glareolus* and *Microtus* spp. spend 30 % of their time on the soil surface and 70 % underground; *A. flavicollis* were assumed to spend equal amounts of time above and below ground.

Whole-body activity concentrations

Whole-body ^{137}Cs and ^{90}Sr activity concentrations have been estimated probabilistically using the Tier 3 level of the ERICA Tool and assuming that soil activity concentrations as presented in Table 6.1 were log-normally distributed. Predicted activity concentrations are compared to live-monitoring measurements in Table 6.3; as CRs for the three species were assumed to be the same only one prediction is presented for each site. For most comparisons, measured whole-body activity concentrations are between the predicted 5th and 95th percentiles. Exceptions are: (i) both *C. glareolus* measurements for ^{137}Cs at the *High* site are in excess of the predicted 95th percentile value; (ii) the maximum measured ^{137}Cs activity concentration for *Microtus* spp. at the *High* site is in excess of the 95th percentiles; (iii) the maximum measured ^{90}Sr for *A. flavicollis* at the *Low* site is in excess of the

95th percentile prediction. All measured values are below the maximum predicted values (not presented here).

A further prediction was made for the *High* site using CR values from the ERICA database for rodent species only of 5.51±6.14 (n=122) for Cs and 0.33±0.31 (n=47) for ^{90}Sr. The resultant mean estimated ^{137}Cs whole-body concentration was 517000 Bq kg^{-1} fresh weight (FW) with 5th and 95th percentiles of 65400 and 1620000 Bq kg^{-1} respectively. Whilst this was in better agreement with the observed data for the two vole species than predictions using the ERICA default CR value for Cs (derived from all mammal species with the exception of reindeer), predictions for ^{90}Sr were considerably lower. The predicted 95th percentile was 60200 Bq ^{90}Sr kg^{-1} (FW) which was lower than the mean of observed values for all three species. Consequently, the use of rodent specific CR values instead of the generic mammal values could not be justified within this comparison.

External dose rates

For comparison with predictions from the ERICA methodology it has been assumed that the TLD measurements equate to estimated external gamma dose rates. Two comparisons have been performed: (i) predictions made for each individual animal for which a TLD result was available; (ii) probabilistic estimations using Tier 3 of the ERICA Tool.

To make predictions for the 85 individual animals from which TLDs were recovered the dose conversion coefficients (DCCs) generated by the ERICA Tool for the small rodent geometry, as defined above, were implemented within a MSExcel workbook. There were two reasons for adopting this approach rather than using the ERICA Tool directly: (i) the Tool version being used did not have DCC values for ^{40}K and ^{155}Eu although the values to be subsequently implemented in a revised Tool version were made available for use; (ii) the exercise would have required 85 separate runs of the ERICA Tool and hence implementation in MSExcel was more efficient for this purpose (note this is not a criticism of the Tool as it is not the purpose for which it was designed). Dose conversion coefficient values for the ERICA default 'mammal (rat)' geometry were used for ^{40}K and ^{155}Eu as these radionuclides were not available within the Tool and hence DCC values for the small rodent geometry were not generated. A comparison, for the radionuclides available within the Tool, of DCCs for the default mammal (rat) geometry and the case specific small rodent geometry showed little difference in the external beta-gamma DCCs (e.g. the external DCC for ^{137}Cs for animals in soil was 7% higher for the small rodent geometry than the default mammal (rat) geometry). Twelve individuals were picked at random for a comparative estimation of external dose rate using the ERICA Tool as discussed below.

Table 6.2 A comparison of measured ^{90}Sr and ^{137}Cs whole-body activity concentrations with those predicted using Tier 3 of the ERICA Tool.

Species/ area	n	Measured whole-body activity concentration (Bq kg^{-1} FW) Mean	Min.	Max.	Predicted whole-body activity concentration (Bq kg^{-1} FW)$^{+}$ Mean	5th percentile	95th percentile
^{137}Cs							
Low							
C. glareolus	3	3820	3140	4660	19900	1500	69500
A. flavicollis	18	3130	1270	9750			
Medium							
C. glareolus	39	70500	17000	252000	116000	8310	411000
A. flavicollis	10	59700	24100	143000			
High							
C. glareolus	2	2260000	1350000	3180000	268000	21600	952000
Microtus spp.	11	611000	252000	1140000			
A. flavicollis	2	145000	108000	183000			
^{90}Sr							
Low							
C. glareolus	3	7710	3050	10300	3940	292	13300
A. flavicollis	18	7410	1390	21100			
Medium							
C. glareolus	39	19500	4290	36000	33400	1940	121000
A. flavicollis	10	24700	16000	34000			
High							
C. glareolus	2	81300	65600	96900	102000	6550	357000
Microtus spp.	11	107000	38100	167000			
A. flavicollis	2	66600	46600	86700			

$^{+}$Predicted value applies to all three of the study species

To derive animal specific soil activity concentration inputs, spatial interpolation of the data were attempted using block kriging (Karssenberg and Burrough, 1996). However, whilst variable, the data demonstrated no significant spatial trend at any of the three sites. Therefore, the average activity concentration was determined for all soil samples falling within a 30 m radius of each trapping location (a 30 m radius was considered as representative of likely home ranges of the three species (The Mammal Society, 2006)). If an animal had been caught in more than one trap weighted soil activity concentrations were derived, *e.g.* if a animal were caught three times in trap number 8 and twice in trap number 18 the weighted soil activity concentration was estimated as:

$$\frac{[3 \times (acitivy\ concentration\ at\ Trap\ 8)] + [2 \times (activity\ concentration\ at\ Trap\ 18)]}{5}$$

Table 6.3 presents a comparison of predicted dose rates using the ERICA DCCs with the dose rates reported for the TLDs and also the gamma-kerma rates measured at trapping sites. Caesium-137 dominated the predicted external dose rates at all sites comprising more than 98 % of the total dose rate. The contribution to the total dose from when the animals were assumed to be underground were approximately 86 % for both voles species and 72 % for *A. flavicollis*.

The predicted dose rates tend to be lower than the TLD results and in better agreement with the gamma kerma rates. As noted above, the doses rate recorded by TLDs 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. If we assume that this additional dose is the result of exposure to beta radiation (excluded by the perspex) and that it is representative of beta dose rates recorded by the TLDs on the animal collars then we can correct the results from the TLDs attached to the collars (i.e. dividing by 1.95) to derive the external gamma dose rate. The resultant 'corrected' TLD results are presented in Table 6.3. Comparison between the 'corrected' TLD dose rates and predicted external dose rates are improved, especially for the *Low* and *Medium* sites; gamma kerma rates are also in good agreement with predicted external dose rates and 'corrected' TLD dose rates at these two sites. At the *High* site the predicted mean external dose rate is approximately half of the mean 'corrected' TLD value which is in excess of the maximum predicted dose rate. The maximum 'corrected' TLD dose rate at the *Medium* site was in excess of predicted external dose rates. Changing the assumed percentages of time spent underground compared to on the soil surface does not resolve the under-predictions observed; assuming 100 % of time is spent underground only increases estimated dose rates by approximately 40 % for *A. flavicollis* and 20 % for both vole species.

Whilst the mean predictions of dose rate are in reasonable agreement with the 'corrected' TLD measurements individual dose rates are not well predicted as demonstrated in Figure 6.2 for the *Medium* site.

Dose rates estimated for 12 randomly selected individuals using Tier 2 of the ERICA Tool were in agreement with those estimated using the DCCs implemented in MSExcel. For all 12 animals the Tier 2 conservative risk quotient estimates (having input measured ^{137}Cs and ^{90}Sr whole-body activity concentrations) were in excess of one.

Table 6.3 A comparison of dose rates as reported for recovered TLDs, with those predicted (deterministically) using DCC values from the ERICA Tool and gamma-kerma rates determined 5 cm above the soil surface.

Site	Mean	SD	Min.	Max.
TLD dose rate as reported (µGy h^{-1})				
Low	3.07	1.23	0.90	5.14
Medium	27.1	15.4	9.04	100
High	90.7	36.3	46.5	188
External dose rate predicted (µGy h^{-1})				
Low	1.82	1.39	1.19	7.66
Medium	10.8	3.8	4.65	27.2
High	19.9	3.2	15.3	26.0
TLD dose rate 'corrected' (µGy h^{-1})				
Low	1.58	0.63	0.46	2.64
Medium	13.9	7.9	4.7	51.4
High	46.6	18.7	23.9	96.4
Gamma kerma (µGy h^{-1})				
Low	1.96	0.42	0.66	3.38
Medium	11.5	3.2	5.8	20.1
High	31.4	7.9	5.7	52.3

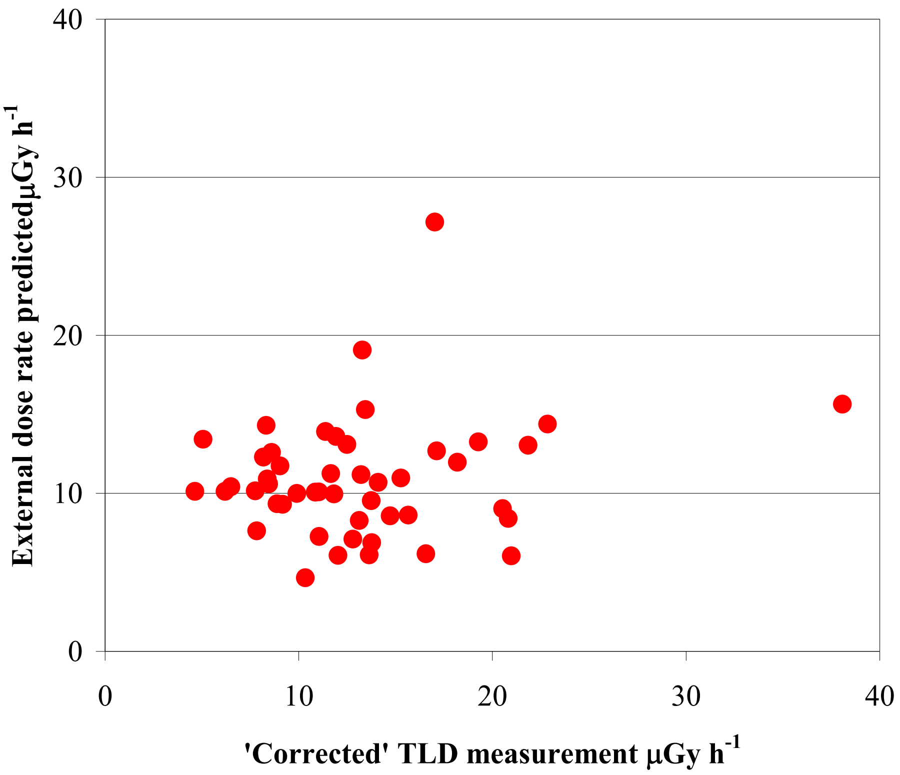

Figure 6.2 A comparison of predicted external dose rates with 'corrected' TLD measurements for animals at the *Medium* site.

For application of Tier 3 of the ERICA Tool the soil activity concentrations as presented in Table 6.1 were assumed to be log-normally distributed. Potassium-40 and ^{155}Eu could not be considered within this assessment (as they were not included within the Tool). However, these two radionuclides contributed less than 1.5 % of the total dose rates presented in Table 6.3 and hence their exclusion is inconsequential. Table 6.4 compares external dose rates estimated using Tier 3 of the ERICA Tool with 'corrected' TLD measurements by study site and species. Predictions for *C. glareolus* and *Microtus* spp. are identical as the assumptions used for the two species were the same.

As for results presented in Table 6.3, predictions compare well for the *Low* site. At the *High* and *Medium* sites differences predicted between the two voles species and *A. flavicollis* were not representative of the TLD results perhaps inferring that the assumptions of different times spent below and above ground were unjustified. For all three species at the *High* site and *A. flavicollis* at the *Medium* site the predicted 95th percentile external dose rates were lower than the mean 'corrected' TLD values.

Table 6.4 'Corrected' dose rates recorded by TLDs compared with external dose rates predicted using Tier 3 of the ERICA Tool.

Site	n	TLD dose rate 'corrected' µGy h^{-1} Mean	SD	Min.	Max.	External dose rate (probabilistic) predicted µGy h^{-1} Mean	5th	95th
Low								
C. glareolus	3	2.11	0.62	1.43	2.64	1.79	0.67	3.67
A. flavicollis	18	1.49	0.60	0.46	2.57	1.52	0.57	3.11
Medium								
C. glareolus	39	13.1	6.21	4.65	38.1	9.13	3.2*	18.7
Microtus spp.	n/a	n/a	n/a	n/a	n/a	9.13	3.2*	18.7
A. flavicollis	10	17.2	12.6	8.87	51.4	7.75	2.72	15.9
High								
C. glareolus	2	66.5	42.3	36.5	96.4	21.2	10.3	36.9
Microtus spp.	11	43.7	14.7	23.9	77.6	21.2	10.3	36.9
A. flavicollis	2	43.2	0.30	43.0	43.4	18.0	8.72	31.3

*Tool outputs results to differing levels of significant figures

6.2.5 Total dose rate predictions – a comparison of Tiers 2 and 3

Within Tier 2 of the ERICA Tool the user can select uncertainty factors to apply within their assessment. An uncertainty factor of three is suggested to give a conservative estimate of absorbed dose rate equivalent to the 95th percentile value. To test this assumption total absorbed dose rates have been predicted using Tiers 2 and 3; Table 6.5 comparing the resultant 'conservative dose rate from Tier 2 with the predicted 95th percentile values from Tier 3. This exercise was conducted for the most numerous species sampled at each of the three field sites. At both Tiers 2 and 3 the calculations were performed using only soil activity concentrations as the input, and again, using soil and available whole-body activity concentrations as the inputs. Within Tier 3, default CR values with associated probability distributions functions were used (when whole-body activity concentrations were not available) and calculations were performed assuming input data was lognormally distributed, and again, assuming exponential distributions. Input activity concentrations were as presented in Tables 6.1 and 6.2; ^{40}K and ^{155}Eu were not included in the assessment as they are not available as default radionuclides within the Tool.

In all instances, the 95th percentile value predicted using Tier 3 of the Tool was similar to, or lower than, the conservative estimate output by Tier 2 (Table 6.5).

Table 6.5 Comparison of total absorbed dose rates for Tiers 2 (conservative prediction assuming an uncertainty factor of 3) and Tier 3 (predicted 95th percentile value).

Species	Site	Total absorbed dose rate (µGy h^{-1})+ Tier 2 Input data Soil	Soil & biota	Tier 3 Input data (lognormal pdf) Soil	Soil & biota	Tier 3 Input data (exponential pdf) Soil	Soil & biota
S. flavicollis	Low	23	21	20	13	23	17
C. glareolus	Medium	158	107	133	57	152	73
Microtus spp.	High	402	585	353	304	394	455

+Note the default mammal (rat) geometry was used for this comparison

6.3 ERICA predictions for the IAEA EMRAS Chernobyl scenario

The Chernobyl scenario used by the IAEA EMRAS BWG (see Beresford et al., 2006) was based upon an expanded database initially compiled for the testing of the FASSET framework early within the ERICA project (Beresford and Howard, 2005; Beresford et al., 2005). A range of terrestrial biota types were selected for the exercise comprising: graminaceous vegetation; invertebrates; birds; wide range of mammal species; amphibians; a reptile (see Table 6.6). 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. All data originate from samples taken after 1992. The scenario instructions present soil activity concentrations (mean with ranges when available) with participants requested to predict activity concentrations and absorbed dose rates. As this exercise is not yet completed by the BWG, discussion here is restricted to a comparison of activity concentrations predicted using the default CR database within the ERICA Tool with available measurements.

Figures 6.2-6.6 present a comparison of predicted and measured whole-body ^{137}Cs activity concentrations in a range of different terrestrial biota. Results are presented normalised to the mean of each observed data point. Where minimum and maximum soil concentrations were provided within the scenario then minimum and maximum whole-body predictions have been made using these values. The majority of predicted values overlap the range in available measurements with mean predicted values being within an order of magnitude of the observed data. No bias on the basis of organism/species between predicted and observed values was evident. Poorer comparisons are most often associated with data points which comprised comparatively few observations (*e.g.* partridge, wild boar, roe deer).

Figures 6.7-6.10 compare predicted ^{90}Sr whole-body activity concentrations with available data in the same manner as for ^{137}Cs above. As for ^{137}Cs, the majority of predicted values overlap the range in available measurements with mean predicted values being within an order of magnitude of the observed data. The few data for ^{239}Pu and ^{241}Am available for Birds and Small Mammals were all predicted to within a factor of 5 of the measured values using the ERICA CR values (Figure 6.11).

Table 6.6 Species (Latin and common names) for which data were available within the IAEA EMRAS Chernobyl scenario.

Latin species name	English species name	Latin species name	English species name
Small mammals		**Birds**	
Apodemus flavicollis	Yellow necked mouse	*Aegithalos caudatus*	Long-tailed tit
Apodemus sylvaticus	Wood mouse	*Capreolus capreolus*	Roe deer
Clethrionomys glareolus	Bank vole	*Erithacus rubecula*	Robin
Microtus arvalis	Common vole	*Hirundo rustica*	Barn swallow
Microtus oeconomus	Root vole	*Parus major*	Great tit
Microtus spp.	Vole species	*Perdix perdix*	Partridge
Sorex araneus	Common shrew	*Sturnus vulgaris*	Starling
Sicista betulina	Northern birch mouse	**Amphibians**	
Large mammals		*Rana esculenta*	Edible frog
Canis lupus	Wolf	*Rana terrestris*	Brown frog
Sus scofa	Wild boar	**Reptiles**	
		Lactera agilis	Sand lizard

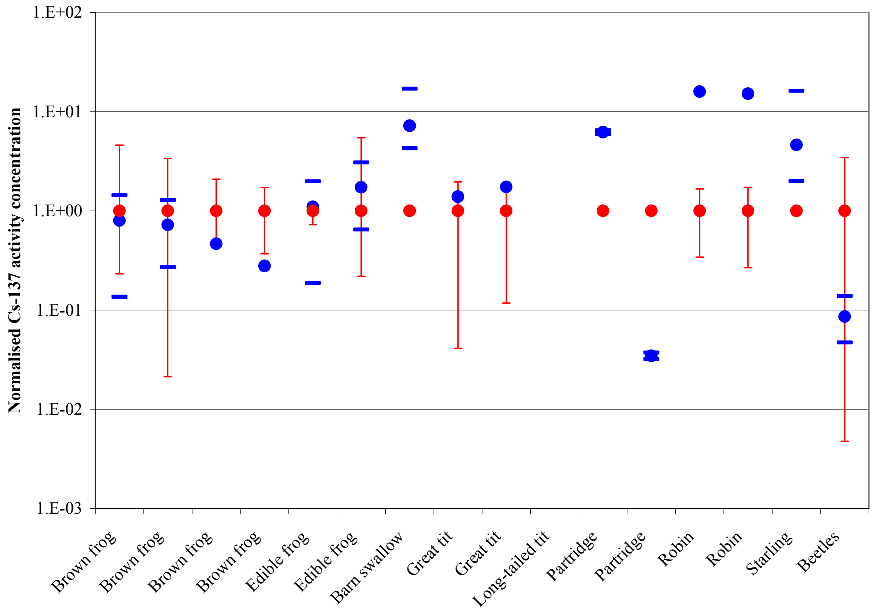

Figure 6.3 Comparison of observed (red) whole-body ^{137}Cs activity concentrations with those predicted using the default ERICA CR values (blue) for amphibian, bird and invertebrate species. Mean, minimum and maximum observed and predicted values are presented when available.

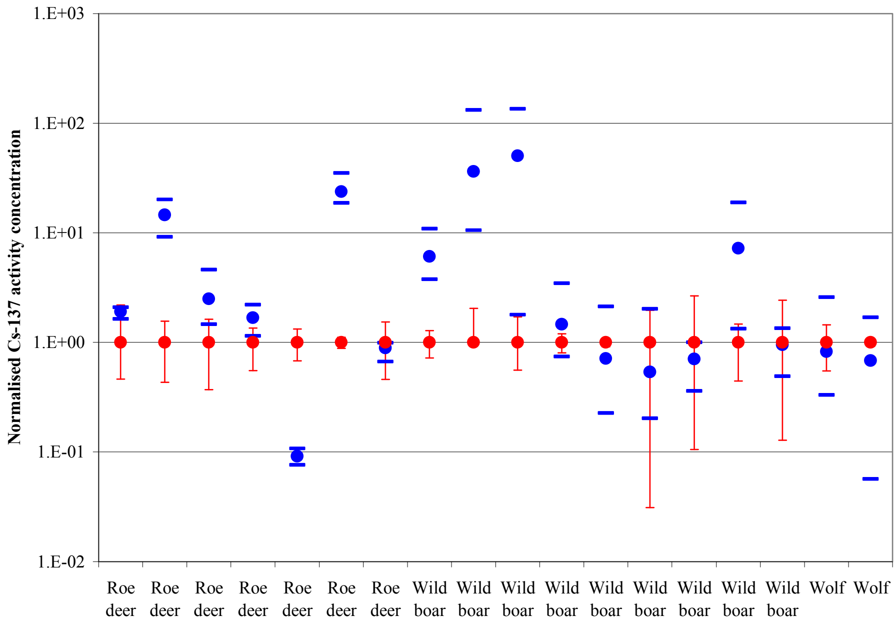

Figure 6.4 Comparison of observed (red) whole-body ^{137}Cs activity concentrations with those predicted using the default ERICA CR values (blue) for large mammalian species. Mean, minimum and maximum observed and predicted values are presented when available.

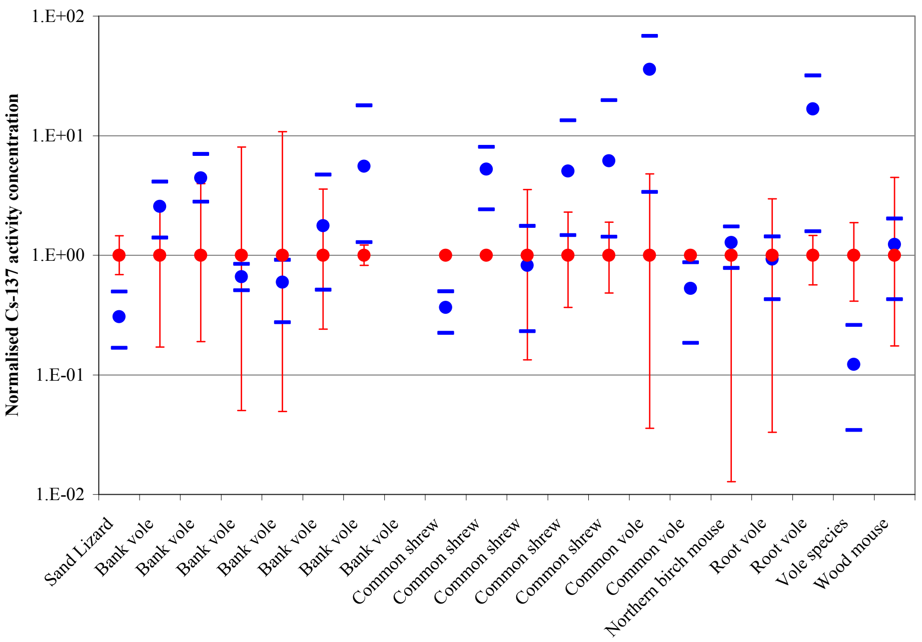

Figure 6.5 Comparison of observed (red) whole-body ^{137}Cs activity concentrations with those predicted using the default ERICA CR values (blue) for small mammalian and reptile species. Mean, minimum and maximum observed and predicted values are presented when available.

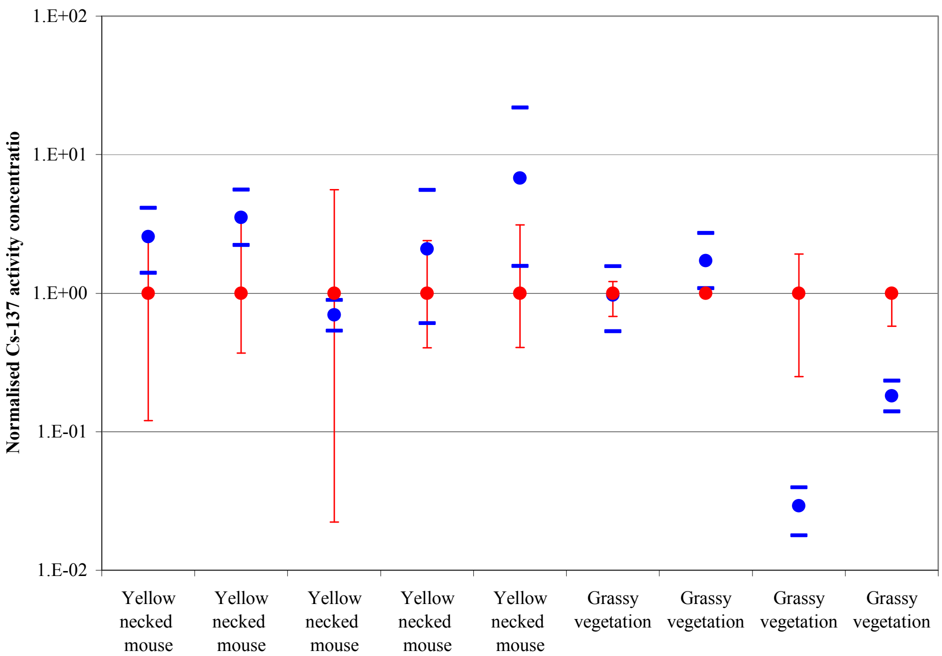

Figure 6.6 Comparison of observed (red) whole-body ^{137}Cs activity concentrations with those predicted using the default ERICA CR values (blue) for small mammalian and vegetation species. Mean, minimum and maximum observed and predicted values are presented when available.

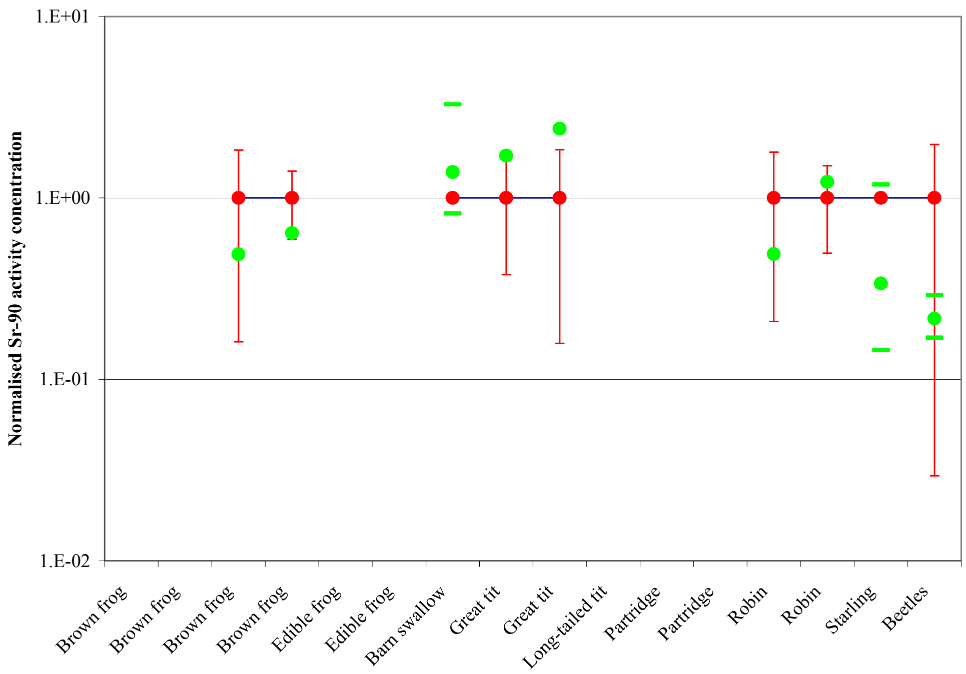

Figure 6.7 Comparison of observed (red) whole-body ^{90}Sr activity concentrations with those predicted using the default ERICA CR values (green) for amphibian, bird and invertebrate species. Mean, minimum and maximum observed and predicted values are presented when available.

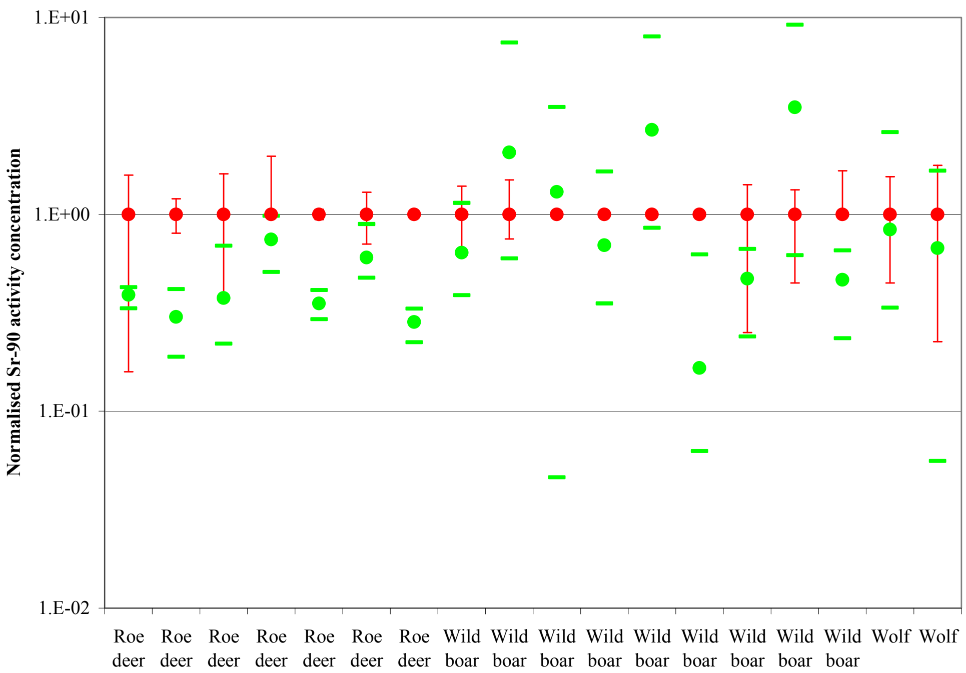

Figure 6.8 Comparison of observed (red) whole-body ^{90}Sr activity concentrations with those predicted using the default ERICA CR values (green) for large mammalian species. Mean, minimum and maximum observed and predicted values are presented when available.

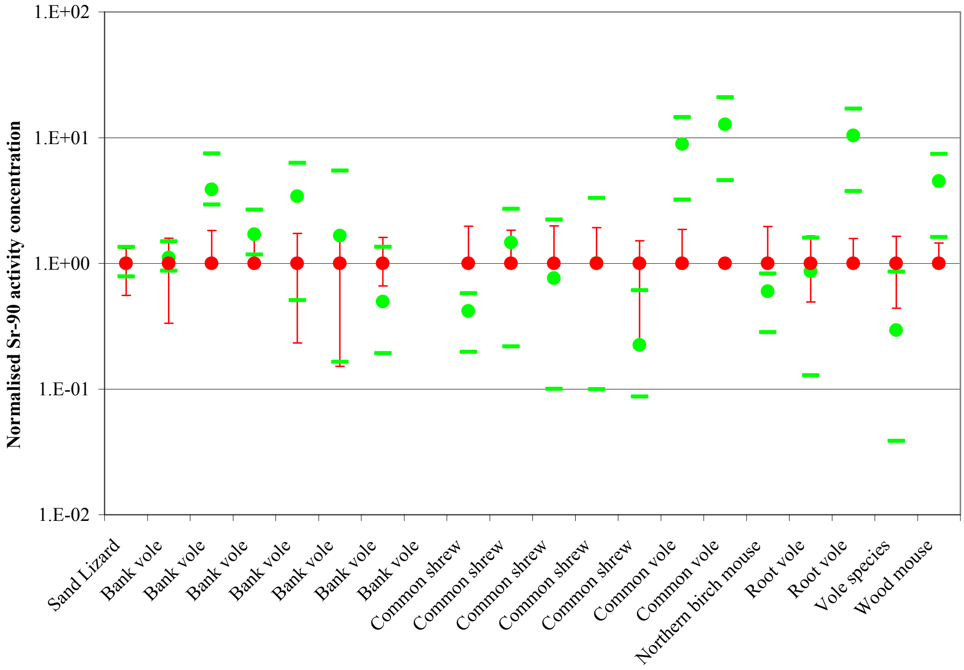

Figure 6.9 Comparison of observed (red) whole-body ^{90}Sr activity concentrations with those predicted using the default ERICA CR values (green) for small mammalian and reptile species. Mean, minimum and maximum observed and predicted values are presented when available.

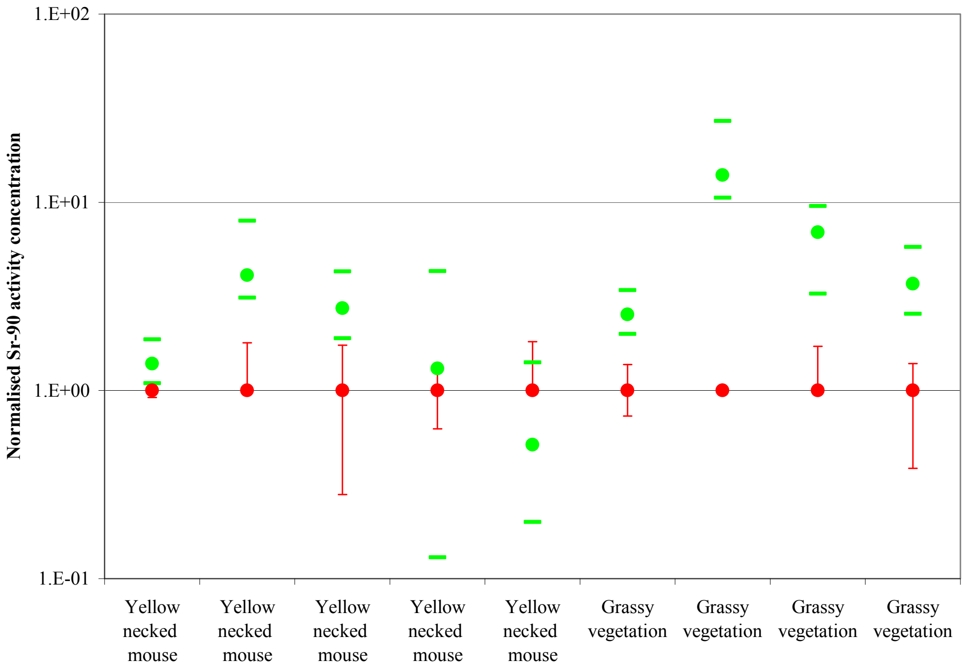

Figure 6.10 Comparison of observed (red) whole-body ^{90}Sr activity concentrations with those predicted using the default ERICA CR values (green) for small mammalian and vegetation species. Mean, minimum and maximum observed and predicted values are presented when available.

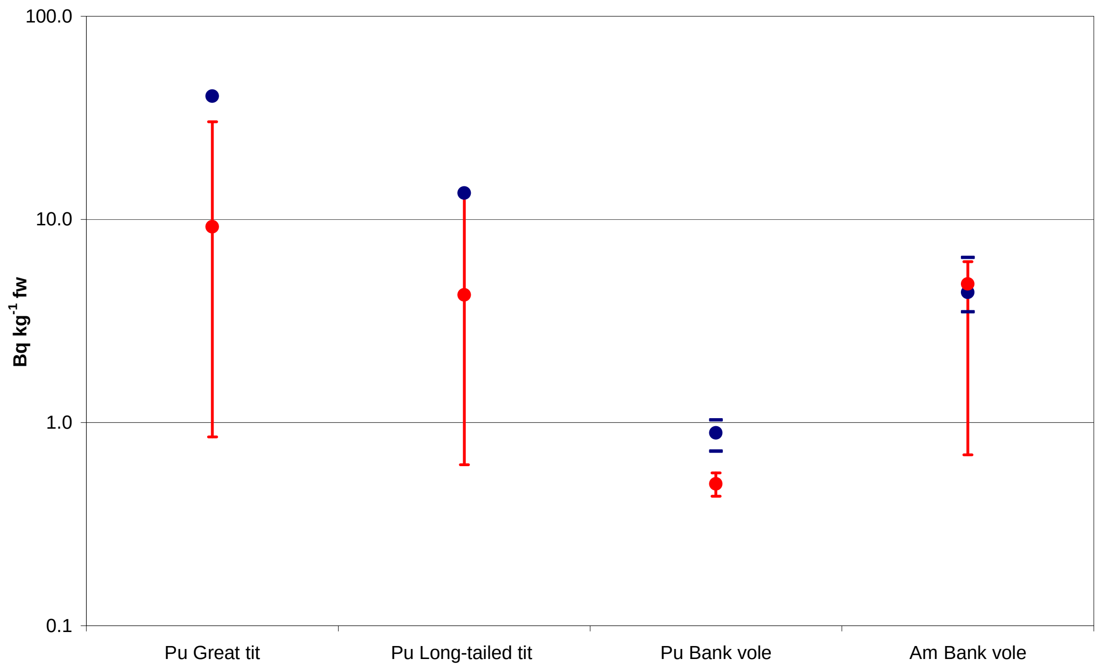

Figure 6.11 Comparison of observed (red) whole-body ^{239}Pu and ^{241}Am activity concentrations with those predicted using the default ERICA CR values (blue) for Bird and Small mammal species. Mean, minimum and maximum observed and predicted values are presented when available.

6.4 Discussion

The mean whole-body ^{137}Cs and ^{90}Sr activity concentrations predicted using CR values from the ERICA Tool were typically within an order of magnitude of measured values for a wide range of terrestrial biota. The few available actinide values were predicted to within a factor of 5 of the available data. The 95th percentile predictions of ^{137}Cs and ^{90}Sr whole-body activity concentrations in small mammals were lower than a few observed data when predicted using the probabilistic option available at Tier 3. Whilst the degree of agreement between measurements and predictions is encouraging it should be noted that the ERICA default CR databases contain some measurements from within the Chernobyl exclusion zone and hence this cannot be considered a truly independent test of the ERICA parameters.

The mean predicted (both deterministically and probabilistically) external doses rates were within a factor of two of measured (TLD) values. However, individual dose rates were not well predicted (see Figure 6.2) and maximum observed values were, in a number of instances, 2-3 times higher than the 95th percentile predictions. This may be because the number of soil samples taken in the study described above were insufficient to adequately describe the spatial variation in contamination of soil making individual predictions uncertain (no spatial trends were evident in soil activity concentrations).

Gamma air kerma rate measurements (here determined at 5 cm above soil surface) were in good agreement with external gamma dose rates estimated from TLDs (see Table 6.3). Air kerma rates could therefore be used to validate predicted dose rates during assessments and this could be included as advice to the user within the ERICA documentation.

In section 1.1 we noted that ^{90}Sr external DCCs used within the ERICA Tool were considerably lower than those used in other approaches (see Vives i Batlle et al., (submitted)) as a consequence of the

ERICA approach including a shielding effect of fur/skin. As only external gamma dose rates have been estimated here for comparison to the TLD results external doses from ^{90}Sr were not estimated. However, it is worth noting that external dose rates estimated using DCC values presented in Vives i Batlle et al. (submitted) for other available approaches (namely Copplestone et al. (2001) and USDOE (2002)) would result in external dose rates for ^{90}Sr to small mammals at study sites discussed in section 6.2 above similar to those for ^{137}Cs compared to estimates *circa* 6 orders of magnitude lower estimated using the ERICA DCCs. This highlights the requirement for the ERICA documentation to state the effect of the method of estimation of external beta dose rates compared to other available approaches.

6.4.1 Comments on ERICA Tool and D-ERICA

The nature of the Chernobyl case study was such that the content of D-ERICA was not tested.

Shortcomings of the Tool identified within this case study application were:

- Restrictions on the size of terrestrial organism did not allow a number of the required geometries to be defined (i.e. wolves and small birds)
- The geometry details input by the user to create organism cannot subsequently be viewed
- Outputs of total external and internal dose would be useful to the user for Tier 3 predictions to compare to the TLD results above this could only be achieved by setting CR values to zero.
- Report format is not user friendly (and does not print legibly)
- The retention of user inputs when altering an assessment is rather sporadic (*e.g.* when moving between Tiers, adding extra radionuclides to the assessment etc.)
- The 'created organism function' generates internal low beta DCC values which are set to zero for default organisms. Whilst those generated are low (and did not have to be used in this case study) the approach is inconsistent
- The user input seed value within Tier 3 assessments is incorrectly reported
- Users will not be given the opportunity to analyse the CR databases as performed here to derive rodent specific CRs

6.5 References

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7 Discussion

The case studies described within this report have enabled different aspects of the ERICA Integrated Approach and associated ERICA Tool to be assessed. For instance:

- both the Sellafield Marine and Drigg Coast Dunes case studies facilitated a full application of the Integrated Approach at sites receiving discharges from a nuclear licensed site (participants playing the role of assessors);
- the Chernobyl case study provided data with which to compare external dose rate predictions;
- the Komi case study concentrated on natural radionuclides and included effects data with which to compare to the outputs of the FREDERICA database;
- the Loire River and Sellafield Marine case studies included comparisons of predictions from the SRS-19 models included within the Tool with bespoke models parameterised for the study sites.

In this chapter we discuss the overall comments on the ERICA Integrated Approach and the performance of the Tool against datasets and bespoke models. An important component of the chapter is the feedback on comments arising from the case studies by the ERICA Consortium including, where appropriate, details of changes already made (or agreed to be made prior to the project's conclusion) to the Tool and/or D-ERICA.

Additional feedback from the case studies can be found within Appendix A. These are predominantly Tool related issues, some of low priority. Whilst some of these have already been responded to (see Appendix A), we recommend any outstanding comments related to uncertainty on Tool functionality to be addressed prior to the project's conclusion.

An important objective of the case studies was to provide feedback to the ERICA Consortium on areas for improvement. Consequently, many of the comments concentrate on the 'negative' aspects of the Tool and D-ERICA as released to the case study participants (i.e. identifying those areas which could be improved). **However, this should not be taken as undue criticism of the overall ERICA Integrated Approach and Tool.**

7.1 Case study versions of D-ERICA and the ERICA Tool

The case study participants had previously had differing levels of involvement with the development of the ERICA Integrated Approach and Tool. The prototype version of the ERICA Tool (as available December 2006) provided for case study applications was the first to have a Help file which was not yet linked to all screens (e.g. 'create organism'). This, together with a lack of a search function within Help or the ability to print the Help file (both of which we recommend are included in future versions of the Tool[3]), undoubtedly influenced the ability of some users to understand some aspects of the Tool and underlying methodology. The draft version of the accompanying documentation (D-ERICA, Larsson et al. 2006) which was intended to introduce the Tool and outline how it, and the underlying science, could be used was repetitive of the Help text, and did not adequately add the required background understanding and advice. As a consequence of interaction with the ERICA end user group (EUG), an early decision to radically revise the initial draft of D-ERICA was taken (Zinger, 2006). However, the lack of a more refined D-ERICA for case study participants again influenced some of the feedback to the Consortium (most especially by those participants less involved with the Tool's development). A good example of this is confusion expressed within the Loire study with

[3] The final Tool release will include a printable version of the Help file (which will be searchable).

regard to organisms living in sediment being the limiting organisms in Tier 1 evaluations when sediment concentrations were entered as zero (see Section 3.3).

7.2 The Tiered Approach and screening dose rate

The case study reports (especially Drigg Coast Dunes and Sellafield Marine) contain a number of comments on the tiered assessment and screening dose rate adopted by the ERICA Integrated Approach. These included:

- A lack of clear description, purpose and exit routes for Tiers 2 and 3 leading to comments such as:
    - What does Tier 3 add to an assessment that has 'failed' at Tier 2?
    - A likely outcome of exceeding conservative risk quotient values at Tier 2 is a de facto dose limit of 3.3 $\mu Gy\ h^{-1}$ being used (as a consequence of an uncertainty factor of three being used).
- Conservative dose rates estimated at Tier 2 cannot always be assumed to be as conservative as stated.
- The derivation of the recommended ERICA Integrated Approach screening dose rate limit was poorly described.
- Comments on exceedance of risk quotients due to natural radionuclides.

Following comments resulting from case study applications, and also the EUG (see Zinger, 2006), D-ERICA has been extensively rewritten (Beresford et al., 2007a) to try to address these comments as described below.

7.2.1 Incremental screening dose rate

Firstly, the documentation provided to the case study participants did not explicitly (or even implicitly) state that the ERICA Tool should be used to assess incremental doses and indeed the screening dose rate should have been defined as the 'incremental screening dose rate'. The incremental screening dose rate is not applicable to background exposure to natural radionuclides (an approach analogous to that taken within human radiological protection). Therefore, in the Drigg Coast Dunes case study natural radionuclides (noted as resulting in risk quotients in excess of unity) should not have been included as inputs into the Tool. On the basis of the methodology as described at the time, this was not apparent to the user. As the background exposure summary tables included within Tier 2 show dose rates in excess of 10 $\mu Gy\ h^{-1}$, it is not surprising that, if included within an assessment, natural radionuclides will significantly contribute to the resultant risk quotient.

In addition to addressing this, the revised D-ERICA (and Tool Help file) now also more clearly explains the methodology used to derive the screening dose rate.

7.2.2 Tiers 2 and 3

The description of the purpose of Tiers 2 and 3 and interpretation of their results has now been more clearly documented in light of the comments received. Whilst intended as a screening tier, Tier 2 allows the user to be more interactive than Tier 1 (*e.g.* to change the default parameters and to select specific reference organisms etc.). The evaluation is performed directly against the screening dose rate, with the dose rates and risk quotients generated for each reference organism selected for assessment. A 'traffic light' system is used to indicate whether the situation can be considered:

I) of negligible concern (with a high degree of confidence) if both the best estimate and conservative risk quotients are < 1;

II) of potential concern if the conservative risk quotient is ≥1 but the best estimate risk quotient is < 1, where more qualified judgements may need to be made and/or a refined assessment at Tier 2 or an in-depth assessment in Tier 3 performed;

III) of concern when both risk quotients are ≥1, where the user is recommended to continue the assessment either at Tier 2 if refined input data can be obtained or at Tier 3.

In those cases where is it recommended that the assessment be continued, or that the assessment and results are reviewed, it does not necessarily mean an automatic progression to Tier 3 is required. For instance, it may be possible to refine the input data or Tool parameters (*e.g.* obtain CR values applicable to the site) if justifiable and to then rerun the assessment at Tier 2. In instances where the conservative risk quotient is above one, whilst the best estimate risk quotient is below one, interpretation of the results may lead to a decision that the assessment can be justifiably exited (e.g. if incremental dose rates are similar to dose rates due to background levels of naturally occurring radionuclides).

Situations which give rise to a Tier 3 assessment, are likely to be complex and unique, and it is therefore not possible to provide detailed or specific guidance on how the Tier 3 assessment should be conducted. Furthermore, a Tier 3 assessment does not provide a simple 'yes/no' answer, nor is the ERICA-derived incremental screening dose rate of 10 $\mu Gy\ h^{-1}$ appropriate with respect to the more specific assessment endpoints that should have been defined by this stage. The requirement to consider aspects such as the biological effects data within the FREDERICA database, or to undertake ecological survey work, is not straightforward and requires an experienced, knowledgeable assessor or consultation with an appropriate expert. Tier 3 is a probabilistic risk assessment in which uncertainties within the results may be determined using sensitivity analysis. The assessor can also access up-to-date scientific literature (which may not be available at Tier 2) on the biological effects of exposure to ionising radiation in a number of different species. Together, these allow the user to estimate the probability (or incidence) and magnitude (or severity) of the environmental effects likely to occur and, by discussion and agreement with stakeholders, to determine the acceptability of the risk.

7.2.3 Conservatism at Tier 2

The aim of Tier 2 is to identify situations where there is a very low probability that the dose to any selected organism exceeds the adopted screening dose rate. To achieve this, in addition to estimating a best estimate dose-rate and risk quotient, conservative values are also calculated. The conservative values are either the 95th or 99th percentile of the risk quotient estimated by multiplying the expected value by an uncertainty factor of 3 or 5 respectively. The uncertainty factor is defined as the ratio between the 95th or 99th percentile and the expected value of the probability distribution of the dose rate (and risk quotient) assuming exponential distributions.

Conservative estimates of dose output by Tier 2 if an uncertainty factor of three is used would be expected to be similar to 95th percentile predictions if Tier 3 is used. The Chernobyl case study demonstrates that at three sites contaminated with a range of radionuclides conservative dose rates estimated for three rodent species using Tier 2 are similar to, or higher than, the 95th percentile predictions of Tier 3 (see Table 6.5). However, for the one organism (mollusc) for which this assumption was tested during the Sellafield Marine case study the 95th percentile risk quotient estimated by Tier 3 (defined by the participants as the 95th percentile dose rate relative to the incremental screening dose rate), based on input water concentrations only, were 1.5 to 3 times higher than the conservative risk quotient estimated by Tier 2 (depending upon the distribution assumed for input water activity concentrations) (see Table 4.14). However, if the sediment data were used as the input the 95th percentile prediction of Tier 3 was comparable to the Tier 2 conservative estimate (see Table 4.15). The water concentrations, and associated probability distribution functions, were derived from a contour map and the chapter authors note that this may contribute to their results. However, the observation does suggest that there is a requirement to further test the conservatism assumed in Tier 2.

One of the case study reports makes the comment that exceeding the conservative risk quotient values at Tier 2 is a *de facto* dose limit of 3.3 μGy h^{-1} (if using an uncertainty factor of three). However, the application of uncertainty factors does not reduce the screening dose rate rather it provides a conservative risk quotient which is based upon the 95th percentile dose rate prediction. This is not suggested as a revised benchmark, rather to highlight cases of potential concern as summarised above. However, the conservative risk quotient was not adequately described within the documentation made available to case study participants; the revised Tool Help and D-ERICA provide a clearer explanation.

7.2.4 'Lichen and Bryophyte' reference organism

The 'Lichen and Bryophyte' reference organism is the limiting organism for a number of radionuclides (predominantly natural series isotopes). For example, Lichen and Bryophyte is the limiting organism for ^{210}Po, the associated EMCL value of 25 Bq kg^{-1} DW soil is the lowest terrestrial EMCL value (see Chapters 2 and 5 for discussion of this within assessments) and falls within the activity concentration range for ^{210}Po occurring naturally in soils. The predominant reason for this reference organism sometimes being the limiting organism is the comparatively high CR values within the ERICA Tool database. Both lichens and bryophytes receive their nutrients, and pollutants, as particulates and in solution from atmospheric deposition (Burton, 1986). The use of a soil-biota CR may not, therefore, always be applicable to this reference organism. It is also possible that some of the contamination of lichen and bryophyte samples is external deposit and not internal contamination. If this is the case the dose-rate per unit activity concentration would be reduced and the EMCL increased (as these parameters are inversely correlated) especially for an alpha-emitter such as ^{210}Po. Whilst FREDERICA contains no chronic dose effects data for lichens or bryophytes, acute exposure data (for mortality) suggest that they are amongst the least radiosensitive of organisms (UNSCEAR, 1996). The implementation of a predicted no effects dose rate (as used to define the screening dose-rate at Tiers 1 and 2) that has been derived to be protective of all organism types within terrestrial ecosystems may therefore be overly conservative if applied specifically to lichens and mosses.

These factors should be taken into account when making judgement on the outputs of Tier 2 and 3 assessments if 'Lichen and Bryophyte' are identified as being organisms of concern.

7.3 Transfer parameters

7.3.1 Terrestrial ecosystems

The three terrestrial case studies enable comparison of predicted biota activity concentrations with measured values for a range of radionuclides in different ecosystem types.

The mean whole-body ^{137}Cs and ^{90}Sr activity concentrations predicted using CR values from the ERICA Tool were typically within an order of magnitude of measured values for a wide range of terrestrial biota assessed within the Chernobyl case study. The few available actinide values were predicted to within a factor of 5 of the available data. The 95th percentile predictions of ^{137}Cs and ^{90}Sr whole-body activity concentrations in small mammals were lower than a few observed data when predicted using the probabilistic option available at Tier 3 (although predicted maximum values were in excess of the observed maximums). Whilst, the degree of agreement between measurements and predictions is encouraging the ERICA default CRs, the database contains <u>some</u> measurements from within the Chernobyl exclusion zone and hence this cannot be considered a truly independent test of the ERICA parameters.

Measured activity concentrations of ^{226}Ra were within the ranges predicted using default CR values (and distributions) within Tier 3 of the Tool at the Komi case study areas for grasses, herbs and shrubs. However, activity concentrations of ^{226}Ra in trees were under-predicted by the ERICA assessment Tool. Note that the ERICA Tool default ^{226}Ra CR value for 'Tree' is derived from Canadian studies from data for twigs, needles and leaves (i.e. as was available for comparison within the Komi case study). For ^{232}Th and ^{238}U there was reasonable agreement (mean values being within an order of magnitude) for all vegetation types at the Middle Timan site (see Table 5.3). However, at the North Ural site predictions tended to be high compared to measured values, with the predicted mean often being in excess of the predicted upper range (see Table 5.6). The use of soil ash weight activity concentrations in assessments at all Komi sites would result in predictions being comparatively high. There were only very limited data with which to compare predicted and observed activity concentration in mammals (Tundra voles) within the Komi case study (and that available had no information on distributions or sample numbers; see Table 5.12). This limited comparison inferred that whilst predictions for ^{226}Ra were in reasonable agreement, ^{238}U and ^{232}Th were under-predicted. The ERICA Tool CR values for Th and U have also been applied to predict likely activity concentrations in a range of biota in England and Wales (Beresford et al., 2007b). Agreement with available measured data was generally encouraging (predicted and measured ranges overlapping) (see Figure 7.1). However, the default CR values for Th and U for some reference organisms (e.g. mammals) are dominated by data from the United Kingdom, although for others (e.g. birds) they are derived from data from the Komi Republic (Beresford et al., 2005). The Komi case study has provided a considerable amount of data which could be used to refine some of the ERICA Tool default CR values at a later date. However, the transfer of some naturally occurring radionuclides from highly contaminated sites may be expected to be non-linear (i.e. CR may vary with contamination level at these sites) (e.g. see Sheppard and Evenden (1988)); the available Komi data have not been analysed to test whether this is the case across the study areas.

Caesium-137 activity concentrations are consistently over-predicted for all organism included within the Drigg Coast Dunes comparison (see Tables 2.6 and 2.7). The majority of the ERICA default CR values for Cs originate from post Chernobyl studies. As such, many will be for ecosystems with more organic soil types with a comparatively high transfer of radiocaesium to biota. These are unlikely to be representative of the very sandy soil at this case study site. Note that CR values for Cs to mammals which can be derived from the data presented in the case study for this site are similar to those estimated from Copplestone (1996) for a sand dune site close to the Drigg Coast Dunes (CR value = 1.3×10^{-2} *cf.* the ERICA Tool default[4] of 2.87). For ^{241}Am, ^{90}Sr, $^{239+240}$Pu and ^{99}Tc at the Drigg Coast Dunes predicted whole-body activity concentrations in all animal types were in reasonable agreement (see Table 2.6). However, activity concentrations of ^{241}Am in higher plants, especially grass species, were under-predicted. As noted within the case study report, this is likely to be because the site experiences aerial deposition via sea-to-land, whereas, the ERICA Tool default CR values aim to predict transfer from soil.

The organism-radionuclide combinations considered within the Drigg Coast Dunes case study include some of the most poorly represented combinations within the ERICA Tool CR database (because of the lack of reported data) with many not being based upon data (the guidance methodology described within D-ERICA having been used to derive default values). These include: Am and Pu CR values for birds, amphibians and reptiles; all Tc CR values with the exception of 'Grasses and Herbs' and; the Cs CR value for reptiles. The reasonable agreement between predictions and observations for most of these organism-radionuclides (see Table 2.6) is therefore encouraging for the approach taken to provide default CR values within the ERICA Tool when empirical data were unavailable. Some of the data obtained from the case study have subsequently been included within the ERICA default database and used to derive new environmental media concentration limits which will be in the final release version of the Tool.

[4]As of February 2007

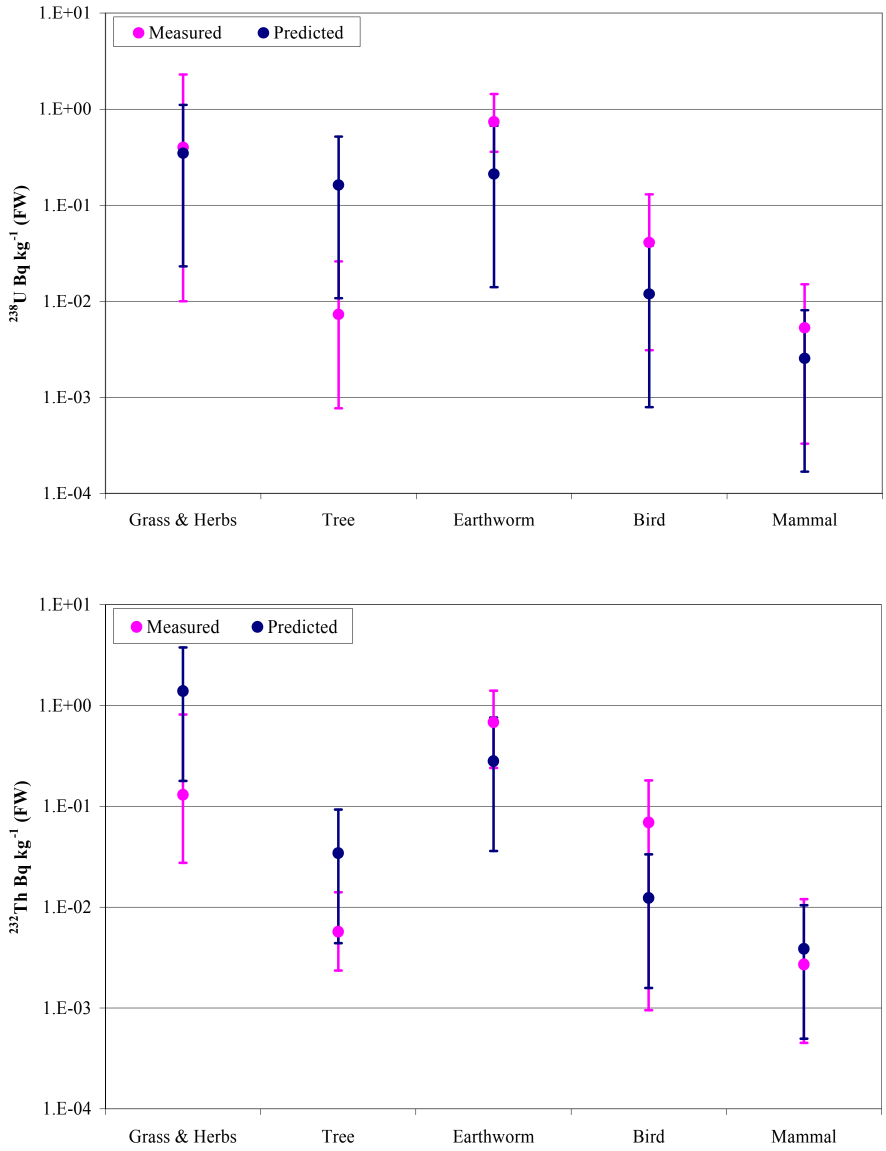

Figure 7.1 A comparison of observed ^{238}U (top) and ^{232}Th (bottom) activity concentration in a range of biota within England and Wales compared to predictions using the ERICA Tool CR values (adapted from Beresford et al., 2007b). Note media concentrations were the mean, minimum and maximum values for 25 km^2 over England and Wales.

7.3.2 Freshwater ecosystems

Following criticism on the source of some of the default values within the Tools freshwater K_d database (see Loire River case study) a number of values have been updated using outputs of the on-going IAEA EMRAS review to update the IAEA transfer parameter handbook (IAEA, 1994). Probability distribution functions were also available for these updated K_d values.

There was no opportunity to assess the predictions of freshwater CR values within the case studies. However, these will be evaluated, and openly reported, for a wide range of biota (for Sr, Cs, Co and ^{3}H) within the IAEA EMRAS Biota Working Group (see Yankovich, 2005).

7.3.3 *Marine ecosystems*

For the Sellafield Marine case study, predictions of ^{137}Cs, ^{241}Am and $^{239+240}Pu$ activity concentrations in a range of marine biota using the Tool default CRs were in reasonable (often within one order of magnitude) agreement with measured data (see Table 4.5). There was a considerable over-prediction of Pu activity concentrations in fish. However, observed data were for muscle only whereas the CR values predicted whole-body activity concentrations (the majority of the Pu body burden is in liver and bone). Caesium-137 activity concentrations in seabirds were predicted to be almost 500 times greater than observed data. However, the observed data were for gulls within an estuarine environment which are likely to have been feeding in terrestrial ecosystems.

Predicted water concentrations were within an order of magnitude of observed data for locations relatively near to the discharge point along the Cumbrian coast.

7.3.4 *Agreement between predictions and observations*

Overall, there was an acceptable level of agreement between predicted and observed activity concentrations within biota. Where this was not the case, reasons can often be put forward to explain this (e.g. see discussions on Cs and Am for Drigg Coast above). Note however, D-ERICA now acknowledges that it is likely that the default CR and K_d databases included within the ERICA Tool will not be applicable to some ecosystems (such as saltmarshes).

The derivation of transfer parameters within the ERICA Tool is generally well documented (and identified to the user) in a manner that makes it more transparent than other existing approaches.

A number of the case study reports commented that predictions were considered acceptable if they were within an order of magnitude of the observed data taking into account the large variability within many of the CR databases and that often few observed data were available for comparison. This was taken into account when trying to provide advice to assessors comparing site-specific data to predictions within the revised D-ERICA: *A reasonable level of agreement would be for predicted and observed data to fall within an order of magnitude of each other (consistent with the approach taken in the development of the SRS-19 screening models). However, if there is consistent under- or over-prediction at a given site, alternative transfer parameters should be considered or sufficient measurements of biota conducted.*

Whilst the case studies provided an opportunity to evaluate default CR and K_d values for a range of radionuclides and ecosystems it was not possible to consider all (indeed the majority) of the Tool's CR and K_d values. Any future opportunities to further evaluate the Tool's transfer database should therefore be exploited to further confidence in predictions and highlight any areas requiring improvement.

7.4 Dosimetry

The Chernobyl case study demonstrated that predictions of mean external dose rate by the ERICA Tool agree well (within a factor of two) with measurements from TLDs attached to a number of rodent species at three sites (see Table 6.4). Some maximum measured (TLD) values were above 95th percentile predicted values although this may have been due to insufficient soil sampling to determine spatial variation at the sites (and indeed uncertainties in interpretation of the TLD readings). Gamma air kerma rates (determined at 5 cm above soil surface) were in good agreement with predicted dose rates and TLD results (see Table 6.3). Comparison of predicted external dose rates and gamma air kerma rates at the Komi case study areas also showed reasonable agreement (see Table 5.9). Gamma air kerma rates will often be available (generally measured at 1 m above soil surface) for sites being assessed. On the basis of observations during the Chernobyl and Komi case studies, it is suggested that future revisions of the ERICA Tool include within the Help file advice that gamma air kerma rates can be used to validate dose rate predictions. Furthermore, future developments of the Tool should consider enabling the use of gamma air kerma measurements as assessment inputs. Table 7.1 presents conversion factors which could be used to convert gamma air kerma measurements to external dose rates for different organisms and included within the Tool Help file.

Table 7.1 A comparison of estimated external dose rates to gamma-air kerma rates for organisms of different sizes (G. Pröhl (GSF) pers comm.).

	External dose rate : gamma-air kerma rate (Gy:Gy)					
Energy (MeV)	Woodlouse (detritivorous invertebrate)	Mouse	Rabbit	Fox	Deer	Cow
0.05	1.01	1.00	0.89	0.81	0.56	0.12
0.3	1.03	1.02	0.90	0.84	0.67	0.26
0.662	1.04	1.03	0.92	0.86	0.69	0.28
1	1.05	1.04	0.93	0.88	0.71	0.30
3	1.05	1.05	0.98	0.93	0.80	0.43

7.4.1 Defining new organisms

The Tool allows the user to create their own organism within Tiers 2 and 3 assessments if required. However, there are various restrictions in allowable mass for created terrestrial organism:

- 0.0017 to 550 kg for animals living above ground;
- 0.0017 to 6.6 kg for animals living in a homogenously contaminated soil layer;
- 0.035 to 2 kg for birds (or flying animals).

This limits the usefulness of the 'create organism' functionality of the Tool. For instance: the lower limit on flying animals does not allow representative geometries to be created for many bird species, probably all European bat species and flying insects; the upper limit on animals living within the ground does not enable the creation of geometries for larger mammals, such as badgers or wolves. Furthermore, there is no ability to assign aquatic birds to an air habitat. These limitations were not acknowledged within the documentation (or Tool screens) provided to case study participants. Whilst it would have been preferable to have been able to create any geometry in any habitat, in the absence of this functionality the revised Tool Help file clearly documents limitations and provides advice on the implications for estimated dose rates and approaches to best try to model user defined organisms. The Tool screen for inputting user-defined organism geometry information also provides guidance on allowable mass ranges for different organism type.

A further criticism by case study participants was that there was no advice on when a user needed to create their own organism nor were details of the ERICA default geometries provided. The Tool Help file now contains a table of the ERICA default geometries, enabling users to compare these with any organisms of interest within their assessment. Some advice on the effect of changing organism size for different radiation types on internal and external dose rates is now also given.

Default dose conversion coefficients for a number of radionuclides and reference organism have been compared to those used within other approaches within the IAEA EMRAS Biota Working Group (see Chapter 1) (Vives i Batlle et al., submitted). Overall, the exercise demonstrated that all 11 participating approaches, including ERICA, compared favourably despite a wide range of assumptions being made. The only exception of note to the ERICA Integrated Approach was that external DCCs for ^{90}Sr as used within the Tool for terrestrial organisms were low (by 4-7 orders of magnitude) compared to other participating models and approaches. This was thought to be the consequence of the

ERICA methodology assuming a shielding skin/fur layer which was not considered within the other approaches. This is now noted within the Tool Help file.

7.5 Application of SRS-19 models

The SRS-19 screening transport models provided within the ERICA Tool (adopted from IAEA (2001)) are designed to minimise the possibility that the calculated media concentrations will underestimate doses (to humans) by more than a factor of 10. They are intended for the estimation of average concentrations in water or air from continuous releases from a single source assuming that an equilibrium, or quasi-equilibrium, has been established between released radionuclides and the environmental medium. Compared to the outputs of a bespoke model parameterised for the Irish Sea the SRS-19 coastal model provided mostly conservative estimates of activity concentrations in water and sediment for a range of radionuclides. However, the Sellafield Marine case study concluded that application of the SRS-19 coastal model to distant sources was questionable when the source-receptor distance is not in the direction of the long-shore current. In the case study application this resulted in very conservative estimates of seawater concentrations (compared to those of the bespoke model) and hyper-conservative estimates for particle-reactive radionuclide (*e.g.* Ce, Cm, Nb, Ru) activity concentrations in sediments (see Table 4.3). As a consequence it was recommended that the Tool Help file be amended to warn the user of the potential inadequacies of the SRS-19 model when applied to distant sources and this has been done. An exception to the conservative nature of predictions of the SRS-19 model in combination with ERICA default K_d values was the predictions of ^{3}H concentrations in sediment which were in some instances predicted to be lower than by the bespoke model. This must be the consequence of a considerably lower K_d value being used within the ERICA Tool compared to the other model.

Predictions using the SRS-19 river model within the Loire River case study again provided generally conservative (by a factor of *circa* 5-10) media activity concentrations and risk quotients compared to a transport model parameterised for the river. The notable exception was the under-prediction of ^{3}H concentrations in water by > 2 orders of magnitude by the SRS-19 model compared to the bespoke model. The reasons for this discrepancy in the model-model comparison are unclear although the assessor suggests it may be the consequence of the time-averaging used to derive model inputs.

Overall, the application of the SRS-19 models included in the ERICA Tool are likely to result in conservative assessments although it was only possible to test two of the six transport models available. However, prediction of ^{3}H by both the river and coastal transport models were not always conservative compared to the predictions of the bespoke models to which they were compared.

7.6 Effects summary tables and FREDERICA

A number of the case studies criticised the usefulness of the effects summary tables available within Tier 2. For some wildlife groups the summary tables contain little, or no data, whilst for others the available data are contradictory. An actual summary of the available data by dose rate category would be more useful than the list of all observations currently available. Furthermore, for comparison with the screening dose rates and conservative dose estimates at Tier 2 initial dose rate bands of 0-10, 10-30 and 30-50 $\mu Gy\ h^{-1}$ may be more useful than the existing 0-50 $\mu Gy\ h^{-1}$ category.

The link to the FREDERICA database at Tier 3 was described as of little aid to practical decision-making as the resulting information required expert interpretation (see Komi and Sellafield Marine case studies). However, by nature situations which give rise to a Tier 3 assessment, are likely to be complex and it is acknowledged that the analyses of information within the FREDERICA database is likely to require consultation with an appropriate expert. The FREDERICA database is an up to date, freely accessible, database on radiation effects in non-human biota and provides a useful tool to scientists (as demonstrated by its use outside of the ERICA Consortium (*e.g.* Chambers et al., 2006)).

7.7 Future development and testing of the ERICA Tool

Whilst the ERICA project will be completed by March 2007, the Consortium is negotiating an agreement to manage potential developments of the ERICA Tool (and D-ERICA) over at least the next three years. Update versions of the Tool will continue to be freely available; comments on the Tool by users outside of the consortium will be considered within its revision..

The Tool and its databases will also continue to be used within the IAEA EMRAS Biota Working Group comparing predictions to those of other models (outputs of these exercises will become available prior to the end of 2007; see http://www-ns.iaea.org/projects/emras/emras-biota-wg.htm). The ERICA Integrated Approach and Tool will also be evaluated, together with other approaches, within the PROTECT project (see http://www.ceh.ac.uk/PROTECT/package2/ for details and progress).

7.8 References

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ANNEX A: Additional feedback

The following table list additional comments from the case study evaluation of the ERICA Tool (including pre-trial versions of the Tool) not discussed in detail within the main report. An indication of likely response to the comment is indicated if known (as of 26/02/07).

Component	Comment	Status
	Help	
	A search function within the Help files is required as demonstrated by the number of response to the case study comments referring to the revised Tool Help.	The Tool will contain the Help as a pdf file for printing – this will be searchable.
	Dosimetry	
DCCs	The external DCCs for 'lichen & bryophytes' appear to be very low compared to all other reference organism categories – **is this correct or an error** ?	It was an error - the external DCCs have been revised and EMCLs will be corrected.
Create organism (Help)	Unclear when CR values should be attributed to new organisms.	Now in Tool Help.
Help	The default reference organism geometries should be available to the user to enable them to judge if new organisms should be created. Whilst the help files present chi and ksi for most users the values of height, width and length would be more useful.	Now in Tool Help.
Create organism	Once an organism is created the user cannot check the geometry values they have input.	Designated as low priority*.
Create organism	Generates internal low beta DCC values which are set to zero for default organisms. Whilst those generated are low (and did not have to be used in this case study) the approach is inconsistent	Designated as low priority*.
Help	Table H5 – ^{228}Ac (t1/2 = 0.25 d) should be included as a daughter of ^{228}Ra – (t1/2 = 5.8 y) this is not listed within table. Is it included?	^{228}Ac is included as a daughter product of ^{228}Ra. Now listed in table.
	Risk	
EMCLs	For terrestrial ecosystems Tier 1 RQs are exceeded at background levels for some natural radionuclides including ^{226}Ra and ^{210}Po (note Tier 2 assessments give much lower RQ values).	Ra and Po CR values had been revised (to lower values) in the case study release of the Tool. However, new EMCL values had not been estimated for Case Study release version. Final release includes revised EMCL (which are also discussed in Chapter 5 of this report)
ECMLs	Chemical toxicity of U and Th isotopes is not considered. This may lead to an underestimation of risk (*e.g.* for the Komi case study sites contaminated by U-Ra).	Some discussion now in Help file.
	Inputs	

	Would be useful to retain user input concentrations as move between tiers.	Note: Tier 1 requires maximum activity concentrations as input, Tier 2 requires best estimates. Hence retention of inputs is not applicable.
Tier 3	Unclear what is required for mu and sigma – replace with arithmetic mean and standard deviation if that is what is meant.	mu and sigma are arithmetic mean and standard deviation. Final release version will use arithmetic mean and standard deviation (Help file will need amending).
Help (Tier 3)	How does user decide on which distribution function to use – provide Help text on when various distributions can be used with some basic references.	Help now has examples of all distribution types
Help	Additional comment could be provided within Help that the 'adjacent ecosystem approach' can be modified and applied to complex assessments for which considerable detail is provided with regards to the time spent by an organism at different locations with different levels of contamination. An alternative may be to suggest the derivation of a spatially averaged input.	Help amended accordingly
	Would like to be able to paste directly into the inputs – either as single data points or list of data (N.B. cntrl V does not paste)	This functionality will be available in Final release version.
Aquatic ecosystems	Order of sediment and water concentration differs between Tier 1 and Tiers 2 and 3. Consistency would be preferable.	Designated as low priority*.
Tier 1 aquatic ecosystems (Help)	No information on how input is handled if e.g. only input sediment concentrations v's inputting only water or both (see Chapter 3).	Tier 1 includes EMCLs for both sediment and water, the most conservative RQ being reported.
Weighting factors (Tier 3)	There is an option to have distributions on weighting factors within Tier 3 – why are no distribution parameters recommended?	
Tier 3 distribution values	Where no distribution parameters provided the default should be '0' – currently 1.0 for mu at least in some instances (*e.g.* for weighting factor).	Will not be changed (a default weighting factor of one will result in unweighted dose rate estimates).
Percentage soil or sediment dry weight	The input is % dry weight – the results and report present this (same value) as % moisture	Amended.
Occupancy factors	Columns can be re-ordered but the greyed out cells do not move accordingly	Will be rectified by removing the re-order functionality (which is of no value for this screen).
Tier 2 and 3	Be useful if CR and activity concentration input screens could have 1st column locked so that could always see which nuclide data is being entered for when scroll right.	Final release version should have first column locked.

Dose rate	Consider allowing dose rate as an input (e.g. from gamma air kerma measurements).	Will be considered for updated Tool version subsequent to the end of the ERICA project.
	Transfer parameters	
CR & K_d databases	Greater 'access' to databases may be usefully to the user in making decisions on what values to use. Examples are – information on number of observations (could this appear on the Tier 2 and 3 CR and K_d screens?); tabulated summarisation by sub-reference organism.	Help file states some of this information is available from the databases – it is not. Database or Help file to be corrected. Note it is intended to submitted papers to refereed literature on the transfer databases.
Freshwater K_d	Why no default value for H-3?	Now provided.
Marine K_d	Why no default value for P isotopes?	Value being identified.
CR Tier 3	Assumed distributions for those CR values selected using one of the CR codes should be explained. (NB selection of values for distributions is not helped by the number of significant figures associated with default values)	Now described in Help.
Tier 2 (&3) CR & K_d	Consider option of being able to by-pass CR and K_d screens if have a complete set of media and biota concentrations	To be investigated in updates of the Tool after the end of the ERICA project.
CR Tier 2 (& 3)	Rather than code letter it would be better to have descriptive text on screen	Done
	SRS-19 models	
Terrestrial	Only applicable to airborne releases. There is likely to be a requirement to use the ERICA methodology at sites with sub-surface contaminant inputs – the addition of a screening model that allows simulation of this route would be useful in any future (post-ERICA) editions of the Tool.	For consideration after the end of the ERICA project.
Freshwater (possibly others)	The purpose etc. of the 'estimate' option for some parameters needs to be explained	See revised Help.
Terrestrial	Confirm that predictions for H, C, S and P are activity concentrations in air and not soil. If they are soil are separate EMCLs required	Predictions in the case study test version of the Tool were for soil. Will be revised.
	Results	
Tiers 2 & 3	Values of total internal and external dose rates would be useful (especially at Tier 3)	Defined as medium priority+.
Background tables	Would be more useful if the predicted dose rates were presented on the table for comparison	Defined as low priority*.
Plots	Option for log-scale needed	Defined as medium priority+.
Tables etc.	Outputs cannot be copied directly from results screen. WSC have offered to refine an EXCEL add-in they have developed to extract results from Tier 2.	Defined as medium priority+.

Tier 1	Results are limited – would be of use to have results for organisms other than the limiting organism (the most exposed organism may not be identified by Tier 1).	The EMCLs have been derived for all reference organisms and nuclides external to the Tool. The lowest EMCL for each radionuclide has been input into the Tool (i.e. the Tool only contains an EMCL value for one organism for each radionuclide). Help and D-ERICA revised to explain.
Tier 3	No units on statistics screens	Will be corrected.
Tier 3	Effects – what are the lower and upper values [units]	See Help.
Tier 3 (sensitivity analyses)	Many of the correlations presented are meaningless and most of the rest obvious. Consider restricting to 'Total Dose'?	Sensitivity analyses will be restricted to total dose rate.
Tables & plots	Remove option to be able to view unnecessary parameters, e.g. zero is assumed for all external alpha and low beta DCCs. Removal would improve overall clarity of output windows.	Defined as low priority*.
Tables & plots	Some inputs repeated as outputs (*e.g.* media concentrations) – remove from outputs	Will be corrected.
	Other	
	If have more than one assessment open it is not obvious which you are in (especially as apparently changing between assessments – does not actually do so) – consider highlighting which assessment, and where in it, user is in left hand window.	Help file will be revised to better explain.
	Be useful if Tool 'remembered' last directory opened.	Defined as low priority*.
	Program behaves different between PCs, *e.g.*: (i) one group reported that next, back and help buttons disappear when maximised (not a problem for other groups); (ii) one user had no scroll bars on a number of screens (Help windows, report); (iii) some users report that some Tier 2 results are outside of window unless divider bar adjusted (not a problem for other groups).	Help file will as for any such faults to be reported listing information required to be submitted (e.g. operating system, available memory).
Stakeholder screens	Some of stakeholder categories are not easily understood by all e.g. next generation, non-human species – specify that these are appropriate representatives of such 'stakeholders'	Explained in Help.
Databases	Some of the abbreviations used for database names do not appear to be defined anywhere (*e.g.* ext-beta-gamma). NOTE these abbreviations are also used within the report.	Explained in Help.
General	Ability to print some screens would be useful.	Defined as low priority*.

*Defined as low priority – unlikely to be addressed for final release version of the ERICA project, may be addressed in subsequent updates;
+Defined as medium priority – will be addressed in final release version of the ERICA project if time always, otherwise will be addressed in subsequent updates.